Hunter self-monitoring and wildlife governance in an industrial forestry concession in the Republic of Congo: context, behaviour change, and wildlife monitoring

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Acronyms

ACFAP	-	*L'Agence Congolaise de la Faune et des Aires Protégées*
AFLEG	-	Africa Forest Law Enforcement and Governance
AGACL	-	*l'Association pour la Gestion de l'Aire Communautaire de Liouesso*
AM	-	*Ami du Monde*
APETDS	-	Association for the Promotion of Tropical Ecosystems and the Protection of the Environment
CACO-		
REDD	-	Consultation Framework for Congolese Civil Society and Indigenous Peoples of REDD+
CARPE	-	Central Africa Regional Program for the Environment
CBFP	-	Congo Basin Forest Partnership
CBNRM	-	Community Based Natural Resource Management
CIB	-	*Congolaise Industrielle des Bois*
CIFOR	-	Centre for International Forestry Research
CIRAD	-	Agricultural Research Centre for International Development
CIREK	-	International Circle of research of the Bakwele civilisation
CITES	-	Convention on International Trade in Endangered Species of Wild Fauna and Flora
COMIFAC	-	Central African Forests Commission
CPR	-	Common Pool Resource
CPUE	-	Catch Per Unit Effort
CSO	-	Civil Society Organisation
EC	-	European Commission
ECOFAC	-	Conservation and Rational Use of Forest Ecosystems in Central Africa
FAO	-	Food and Agriculture Organisation of the United Nations
FLEG	-	Forest Law Enforcement and Governance
FLEGT	-	Forest Law Enforcement, Governance and Trade
FMU	-	Forest Management Unit
FPIC	-	Free Prior and Informed Consent
FSC	-	Forest Stewardship Council

HSM	-	Hunter Self-monitoring
IFO	-	*Industrie Forestière d'Ouesso*
LDF	-	Local Development Fund
MEFDD	-	Ministry of Forests and Sustainable Development
NGO	-	Non-governmental Organisation
OCBE-Vert	-	*Universe of Ecosystem Defenders of Miélé-Kouka*
PGDF	-	Platform for the Sustainable Management of Forests of FLEGT
PNOK	-	Odzala-Kokoua National Park
REDD+	-	Reducing Emissions from Deforestation and forest Degradation
SAM	-	*Sangha Assistance Medical*
SCBO	-	*La Société Congolaise des Bois de*
SFAC	-	*La Société Forestière Algero-Congolaise*
SM	-	Self-monitoring
SNA	-	Social Network Analysis
TRIDOM	-	Tri-National Dja—Odzala—Minkebe Landscape
UDEMK	-	Observatory of Bantu cultures and Biodiversity and Environmental education
USAID	-	United States Agency for International Development
VPA	-	Voluntary Partnership Arrangement
WCS	-	Wildlife Conservation Society
WWF	-	World Wild Fund for Nature

Summary

In the Congo Basin hunting is crucial, but unsustainable. It is major livelihood activity and source of protein and fungible income, but will be largely depleted by the middle of the century, as wildlife populations collapse under unsustainable levels of extraction. Cooccurring with the predominantly local and informal bushmeat trade is industrial forestry, often conducted at a large scale, operated by Asian and European owned companies, and serving export markets in Asia and Europe. Forestry tends to intensify hunting, and so forestry companies and consumers of forestry products are complicit in the depletion of wildlife, and hence for mitigating against it. Unfortunately, successfully managing hunting in the Congo basin has thus far proven extremely difficult.

Chapter two of this thesis assesses how wildlife is managed in one such concession, Forest Management Unit Ngombé, and using historical analysis attempts to explain how things came to be like they are. Over several decades, processes occurring at the national, regional, and international levels have led to major changes in the way wildlife is managed. These processes led to a number of new actors assuming influential roles in wildlife management, particularly a co-management entity comprised of the state, an international NGO, and a forestry company. At the same time, the role of hunters and their communities has remained weak: they are increasingly managed, but are not themselves managers.

Community Based Natural Resource Management (CBNRM) has been proposed to try to redress this situation. At the level of a rural community, managing wildlife and managing hunting are one and the same thing, and so hunter self-monitoring schemes are often central to community wildlife management initiatives. Hunter-self monitoring, and locally based monitoring schemes in general, may benefit natural resource management in two important ways: by providing estimates of wildlife populations and by encouraging more sustainable resource harvesting. However, there is a scarcity of empirical evidence that hunter self-monitoring can actually be effective for either. The latter chapters of this thesis attempt to address this deficiency.

Chapter three shows, using a lab-in-the-field behavioural economic experiment in the form of a game about hunting using groups of hunters as subjects, that self-monitoring may help resource users to coordinate themselves to harvest resources at a lower rate. In the experiment, self-

monitoring was sufficient to reduce hunting effort compared to when it was absent. Furthermore, subjects who could communicate with each other, but who lacked a more formal self-monitoring system, harvested at the same level as those who couldn't communicate with other subjects at all. The ultimate reasons for this are impossible to determine from the experiment, but subjects did not appear to gain a better understanding of resource state from the self-monitoring. Instead the affect appeared to be social, facilitating coordination more directly, and potential explanations for this are discussed.

Chapter four shows that it is possible to produce estimates useful in wildlife monitoring from hunting records, by converting them into wildlife indicators. Shotgun and snare hunting records were taken from a hunter self-monitoring scheme implemented in eight villages spread across FMU Ngombé, and a camera trapping survey provided an independent comparison dataset. We tested three different indicators, each of which have been used in the study of tropical wildlife. Indicators calculated from shotgun, snare, and camera surveys were often correlated, responded predictably to hunting pressure, and were correlated with the abundances of many individual species. However, the smallest and largest species are underrepresented in hunting surveys because of the selectivity of hunting. This reduces the effectiveness of shotgun hunting as a survey method, and of one commonly used wildlife indicator. This result is useful for both CBNRM schemes and also for the large-scale biomonitoring programs that are ongoing across much of the Congo basin that currently focus on elephants and great apes, and so are of limited value to hunters.

This thesis contributes to the understanding of how wildlife is managed in industrial forestry concessions and why, and the potential of hunter self-monitoring as a tool to improve management and monitoring. The chapters highlight the failure to engage hunters despite decades of progress in wildlife and forestry management, and show that hunter self-monitoring may contribute to better wildlife management by encouraging more sustainable levels of hunting and effective wildlife monitoring.

Chapter 1: Introduction

The hunting of tropical wildlife for food, or "bushmeat", is a major threat to biodiversity (Ripple et al. 2016), but is also a major contributor to livelihoods and food security (Fa et al. 2003). A survey of ~8000 households across 24 countries in Latin America, Asia, and Sub-Saharan Africa found that 39% engage in hunting, primarily for their own consumption, which represents somewhere in the region of 150 million households (Nielsen et al. 2017). However, hunting is often carried out in an unsustainable manner, threatening the viability of wildlife populations and the human livelihoods and communities that depend on them.

The impact of hunting has intensified over the last few decades as traditional hunting technology has been replaced by shotguns, metal snares, and flashlights, and road and settlement expansion has increasingly connected forest interiors to distant urban centres, with supply chains reaching from hunting camps to towns and cities, via wholesalers, traders, markets and restaurants (Cowlishaw et al. 2005). The estimated annual offtake of bushmeat in Africa is five million tons (Fa et al. 2002), and wildlife populations there face widespread collapse by the middle of this century (Fa et al. 2003). Although the offtake is overwhelmingly composed of game species, including rodents, forest antelopes, and monkeys, bushmeat hunting is a major threat to more vulnerable and strongly protected species, including great apes (Strindberg et al. 2018).

Unfortunately, these pressures are set to increase. The human population of Congo basin countries will more than double by 2050 (UN, 2017), and so the demand for bushmeat will likely continue to grow, even as the supply collapses (Fa et al. 2003). In the context of growing local and global demand for resources, central Africa's currently underdeveloped agricultural sector and substantial forest area well-suited to agricultural production, all set the stage for even more extensive resource extraction and land use change in the near future (Doetinchem et al., 2016).

Hunting is generally permitted, with restrictions, in the non-protected forests of the Congo basin, which account for ~90% of the total forest area (Eba'a Atyi et al. 2008). Industrial forestry is also a major economic activity in these forests. Forestry concessions expanded significantly during the 20th Century, into many of the same forests where hunting had occurred for millennia. The expansion is still ongoing, and now extends throughout the western half the Congo rainforest. Since the beginning of the 21st century the global surface area of Intact Forest

Landscape (IFL[1]) fell by 7.2%, with industrial timber extraction being the primary cause (Potapov et al. 2017). Forestry impacts hunting in a number of ways. Negative impacts include the exacerbation of hunting, by bringing money and workers to forests, creating new markets, and by opening up intact forests with logging roads which allow hunters to more easily access remote forest interiors (Robinson et al. 1999).

Thus, forestry is a contributor to a bushmeat crisis. This culpability has brought scrutiny from the international community, and with it an opportunity, in the form of a new lever through which to influence what is a local, informal, and unregulated bushmeat trade. By creating an environment in which responsibly run forestry companies can remain commercially viable, via economic incentives and the sanctioning of less responsible companies, it is hoped that forestry can contribute to better management of wildlife resources. In the Congo basin, this has been manifested in wildlife co-management arrangements involving forestry companies, international conservation NGOs, and the state (Clark and Poulsen 2012). This development suggests that as well as the better understood negative impacts, the relationship between bushmeat and forestry is more complex, still evolving, and still only poorly understood.

The complementary tools of Community-Based Natural Resource Management (CBNRM) and hunter self-monitoring (HSM) schemes have been proposed to address the challenges of wildlife governance and monitoring. Both potentially offer a means to integrate hunters and communities into management processes, from which they are largely excluded today. CBNRM is an attractive idea for reasons that include notions of equity, legitimacy, and the right to self-determination of communities (Shackleton et al. 2002). These concerns are highly relevant to the Congo basin context, where despite some forms of hunting being legal, hunting as it is generally practiced is criminalized. Some have argued that this has undermined the possibility of meaningful community participation in wildlife management (Brown, 2003) and thus shared goal of sustainability.

Ultimately, CBNRM success is contingent on the ability of schemes to ensure the sustainability of harvesting systems, which includes the long-term viability of wildlife populations. Evidence from other natural resource systems suggests that community management can result in more immediate action taken at the local scale, rather than top-down attempts which tend to take many years to implement and have less local-scale influence (Danielsen et al. 2007). Unfortunately,

[1] An intact forest landscape (IFL) is a seamless mosaic of forest and naturally treeless ecosystems with no remotely detected signs of human activity and a minimum area of 500 km^2.

there are no successful examples of CBNRM concerning wildlife in the Congo basin, and so questions remain as to its utility in this challenging context. Recently however, CBNRM has become a focus of attention in the form of a fairly large-scale pilot project attempting to implement sustainable bushmeat management at several sites across the Congo basin (van Vliet et al. 2017). The project, if successful, could serve as a guide for much more widespread implementation.

In hunter-self monitoring (HSM), the outcomes of hunting trips are recorded, usually by hunters to monitors from the same community, and with external support from a research team or NGO. Because harvesting is the major threat to wildlife in tropical forests, HSM is often a core component of CBNRM.

The primary objective of HSM is to provide information about wildlife populations and/or hunting behaviour that can be used to support decision making processes. Yet researchers have thus far been unable to verify estimates of wildlife populations derived from HSM by comparison to estimates from more standardised survey techniques. The difficulty in accomplishing this stem largely from the errors associated with estimating wildlife populations, regardless of the method used (Plumptre 2002; Sollman et al. 2013). Finding a way to convert hunting records into useful estimates would help elevate HSM from an interesting idea used in isolated cases, to an important component of wildlife monitoring, whether at the village or landscape scale, and so has relevance to both conservationists and, critically, to hunters and their communities whose livelihoods and wellbeing depend far more on game species than they do on elephants and great apes. Furthermore, there is growing evidence from non-bushmeat natural resource systems that self-monitoring schemes bring additional benefits, including catalysing communities to take a more active role in natural resource management. Although it has not been systematically studied, anecdotal evidence suggests that this could also be the case for HSM (Noss et al. 1994).

Chapter overview

This thesis addresses challenges to sustainable wildlife harvest in the context of industrial forestry concessions, arising from two sources: the governance challenge, and the technical challenge of wildlife monitoring. Wildlife governance is the formation and stewardship of the institutions (or rules) by which people use and manage wildlife (Hyden and Court, 2002). It concerns all of the numerous stakeholders who participate in the use and management of wildlife, and its study falls largely within the domain of the social sciences. Wildlife monitoring is the measurement of wildlife populations over time, and is a challenge falling within the domain of the natural sciences.

Chapter 2 assesses the current state of wildlife governance at the scale of the concession, using FMU Ngombé as a case study, network analysis to model the relationships between stakeholders, and an historical overview that links the network structure and its evolution to developments occurring at the national and international levels. The number and diversity of stakeholders have increased from the 1990s, as two domestic forestry companies were replaced by a single European owned company, which subsequently entered into a powerful wildlife co-management agreement including the state and an international wildlife NGO, and as local civil society organisations proliferated. Most recently, an intergovernmental project has attempted to implement a CWM project, with the aim of establishing a more active and legitimate role for hunters in wildlife management processes. All of these developments happened in concert with evolving national, multilateral, and bilateral forestry governance regulations and initiatives, as local and international governments and non-governmental organisations have sought to influence wildlife hunting in the remote villages of Congo.

Chapter 3 addresses the potential of HSM schemes to change hunter behaviour. Because of the difficulty in testing this directly, we use an alternative method to address this issue: a behavioural economic experiment in the form of a game framed around bushmeat harvesting. We played the game with 150 people living in small villages in the same forestry concession in Congo, all of whom had some experience of hunting. We tested whether the addition of a voluntary self-monitoring scheme could change the behaviour of players in the game, akin to a village level CBNRM situation. The addition of a self-monitoring scheme reduced the rate at which players hunted. This result supports the notion that HSM might encourage more sustainable hunting, an outcome reported in several other real-world natural resource management systems.

Chapter 4 addresses the challenge of producing useful wildlife monitoring estimates from a hunter self-monitoring scheme. To do this, we implemented a short-term hunter self-monitoring scheme in eight villages in the Republic of Congo. We compared shotgun and snare hunting records from the scheme with records from a concurrent camera trapping survey. We found that by converting raw hunter records to two commonly used wildlife indicators, we could obtain correlations between hunter and camera records. These indicators were in turn correlated with the abundances of individual species, with larger species (>10 kg) declining rapidly under hunting pressure, and smaller species becoming more abundant. The indicators also responded predictably to a proxy of hunting pressure, estimated from human population density. These results suggest that indicators can offer a solution to the challenge of wildlife monitoring with HSM, that is simple to calculate and understand and that carries a substantial amount of information about the state of wildlife populations under hunting pressure.

Chapter 5 synthesises the previous three chapters, and explores their implications for wildlife management.

Chapter 2: The co-evolution of wildlife and forestry management in tropical forestry concessions: A case study from the Republic of Congo

Sergio Marrocoli, Martin Reinhardt Nielsen, David Morgan, and Hjalmar Kühl

Abstract

Forestry and wildlife hunting co-occur over much of the Congo basin, giving rise to a complex, interconnected systems that present considerable governance challenges. Efforts to overcome these challenges have facilitated the emergence of new collaborative approaches involving a multitude of government and non-government actors. This includes increasingly involving stakeholder from the forestry sector, in recognition of the negative impact of forestry on wildlife populations. However, these wildlife governance systems in tropical forestry concessions have arisen only recently, and so remain relatively unknown. Using a forestry concession in the Republic of Congo as a case study, we describe a wildlife governance network and outline the historical processes that led to its formation and current form. We found substantial changes occurring over recent decades. Before the 1980s, villages and bushmeat traders were the only major actors and the only hunting regulation was customary. The first attempts to regulate hunting began in 2007 with the establishment of a wildlife co-management scheme, including the state, an international wildlife NGO, and the European owned forestry company that manages the concession today. At present, 21 stakeholder organisations and groups participate in wildlife governance, and actively attempt to manage through law enforcement, community wildlife management, providing livelihood opportunities as alternatives to hunting, and public awareness campaigns. This development was driven by processes occurring at national, regional and international levels, including legal reforms, market incentives, bi- and multilateral initiatives, and financial support for activities on the ground. Despite these changes, communities remain largely excluded from participation in wildlife management, and stakeholders face a complicated legal environment and difficult economic context which means that wildlife management is still a major challenge.

Introduction

The complexity of social-ecological systems makes their governance[2] a major challenge, and has often rendered natural resource conservation a "wicked problem"[3] (Game et al., 2014). This is especially true in the Congo basin, where factors such as poverty, corruption, political instability, and the state's weak control over national territory have all hampered forest governance (du Preez, 2010), despite decades of regional and international efforts. Forests produce many natural resources, that are harvested by multiple different resource users under an array of different management regimes, that are often overlapping, and thus interconnected. Efforts to meet this governance challenge have facilitated the emergence of new collaborative approaches involving government and non-government actors from the private sector and civil society (Lockwood et al., 2010).

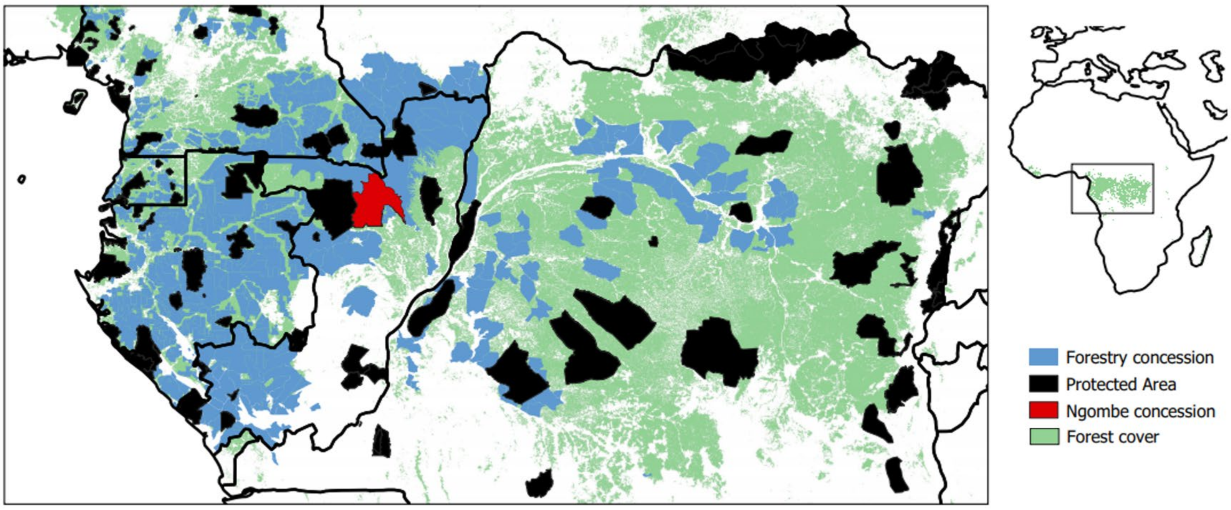

Figure 1. Map of forest cover and land designation in the Congo Basin. The vast majority of forests in the western basin are found within forestry concessions.

In the Congo basin, two of the most important forest products are timber and wild meat, also known as "bushmeat". The first, forestry, is an important part of both the formal and informal

[2] Here we use the definition of governance as 'the interactions among structures, processes and traditions that determine how power and responsibilities are exercised, how decisions are taken, and how citizens or other stakeholders have their say' (Graham et al. 2003, p. ii).

[3] A problem that is difficult or impossible to solve largely due to complexity, interdependency, and have no definitive solution (Rittel and Webber 1973), as opposed to more simple, or "tame", problems, such as those in mathematics that can be solved, even though they can be extremely difficult.

economy. Forestry supplying European markets is generally conducted on an industrial scale by large companies and contributes to governments' budgets through taxation (Bayol et al., 2012). Bushmeat hunting (henceforth "hunting") occurs almost entirely within the informal economy, conducted by many thousands of independent hunters in local communities, and is a major contributor to rural livelihoods, diets, and household income (Nielsen et al., 2018). Hunting is permitted in non-protected land that make up ~90% of the Congo Basin forest estate, much of which is also designated as forestry concessions (Laporte et al., 2007. Figure 1). Therefore, forestry often co-occurs with hunting, with multiple traditional hunting territories of villages encompassed by a single forestry concession.

Although the direct impacts of forestry does alter the abundances of wildlife species (Poulsen et al., 2011), logged forests are still capable of supporting large wildlife populations (Clark et al., 2009; Morgan et al., 2018; Stokes et al., 2010), particularly under reduced impact logging regimes which prevail over the majority of the Congo basin. However, forestry operations invariably exacerbate the hunting of wildlife, by opening up remote forest areas with roads that provide easy access to hunters, by increasing local demand, and by facilitating export of bushmeat from forests to urban areas (Wilkie et al., 2000). The importance of these indirect effects mean that low impact logging alone is insufficient to mitigate the impact of forestry on wildlife. In recognition of this fact, forestry management in areas of the Congo basin has evolved considerably over recent decades (Nasi et al., 2012), becoming increasingly linked to wildlife management. Today, an increasing number of stakeholders participate in forestry and wildlife governance, including international wildlife NGOs, forestry companies, intergovernmental organisations, and civil society organisations (CSOs), alongside the communities, traders, and state agencies that predate them. These often work together towards various objectives, in formal platforms or collaborations.

This complex set of interactions can be represented as a network (figure 2). Social networks are useful for conceptualising the interactions between stakeholders, and social network analysis (SNA) can provide descriptive and analytic insights into the governance system as a whole (Morgans et al., 2017). SNA has increasingly been applied to environmental governance issues, but to the best of our knowledge has not yet been applied to wildlife governance in tropical forests. While SNA provides information about how governance networks are structured, they do not generally provide information about why it came to have a particular structure. A historical approach, analysing documents, oral histories, and observations, can contribute to forming an account of the evolution and development of the context in which a governance network exists (Pavlovich, 2003), and thus help to explain its current structure. Here we apply social network analysis to wildlife stakeholders in a single forestry concession in northern Congo, combined with a historical approach that seeks to link the structure and function of the network as it exists today to processes occurring at the local, national, and international level, over the past several decades.

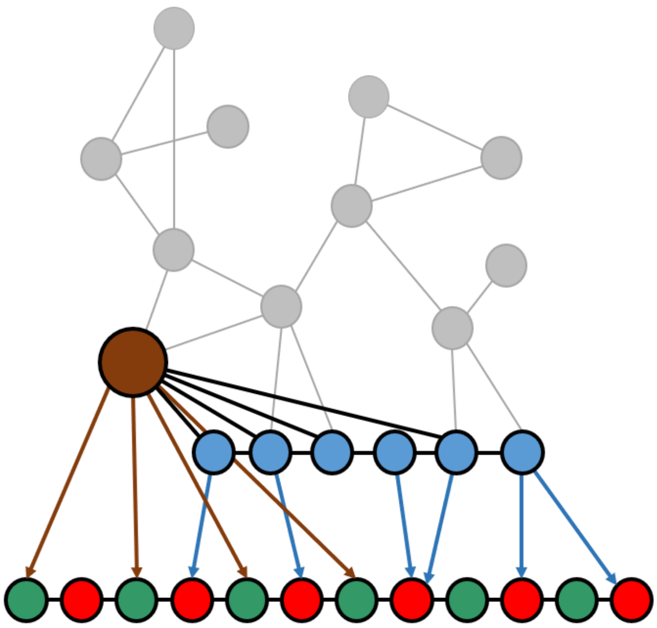

Figure 2. A conceptual diagram of a governance network in a tropical forestry concession, including an ecological system of interacting trees (green) and wildlife (red), villages that engage in hunting (blue), a forestry company (brown), and other governance actors (grey). The lines linking different resources, resource users, and other actors, indicate an interaction between the two. The forestry company and villages both use resources directly, as well as interacting with each other and with other actors. The other governance actors do not interact with resources directly, but exert influence via the resource users and via each other.

Materials and Methods

Study Location

This case study is centred on the area that is today designated as Forest Management Unit (FMU) Ngombé. FMU Ngombé is located in the Sangha department of northern Congo (figure 1) and covers more than 10,000 km^2 of dense moist forests. According to national law, wildlife, timber, and other forest products can be harvested, within specific restrictions. Despite industrial forestry and hunting, FMU Ngombé is home to some of the highest concentrations of gorillas, chimpanzees, and elephants on earth (Strindberg et al., 2018). It is part of the TRIDOM (Tri-National Dja—Odzala—Minkebe) landscape, a 178,000 km^2 rainforest mosaic of protected areas and forestry concessions and adjoins Odzala-Kokoua National Park (PNOK) in the south-west of the concession, and the Ntokou-Pikounda National Park in the south-east. The concession's northern and eastern boundaries are marked by two large rivers, the Sangha and Ngoko. Two major roads link the concession to the capital to the south and to the town of Sembe towards the west. Approximately 6,000 people live in villages within the concession area, almost entirely in permanent villages along the roads and rivers. Rural livelihoods usually rely on some mix of hunting, fishing, gathering, agriculture, and casual wage labour. Short hunting trips are made from villages, and multi-day trips from semi-permanent camps deeper in the forest. An additional 10,000 people live in the town of Ngombé, which is the site of the forestry company, and 50,000 in Ouesso and its immediate surroundings, many of whom also depend on forest products for subsistence and commerce. Until recently, Ouesso was a frontier town where bushmeat hunting and trading was unregulated, leading to the depletion of large mammals in its near vicinity (Hennessey and Rogers, 2008). Today, efforts to limit the trade are focused on checkpoints on the main road and forest patrols, carried out by ecoguards, who are the enforcement units of a wildlife co-management arrangement including the state, the Wildlife Conservation Society (WCS), and the forestry company *Industrie Forestière d'Ouesso* (IFO).

Data Collection

Data collection consisted of interviews with villagers and higher-level stakeholders to establish the membership and structure of the network and exploration of published articles, reports, and secondary documents, as well as satellite imagery, to form an account of historical events. We conducted two rounds of interviews with wildlife governance stakeholders active in FMU Ngombé in 2016. Beginning with those we were already aware of, a first round of exploratory interviews was conducted in January and February 2016 with the goal of establishing who the relevant actors were, what they did, and what they saw as concerns in wildlife governance. We also used their insights as a basis for snowballing to identify additional stakeholders within the concession concerned with wildlife, who we then also interviewed. We eliminated stakeholders that did not work directly with wildlife or bushmeat issues in some way. Using this approach, we compiled a list of relevant stakeholders that we used to define the network boundary. Stakeholders included organisations (e.g. NGOs, companies, and government departments) and "groups" (villages and traders), but not collaborative arrangements such as platforms or co-management schemes. The network included 19 stakeholder organisations, and an additional two stakeholder groups. We conducted a second round of interviews July 2016, with the intention to describe the governance network quantitatively. Stakeholders (organisations and groups) are the nodes in our network, and we collected information to determine the links between each of them. Using the list compiled in the first round of interviews, we asked each organisation about their relationships to all other stakeholders, including how often they had contact with them (multiple choice between the following categories: once per year or less, less than once per month, once per month, once per week, most days), whether or not they were involved in official collaborations or platforms, and how influential they perceived other organisations to be in wildlife management (zero to five scale). Interviews took between 30 minutes and 2 hours. We conducted the interviews with either one or two people from each organisation, depending on the size of the organisation and if the organisation's staff were spread over one or two locations.

We treated villages and traders, two rather disparate groups, differently from organisations, because they consist of dozens to thousands of largely independent actors. We did this by treating villages and traders as a single actor in our questionnaire, asking stakeholder organisations questions on their relationships with the groups as a whole, i.e. we asked how frequently the organisation had contact with villages in general, not with a specific village, and we did the same for traders. Where villages had some form of organisation concerned with the regulation of hunting, we treated it as an organisation, separate from the village group. To understand the role of villages in the network, we conducted interviews in twelve villages. In these villages, we asked groups of hunters and others, such as village Chiefs or secretaries where present, to recall the number of times different organisations and groups had visited the village in the past 12 months. Hence, we treat villages and traders each as single nodes in our network analysis, and present the village perspective separately. We also interviewed older members of the villages to reconstruct the history of wildlife use and management in the concession.

Network Analysis

The network was visualised and analysed using Gephi (version 0.9.2. Bastian et al., 2009), with the wildlife stakeholders (organisations and groups) forming the nodes, and contact between them determining the links (edges). Edges were weighted by the frequency of contact, so that stakeholders in more frequent contact were more strongly linked, and depicted as closer together. We converted the categories in the questionnaire into a numeric value between zero and one, by dividing the number of days of contact per year implied by the category by the estimated number of working days in a year (260): once per year or less = 0; less than once per month = 0.023, once per month = 0.046, once per week = 0.2, most days = 1. Two stakeholders often gave different estimates of their frequency of contact with each other, so we used the mean of the two estimates as the weight. When we were unable to conduct an interview with one of two stakeholder organisations in contact (n = 3), and when respondents were unable to estimate the frequency of contact (n = 2), we assumed the frequency of contact was equal to that given by the other party.

Using the weighted contact data, we calculated two metrics for each node in the network: weighted degree centrality and betweenness centrality (Bodin and Crona, 2009). Weighted degree centrality is the sum of all weights a stakeholder has with all other stakeholders in the network. Stakeholders with higher degree centrality are likely to be "well connected”, and so may have more access to many alternative sources of information, resources and so forth, and the number of ties may have a positive effect on an actor’s influence (Degenne and Forsé, 1999). Betweenness centrality is the frequency with which a stakeholder falls on the path between pairs of other members. Stakeholders with high betweenness centrality may act as intermediaries or bridging organisations, linking more isolated stakeholders to the larger network, and may exert control by being able to control the flow of information and resources (Burt, 2004). To assess which organisations formed sub-networks of highly interconnected nodes we used Gephi’s modularity function (Blondel et al., 2008), choosing a resolution of 0.31. This split the network into five communities that corresponded with our real-world experience of the network. We compared the metrics and subnetwork membership to perceived influence on wildlife management decisions as rated by stakeholder organisations. We did not include collaborative platforms in the network except where the platform had a dedicated coordination entity, but instead considered how collaborations were reflected in the network structure.

Results

Overview

Starting around 1980, the number of wildlife stakeholders began to increase, before accelerating around 1990 and increasing steadily until today (figure 3 and table 1). At the beginning of the study period communities and traders were the only influential actors, and the only regulation of hunting occurred at the village level, via traditional or customary institutions. Beginning in the 1980s forestry, initially carried out by state owned companies, began to exacerbate hunting and trade of wildlife, by purchasing bushmeat and ferrying hunters into the forest on forestry vehicles (Wilkie et al., 2001). The network began to grow with the establishment of a number of Civil Society Organisations (CSOs) which include wildlife and bushmeat within their remit, which were banned in Congo prior to 1991 (Mavah, 2011). Later, international NGOs and others more focused on and with greater capacity to manage wildlife entered the arena, while a European owned company took over forestry operations. Today we identified 21 organisations and groups (table 2) with some involvement in wildlife issues. This includes dedicated wildlife management organisations that attempt to control hunting and the wildlife trade through law enforcement, and recently by engaging villages in wildlife focused Community Based Natural Resource Management (CBNRM). This transition was precipitated by a diverse array of processes occurring at different scales, including improved state capacity and regional cooperation, forestry legal reform, an international drive towards decentralisation, and innovative wildlife management arrangements pioneered in other concessions, many of which were achieved with international support or pressure, in the form of bilateral and multilateral initiatives, sustainable certification schemes, and funding of conservation organisations and initiatives on the ground.

Case Data

Pre-1990

At the beginning of the 20th century, the study area was contested by shifting French and German colonial powers, and ruled relatively autonomously, and often brutally, by European concessionaires, who extracted rubber and ivory rather than timber (Coquery-Vidrovitch, 1998). The Odzala-Kokoua National Park was established by the French colonial government in 1935 (Gami, 2016). When the value of rubber declined at the end of the second world war, so did the interest of the concessionaires. By the 1960s timber exploitation became possible in the remote north of Congo with the arrival of heavy machinery, but remained too risky to be common. Forestry was Congo's most important industry at this point, but was harvested almost entirely in the more accessible south (Codou and Gibert, 1975). At the international level, the Declaration of the 1972 Stockholm Conference included in its principles the safeguarding of natural resources, wildlife, and support of the development of environmental safeguards in developing countries, signalling the rise of global environmental consciousness (Nasi et al., 2006). In Congo, the state strengthened its role in forestry via investment in state-controlled forestry enterprises (WRI, 2005) and through the implementation of a new forestry code in 1974, which included defining FMUs, maximum allowable annual harvests and promotion of local processing (WRI, 2007).

In northern Congo, forestry operations began in earnest in the 1980s, often supported by loans from multilateral development banks (Wilkie et al., 1992). By the end of the 1980s forestry roads were visible in satellite images in the north, centre, and west of the study area, as well as in the neighbouring *Congolaise Industrielle des Bois* (CIB) concessions north of the Sangha river. The state owned company *La Société Congolaise des Bois de Ouesso* (SCBO), began forestry operations during this period, and was based near Ouesso on the road to Liouesso between 1983 and 1987, before building infrastructure, including a sawmill and port, in the town of Ngombé in 1987 (Mengho, 1994). A second, semi-public, company, *La Société Forestière Algero-Congolaise* (SFAC), was allocated a concession in 1983, in the area covering the western third of what is now FMU Ngombé (Wilkie et al., 1992). Researchers working in the area report that forestry operations intensified hunting, with SFAC employees actively facilitating hunting by ferrying hunters and meat to and from remote forest, and by lending guns to hunters in exchange for meat (Wilkie et al., 2001).

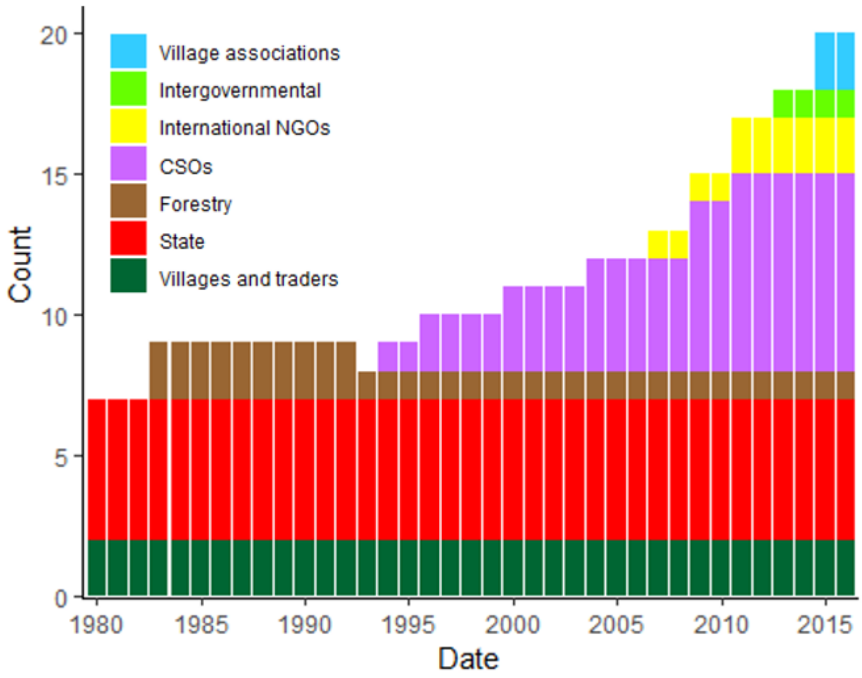

Figure 3. The number and type of actors with some involvement in wildlife governance in FMU Ngombé by year

The population of Ouesso more than doubled in size from 4,500 to 10,980 individuals between 1960 and 1980, as its modern facilities attracted migration from the immediate countryside, contributing to the depopulation of its hinterlands (Mengho, 1984). Semi-nomadic hunter-gatherer populations became sedentary and began to live in settlements along roads and rivers alongside bantu farmers. By this period modern hunting equipment, including guns, snares, and torches that enabled hunting at night were already widely available. As today, bushmeat was traded primarily from small villages along roads running south and west through the concession, and along the Sangha and Ngoko rivers, including into Cameroon. Traders carried hunting equipment and other goods, such as batteries and medicine, and returned with bushmeat and fish, using trucks or pirogues travelling several times per week, or on foot. Access to remote forest areas was limited by the road network. Satellite images show that the road south from Ouesso sometimes ended at Liouesso, disappearing as it became overgrown by vegetation, and sometimes continued beyond, linking to the north to the south. During the 1980s it was mostly closed. Where roads ended traders would continue on foot to more remote villages. During this period, and continuing into the 1990s, chimpanzee, gorilla, and elephant meat and tusks were traded freely in Ouesso's markets, and traders and hunters conducted business with no fear of persecution (Hennessey and Rogers, 2008).

Hunting was largely managed under Customary land rights, which were recognised under the tenure law of 1958, giving communities ownership over specific areas (Mavah, 2011). Customary practices included exclusion of outsiders or allowing access only in return for a share of the catch, as well as closure of over-hunted areas. These customary rules were often enforced through appeal to the supernatural, a situation still remembered by the oldest members of surveyed communities. Customary land rights were eventually abolished under law no 52/83 of 1983. Wildlife became the property of the state under law n° 48/83, which established different levels of protection for different species, prohibited hunting at night, banned the use of metal snares, required hunters to hold a valid license, and included provisions for quotas (although these are never used in practice). Traditional hunting practices were unrestricted, and the new law did not specify the sale of bushmeat as illegal. Most importantly the abolishment of traditional hunting management meant that the ability to exclude outsiders was revoked. However, the state did not possess the necessary capacity to enforce the new wildlife law (Mavah, 2011). This left wildlife a de facto open access resource, with hunting intensity probably only passively constrained by low population densities and economic factors. In 1983 Congo became a party to the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), an international agreement between governments regulating the international trade in wildlife products.

1990s

The 1990s saw moves by Congo basin countries towards a more integrative regional approach to forestry management, and greater attention from the international community. In 1992, ECOFAC (Conservation and Rational Use of Forest Ecosystems in Central Africa) of the European Commission (EC) was initiated (EC, 2006), with the aim of establishing a coordinated framework to promote regional conservation in central Africa. Early phases were concerned with assessment and the establishment and improvement of Protected Areas (PAs), and ECOFAC began to support the Odzala-Kokoua PA almost immediately, which had been under financed since its gazettement in 1935 (African Parks Network, 2010). The United

Table 1. The wildlife stakeholder organisations and groups present in FMU Ngombé in 2016

	Name	Active since	Role
	Village Level		
1	Villages	Before 1980	Resource extraction, consuming, and trading
2	Traders	Before 1980	Trade of bushmeat from villages to urban centres
3	Association for the Management of the Community Area of Liouesso (AGACL)	2015	Represent hunters in negotiations with other stakeholders, and participate in development of bushmeat strategy
4	Village Association of Lango	2015	Represents village, primarily in regards to Africa Parks and Odzala-Kokoua NP
	State		
5	MEFDD (Departmental Director of Forest Economy)	Before 1980	Government ministry responsible for environment
6	PROGEP-PNOK (Project for Ecosystem Management in the Odzala-Kokoua Periphery)	2007	The state component of PROGEP, which acts as coordinator and includes the ecoguard enforcement units
7	Judicary	Before 1980	Prosecute legal cases related to illegal hunting
8	Department Council	Before 1980	Role includes regulating investment plans and programs
9	Sous-prefecture	Before 1980	Department level government
10	Gendarmerie	Before 1980	Law enforcement
	Forestry		
11	IFO (Industrial Foresty Ouesso)	2000	Forestry company. Contributes towards PROGEP and Local Development Fund via financing and partipating in management
	International NGOs		
12	WCS (Wildlife Conservation Society)	2012	International wildlife conservation NGO, participating in PROGEP with technical support, wildlife monitoring, and education among its activities
13	Africa Parks	2011	Management Odzala-Kokoua National Park
	Intergovernmental		
14	FAO (Food and Agriculture Organisation)	2013	Implements the sustainable bushmeat project at Liouesso, including ecological and social assessments, institution building, and advising and monitoring of program
	Civil Society Organisations (CSO)		
15	AM (Friends of the Earth)	2004	Engages with bushmeat traders, especially education concerning legality of hunting and trading
16	CIREK (International circle of research of the Bakwele civilisation)	2000	Advocates for indigenous people, with focus on culture
17	SAM (Sangha Medical Assistance)	1994	Health, including bushmeat and disease education e.g. ebola
18	APETDS (Association for the promotion of tropical ecosystems and the protection of the environment)	1996	General biodiversity and conservation
19	UDEMK (Universe of Ecosystem Defenders of Miélé-Kouka)	2011	Biodiversity and conservation at single site
20	APVPS (Professional association for the valuation of subsidiary forest products)	2009	Researches and implements alternative livelihoods like cocao and porcupine husbandry
21	OCBE Vert (Observatory of Bantu cultures and biodivervsity and environmental education)	2009	Advocates for Bantu peoples, culture and environment

States Agency for International Development (USAID) supported the establishment of the Central Africa Regional Program for the Environment (CARPE), which launched in 1995, with the aim of assisting forest governance, research and monitoring and conservation in central Africa. The program began with an assessment to understand the legal, social, biological and administrative contexts in the region (Trefon, 2017). Both ECOFAC and CARPE remain major funders in the region today.

In 1999 the Yaoundé summit brought together heads of state and 600 delegates from the Congo basin region, and a treaty signed by the presidents of the ten member-states laid the legal basis for the establishment of the intergovernmental Central African Forests Commission (COMIFAC). COMIFAC remains the central policy and decision-making body for cross-border conservation and management initiatives for Central African forests. A wave of forestry sector reforms that eventually influenced forest governance in all Congo basin countries, was pushed forward mainly by the World Bank and the International Monetary Fund (IMF), using "conditionalities" attached to critical loans and grants (Karsenty, 2007). These began in Cameroon in 1994, which served as a "laboratory" for technical and economic reforms, due to early intervention of the World Bank and other donors in the 1990's, and the wide-ranging reforms initiated included environmental regulation, taxation reform, and social transfer requirements. The World Bank did not hold the same string financial levers over Congo but the effects of interventions for introducing reforms were still tangible (Karsenty, 2017).

While change began to accelerate at the policy level, they were often slow to materialise on the ground. Lack of progress led frustrated international environmental organisations to consider an alternative course of action. The Forest Stewardship Council (FSC) was founded in 1993 by several high profile international environmental NGOs, including Greenpeace, World Wild Fund for Nature (WWF), and the Rainforest Alliance, along with international wood and wood-product companies, including B&Q and later IKEA (Moog et al., 2012). The FSC was conceived as a financial mechanism by which forestry companies could be incentivised to comply with, and exceed, the standards set in forestry codes. Although founded in 1993, the FSC did not certify a concession in the Congo basin until a decade later.

In the study area, SFAC abandoned their concession in the west due to bankruptcy in 1992 (Wilkie et al., 2000), while satellite photos show SCBO expanded their forestry roads into the centre of the concession. The dirt road linking Ouesso to the south of the country and the capital was present in 1990, but appears to have been abandoned again by the end of the decade. By this period the 20 km^2 area immediately surrounding Ouesso was already devoid of large mammals (Hennessey and Rogers, 2008). A new forestry company, *Industrie Forestière de Ouesso* (IFO), was created in 1999 by the Danzer group, a European based company specialising in high quality veneers. IFO signed a forestry exploitation contract with the government in 1999, and acquired SCBO operations in Ngombé in 2000, in line with the state's policy to withdraw from parastatal companies to the benefit of the private sector. Prior to 1991 it was illegal to create an NGO in Congo (Mavah, 2016), but in 1994 and 1996 the first local NGOs, or Civil Society Organisations (CSOs), began operating. CSOs are often critical actors in the public policy sphere, and in the Congo context can act as independent watchdogs, human rights advocates, or service providers assisting communities organise at the grassroots level (Satyal, 2017). The earliest we recorded were *Sangha Assistance Medical* (SAM), which undertakes health education, including in relation to risks associated with bushmeat, and the Association for the Promotion of Tropical Ecosystems and the Protection of the Environment (APETDS), which has a specific environmental advocacy role, which includes wildlife.

2000s

The 2000s saw continuing regional integration and international involvement in Congo basin forest management, as well as the first concerted attempt to regulate wildlife exploitation in FMU Ngombé. The Congo Basin Forest Partnership (CBFP), an association of over 70 states, institutions, organisations, and private sector partners, was launched in 2002 at the Johannesburg Conference (USAID, 2017). The CBFP is complementary to COMIFAC, which is composed of forestry ministries, and has been facilitated in turns by the US, France, Germany, Canada, and now the EU. In 2006 the CBFP presented 12 priority landscapes across the Congo basin, an approach which includes land-use planning, and incorporates extractive resource zones, along with PAs and Community Based Natural Resource Management (CBNRM) areas. CBFP landscapes are the primary means by which CARPE directs its support, via implementing partners including WCS, WWF, and others. Ngombé falls within the Dja-Odzala-Minkébé (TRIDOM) landscape, and the TRIDOM project is funded by USAID and WWF amongst others (African Parks Network, 2010).

As local and regional initiatives began to solidify, western governments sought to increase influence via bilateral and multilateral initiatives. In 2001, the World Bank-supported Forest Law Enforcement and Governance program emerged as a first response by the international community to address illegal forestry activity. In 2003, this led to the founding of the Africa Forest Law Enforcement and Governance (AFLEG), one of several regional Forest Law Enforcement and Governance (FLEG) entities. AFLEG objectives are broad, but include law enforcement and monitoring of wildlife resources (Gasana and Samyn, 2008). Also in 2003, FLEGT (Forest Law Enforcement, Governance and Trade) funded by the EU began with the purpose of supporting and building on the World Bank coordinated AFLEG and other regional initiatives. Under FLEGT, timber producing countries can enter into voluntary partnership arrangements (VPA) with the EU, in which partner countries and the EU set out commitments and actions to tackle illegal logging, with intended outcomes including improved forest governance, improved access to EU markets, increased revenues collected by partner country governments, and implementation of more effective enforcement tools in partner countries (FLEGT 2007). The Congo VPA process began in 2008 and was signed in 2010. Congo also participates in Reducing Emissions from Deforestation and forest Degradation (REDD+) mechanisms, and is involved with various preparation activities, with a pilot project at the Pikounda North concession adjacent to FMU Ngombé announced in 2012 by CIB, the concessionaire (the REDD desk, 2018). The REDD+ initiative's focus includes institutional arrangements, stakeholder engagement and participation, land tenure, and forest management, amongst others. Congo began its REDD+ readiness activities in 2008. These initiatives would result in the creation of local platforms harnessing CSOs in Ngombé in the following decade.

Forestry code reform reached Congo in 2000, placing multiple demands on forestry companies, including economic, ecological, and social. As in earlier forestry regulations, forestry companies were again required to produce Forestry Management Plans (FMPs). FMPs now became a major investment including extensive ecological (including wildlife) and social surveys, and delineation of wildlife management plans for wildlife, the delimitation of a hunting zone around populated areas, and High Value Conservation Areas. IFO's management planning process for Ngombé, began in 2001, was approved in 2007 by the Forest Ministry, local population and stakeholders, and approved by Decree in 2009. The management plan defines in detail the use rights of local populations to forest resources, including wildlife. In the management plan IFO committed to a voluntary tax levied on each m^3 of wood processed, which finances a Local Development Fund (LDF) with the purposes of supporting community micro-projects such as alternative livelihoods, with the aim of alleviating poverty and reducing hunting. The LDF is an example of a benefit sharing or company-community agreement, which may be more likely to benefit local communities than taxes, as tax revenue is typically not redistributed and so only weakly affect communities living in forestry concessions (Rickenbach and García 2015). Between 2007 and 2012 the contribution reached ~55,000 USD per year.

The LDF is managed by a voluntary committee consisting of representatives of the forestry administration, the subprefecture, local communities, local and international NGOs, and IFO themselves, meeting several times a year. The LDF financed 14 projects in 2009 and 2010, concerned with crop farming, fisheries, and livestock rearing (Rickenbach and García 2015). The effectiveness of the LDF has been called into question, with projects failing to show significant harvest or revenue creation, a high percentage of people abandoning the projects, and dissatisfaction amongst villagers about the impact of projects and level of investment. Lack of LDF committee management capacity has required that IFO take a central role in the implementation and management of initiatives they were initially expecting only to finance. However, the LDF does serve as an important point of contact for a diverse range of groups, who otherwise may not have cause to meet, and the second day of each meeting, held three times a year, is concerned with wildlife issues.

IFO was audited by the FSC in 2009 and was certified the same year, following the award of the first FSC certificate in Congo to CIB in 2006, with a number of consequences for wildlife management. The FSC certification includes procedures and guidelines for the protection of rare, endangered or threatened species and their habitats, compliance with national and/or international regulations on protection, hunting and trade in animal species or parts, supporting the community management of wildlife in collaboration with the competent authorities, to monitor wildlife, and to protect high conservation value forests (FSC, 2015). IFO are also required to ensure that low-cost alternatives to bushmeat are available in Ngombé town, and to prevent bushmeat being transported on the road leaving it, and hunters and bushmeat from moving with company vehicles.

Wildlife regulations continued to change over this period and a new wildlife law introduced in 2008 superseded the old law of 1983. The new law included the stipulation that hunters must be part of a village association able to negotiate, along with other village associations, at the departmental or national level, and that village permits must be issued only through this association. However, these rules have never been implemented. It also includes the legal basis for anti-poaching units, including conditions for arrests and confiscations.

In 2007 the first dedicated wildlife management entity, PROGEP-PNOK (Project for the management of boundary ecosystems of Odzala-Kokoua National Park, henceforth "PROGEP"), was formed, building on a model developed almost a decade earlier in the concessions immediately to the north of FMU Ngombé, by *Congolaise Industrielle des Bois* (CIB) and WCS (Clark and Poulsen, 2012). The three members of PROGEP are IFO, WCS, and the Ministry of Forests and Sustainable Development (*Ministère de l'Economie Forestière et du Développement Durable* - MEFDD), defining the PROGEP coordination and committee, with roles divided between financing and logistic support, monitoring and technical expertise, and law enforcement. The establishment of PROGEP also included the creation of a dedicated Surveillance and anti-poaching unit (henceforth "ecoguards"), equipped with military weapons and operating under the authority of *L'Agence Congolaise de la Faune et des Aires Protégées* (ACFAP), under the MEFDD. Major activities of the ecoguards include locating and confiscating metal snares, illegal firearms, ammunition, and carcasses of protected species, and arresting rule breakers where deemed appropriate. This is done through patrolling in the forest, and at checkpoints on the main roads. In 2007 WCS conducted a first landscape-wide survey of wildlife in the area, finding a very large gorilla population, and significant numbers of elephants and chimpanzees (Maisels et al. 2015). While providing important information on critical conservation species, transect surveys do not provide good information on the state of game species. However, hunting signs were most frequently observed in the north of the concession where the vast majority of people live.

Four more CSOs arrived or were established during this period, *Ami du Monde* (AM, not associated with the international NGO), International Circle of research of the Bakwele civilisation (CIREK), Observatory of Bantu cultures and Biodiversity and Environmental education (OCBE-Vert), and Universe of Ecosystem Defenders of Miélé-Kouka (UDEMK), working on the education of bushmeat traders, advocacy for different ethnic groups, and alternative livelihood projects.

2010 - 2016

In this period national level APV-FLEGT and REDD+ processes began to be operationalised locally, with the establishment of two CSO platforms: The Platform for the Sustainable Management of Forests (PGDF) of FLEGT and The Consultation Framework for Congolese Civil Society and Indigenous Peoples (CACO-REDD) of REDD+ in 2012. Members of the platforms meet several times a year. Despite the different concerns of each of these programs, i.e. illegal timber and climate change, they both aim to contribute to improved forest governance by facilitating legal and institutional reforms and by strengthening inclusive and transparent multi-stakeholder participation (Broekhoven and Wit, 2014). Wildlife is not a significant part of the agenda of either platform, but they still provide a major point of contact for stakeholders who do have some role in wildlife governance.

In 2013 the Food and Agriculture Organisation of the United Nations (FAO), in association with the Centre for International Forestry Research (CIFOR) and Agricultural Research Centre for International Development (CIRAD), began implementing a sustainable bushmeat project (*Sustainable management of the wildlife and bushmeat sector in Central Africa*), and the village association related to the project (*l'Association pour la Gestion de l'Aire Communautaire de Liouesso*, AGACL). The project objectives include, 1. Establishing baseline conditions in the project village, 2. Development of a participatory management plan, including the development of a local governance structure to design and implement it, and 3. Implement the plan, including tools for monitoring and conflict resolution (van Vliet et al. 2017b). The project proposes that legal frameworks must be changed to allow for active participation of local communities, that game meat must become part of the formal sector and part of the government's poverty reduction and food security policy, and that providing local communities with the responsibility for wildlife management must be accompanied by a strong political will, decentralization of wildlife resources management and the strengthening of civil society (van Vliet et al. 2017a). Finally, a second village association was formed in a small village in the edge of the National Park, which is not related to the FAO project, but rather village-National Park relations.

Wildlife hunting came under increasing regulatory pressure, but economic incentives for hunting increased simultaneously. By 2016 there were 23 active ecoguards, with IFO contributing a combined 250,000 USD per year to the LDF and PROGEP (Desmedt 2016). PROGEP was now controlling seven checkpoints on main roads and logging roads, and had conducted thousands of patrols. This included on foot into the forest and by motor vehicles and boats, but also targeted raids in response to information provided by informants (WCS, 2014), and had through these activities confiscated many firearms, carcasses, live animals, and tens of thousands of snares. Pressure on the use of snares appears to have resulted in more use of shotguns, which at this site tend to catch relatively fewer large mammals and more hunting resilient species, such as small duikers and porcupines (Marrocoli et al., In review), but also primates. The checkpoint on the main road prevents protected species from reaching Ouesso, but is in practice contentious because of the constant tension between the ecoguards and the traders who pass through there, and because of the risk of corruption. Additional PROGEP activities included monitoring of two bushmeat markets and an education program in schools in Ouesso, training journalists about conservation issues, and radio broadcasts focusing on the legality of hunting different species, and the importance of some species such as chimpanzees.

Between 2009 and 2014, WCS observed several trends in the two bushmeat markets they monitor, in the forestry town of Ngombé and a medium sized village in the south of the concession (Mokouangonda, ~300 people). In both markets, the price of bushmeat increased by 70% to 100% (WCS, 2014). At the same time, the quantity of bushmeat passing through Ngombé market changed from year to year, but was at the same level in 2014 as it was in 2009. At Mokouangonda the quantity appeared to decline by around ~35% over the same period. WCS propose several possible explanations for the decline in bushmeat supply seen in Mokouangonda, despite the rapidly increasing price, including the increased suppression and confiscation of meat leaving the concession at an ecoguard checkpoint at the southern end of the concession, particularly with respect to the large buses travelling to the capital, the hiring of youth who might otherwise be engaged in hunting by Chinese construction crews, and the impoverishment of wildlife around the villages. WCS also propose that supply at Ngombé did not fall because local demand is higher, reflected in higher price of bushmeat (~20% higher).

A survey of households throughout the concession conducted in 2013 and 2014 found one quarter of households earned cash from hunting, that bushmeat was the most important component of meals in nearly all study villages, and that wildlife was perceived to be in decline around all villages (Mavah et al., 2018). A second biomonitoring survey conducted by WCS (Maisels et al. 2015) found no significant change in great ape, elephant, or ungulate densities between 2007 and 2014 (although dung surveys for ungulates are of questionable accuracy, Bowkett et al. 2006), but did find an increase in hunting signs in the south of the concession along the main road and in the Ntokou-Pikounda National Park at the south east of the concession, created in 2012 but where no eco-guards were active. A separate survey found that the relative abundances of larger species (>10kg) declined with human population density, while smaller, more hunting resistant species abundances showed no change, or even rose slightly (Marrocoli et al., In review), confirming that areas around large population centres in the north of the concession are depleted of large mammals, as reported in the 1990s (Hennessey and Rogers, 2008).

In 2010, a second international conservation NGO, African Parks, took over management of Odzala-Kokuoa National Park, in another example of a Public-Private Partnership, in which the state is responsible for legislation and policy and African Parks is responsible for the execution of management. Park operations are primarily funded by EU.

Present day

The Governance Challenge

To aid understanding of how the network it is structured, we asked stakeholders what they thought the major challenges to wildlife management are (table 2). The three most frequently cited challenges for bushmeat management were poverty and lack of alternative income, inappropriate wildlife law, and commercialisation of bushmeat hunting for trade. The lack of viable alternatives was considered important for hunters, traders, and consumers, and was seen as a major constraint to changing behaviour in relation to hunting and trading bushmeat. The law was considered inappropriate, with reasons given including that it is impossible to apply because of the impact it would have on villages, and so poorly suited to local conditions. This was seen in part as a consequence of lawmakers not understanding reality on the ground. The economic incentives for hunting included its status as a good source of income and also drivers such as the new road providing access to markets in the capital where demand is high.

Network factors were cited as a challenge to management an intermediate number of times, and included a lack of cohesion between actors, lack of information retention, and lack of connectivity between the state and communities, as well as lack of trust between resource users and legislators. Lack of funding preventing organisations, particularly the CSOs, from having a significant role in wildlife management, and the technical challenges of managing wildlife, including the dispersed nature of hunting, slow reproductive rates of wildlife (relative to fish), and the difficulty of apprehending perpetrators, were also cited an intermediate number of times. Some cited a cultural preference for bushmeat, which includes the high meat content of diets, as well as lack of interest in alternatives. Lack of collective action in villages, which also included the abolition of traditional or customary institutions for wildlife management and community discord on hunting, was infrequently mentioned, as was corruption, which included the involvement of powerful interests in elephant poaching for ivory.

Network Structure

The wildlife governance network, weighted by frequency of contact, is shown in figure 3a, and the result of the modularity analysis which cleaved the network into a number of different communities or subnetworks based on their frequency of contact is shown in figure 3b. The network and clustering analysis show five subnetworks:

1. Resource users and CBNRM associations – This group includes communities, traders, and the two CBNRM associations.
2. PROGEP – The three members of PROGEP, which include the PROGEP coordination, WCS, and IFO, which share roles and responsibilities in wildlife management and enforcement, as well as offices at the IFO sawmill.
3. The State – Including the courts, law enforcement, and office of the Subprefecture, for whom bushmeat is not a primary concern, but rather only one of many.
4. CSO subnetwork – the numerous CSO groups, all of which participate in the REDD+ and FLEGT platforms, and some of which participate in the LDF platform. The group also includes the Department Council, who also participate in these platforms.
5. Other – The forestry ministry (MEFDD), FAO, and African Parks.

The inclusion of MEFDD, African Parks, and the FAO in a final community is likely an artefact of the clustering algorithm and survey method used, rather than reflecting a group that exists in the real world. both had links to a range of different stakeholders from different subnetworks because of their roles, but were not like other member of those subnetworks.

Table 2. Governance challenges cited by wildlife organisations in FMU Ngombé

Factor	Count	Examples
Poverty and lack of alternative livelihoods	15	Few economic alternatives for hunters or traders, few nutritional alternatives to bushmeat
Inappropriate wildlife law	13	Law does not fit reality on the ground, impossible to apply because of the impact it would have on villages, lawmakers do not understand the reality on the ground
Economic incentives for hunting	11	high value of bushmeat, strong demand for it in the capital, new transport links enabling supply to reach the capital
Lack of funding for wildlife management	7	prevents organisations, particularly the CSOs, from having a substantial role in wildlife
Network factors	7	lack of cohesion between actors, lack of information retention, and lack of connectivity between the state and communities, and lack of trust between resource users and legislators
Technical challenge of bushmeat management	6	dispersed nature of hunting, slow reproductive rates of wildlife (relative to fish), and difficulty in catching perpetrators
Cultural preference for bushmeat	6	high meat content of diets, as well as lack of interest in alternatives to hunting
Corruption	5	involvement of powerful interests in elephant poaching for ivory
Law not enforced	4	Laws would protect wildlife, but not enforced
lack of collective action in villages	4	Lack of collective action in villages, loss of customary means of wildlife management, community discord on hunting

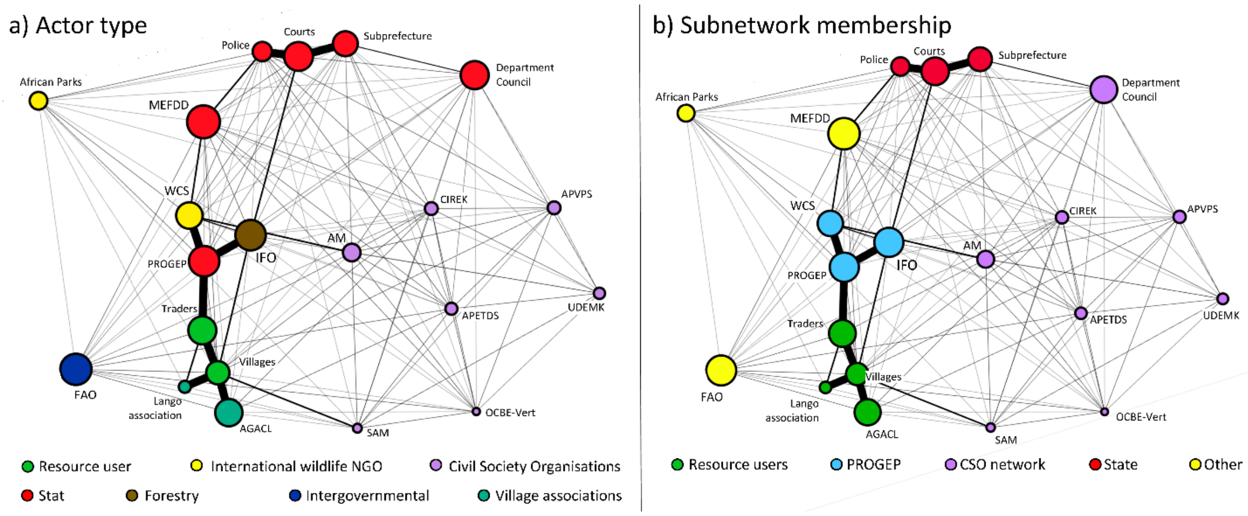

Figure 3. The network of governance actors in Ngombé, showing the weighted network of interactions, with the thickness of links between stakeholders representing frequency of contact. Point size is scaled by perceived influence of the actor. Nodes are coloured by a) actor type, and b) subnetwork membership, as determined using modularity analysis

Table 3. Governance collaborations and platforms present at FMU Ngombé

Name	Description and membership	State	International NGOs	CSOs	Village groups	Forestry	Intergovern-mental	Total
Local Development Fund (Conseille de Consultation)	An administrative council tasked with the selection or projects and allocation of funds raised by a voluntary tax paid by IFO on processed timber. The second day of the platform's thrice yearly two day meeting is concerned with wildlife issues.	5	2	4	12	1	1	25
Platform for the sustainable development of forests (PGDF)	The first civil society platform, of APV-FLEGT. Civil Society Organisations participate at the national level, and branches operate at the local level.	3		8				11
Framework for consultation of civil society organizations and indigenous peoples (CACO REDD)	The second civil society platform, CACO REDD of REDD+, is similar in structure to CACO REDD, using it as a model for implementation.	3		8				11
l'Association pour la Gestion de l'Aire Communautaire de Liouesso (AGACL)	A community wildlife project implemented by the FAO, with the goals of developing and implementing a participatory management plan.	1	1		1	1	1	5
Project for the management of boundary ecosystems of Odzala-Kokoua National Park (PROGEP-PNOK)	A multilateral partnership between the forestry company (IFO), the Wildlife Conservation Society (WCS) and the state, including armed ecoguards.	1	1			1		3

Platforms

We found five major collaborative platforms at Ngombé (table 3). Wildlife was a major focus of three of these. The Local development Fund spends one day of its thrice yearly meetings focusing on wildlife issues, and involved the largest number of stakeholders, and the greatest diversity of stakeholder type. The FAO project, AGACL, is far smaller and focused on only one village, and has fewer members, but also includes a diverse range of stakeholders. PROGEP, which has the greatest capacity of any of the platforms, includes only three members. The two CSO platforms, the PGDF and CACO-REDD, contained CSOs and state actors, and are not focused on wildlife, but forest governance more generally.

Village contact

Villages were most commonly in contact with traders, who visit several times a week to buy meat, and facilitate hunting by selling ammunition and batteries for torches used for night hunting, as well as other consumables. All three members of PROGEP were amongst the most frequent visitors, after traders. IFO was the most common PROGEP visitor, but although visits sometimes included wildlife related objectives, often they concerned forestry, such as during audits and Free Prior and Informed Consent (FPIC) activities required by the FSC. The ecoguards were the most common visitor whose activities related primarily to wildlife management. WCS staff were less common visitors, as were African Parks, whose activities are restricted to villages close to the park boundary. The FAO only visited three villages, but visited those villages relatively often. CSO visits were not mentioned by any villagers. When CSOs do visit villages, they often accompany the forestry company and others as part of joint missions, and so their presence may not have been salient to the communities.

Table 6. Number of days in the previous year in which organisations were present in 11 villages in the concession, according to people living in the villages. Traders includes both dedicated traders and opportunistic traders, such as long-distance taxies.

Actor	A	B	C	D	E	F	G	H	I	J	K	L	Villages visited	Mean days
Traders	350	350	350	350	260	350	260	350	260	350	350	350	8	328.2
IFO	5	5	12	8	.	2	1	2	2	7	2	12	12	5.8
Ecoguards, under PROGEP coordination	1	5	.	.	.	14	.	1	4	8	11	4	8	4.7
FAO	.	.	.	.	.	.	.	3	.	6	.	15	3	2.3
WCS	1	5	.	1	.	1	2	2	1	.	3	3	9	2.3
African Parks	.	.	.	.	.	3	.	1	1	4	1	.	5	1.3
Micro-development projects	.	.	.	4	.	.	.	.	.	3	.	.	2	0.8
MEFDD	.	1	1	.	.	.	.	.	.	.	.	.	2	0.3
Subprefecture	.	.	.	.	.	.	.	1	.	.	.	1	2	0.3

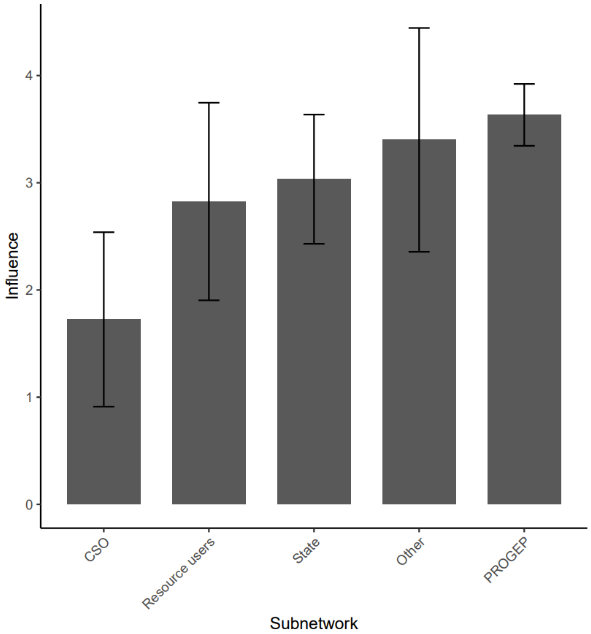

Figure 4. Influence of different subnetworks, as rated by other stakeholder organisations on a zero to five scale. Bars show standard deviation.

Influence

The perceived mean influence of stakeholders in each of the subnetworks was lowest for the CSO subnetwork, and highest for PROGEP (figure 4). Resource users were perceived as being the second least influential subnetwork. Each of the four more influential subnetworks tended to contain a range of more or less influential members. The way in which each subnetwork is able to exert influence appears to be quite different, and sometimes antagonistic to other subnetworks.

While PROGEP have the legal authority and greatest capacity to influence wildlife management, their actions are also constrained, for some of the reasons already mentioned, such as the extreme hardship full enforcement would place on communities, difficulty in policing a very dispersed activity, and corruption. The “Other state” subnetwork’s influences derives from their ability to resist unpopular wildlife management initiatives in the case of the subprefecture, and in deciding whether or not to convict people arrested for wildlife crimes in the case of the courts. Weighted degree, the sum of the weights of all links a stakeholder has to other stakeholders, was significantly correlated with influence (figure 5A), suggesting that actors in contact with more different actors and more frequently are more influential ones. Betweenness centrality was not correlated with influence (figure 5B), suggesting that actors linking isolated members of the network to more central members, possibly because the network is small and most members were well connected to the most central members.

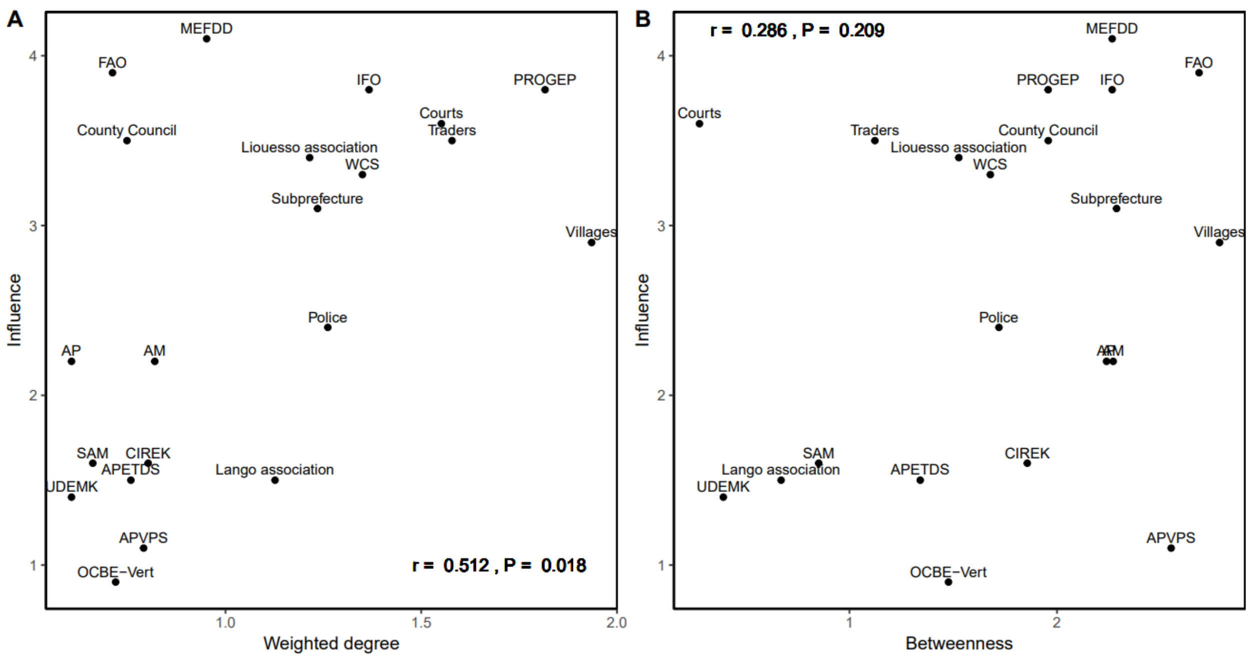

Figure 5. Influence in relation to network metric estimates for each stakeholder, showing weighted degree centrality (A) and betweenness centrality (B)

Discussion

In the Congo basin, the co-occurrence of forestry and wildlife has meant that wildlife management has increasingly become a component of forestry over the last few decades. This period saw substantial changes to forestry and wildlife management, as national governments in the Congo basin, in concert with European, American, and multilateral actors, sought to realise benefits from natural resources, while conserving them. Wildlife, once largely under the jurisdiction of local communities, became the property of a state that had no capacity to enforce its claim. Wildlife became a de facto open access resource, and populations of large game species were depleted near urban areas. To try to rectify this situation, forestry companies have been incorporated into collaborative wildlife management arrangements with the state and an international NGOs, in order to share resources, expertise, and legal legitimacy. CSOs with a range of remits grew in number and eventually became organised by platforms associated with bi- and multilateral forestry governance initiatives, but despite their interest in wildlife had only a peripheral role in its management. Finally, the FAO pilot projects are attempting to implement CBNRM schemes that have the potential to allow resource users to participate meaningfully in wildlife management for the first time in many decades.

Today, at the local level, these processes are realised in wildlife governance that is distributed across several subnetworks. First, a resource-user subnetwork harvests and trades wildlife from remote villages to urban centres, a situation that has existed for over a century. However, over this period hunting has intensified as roads linked villages to towns, as hunting technology proliferated, and as village institutions were eroded, or perhaps were simply inadequate to deal with this change. Now the PROGEP subnetwork attempts to control hunting in the forest where it occurs, and along roads and rivers where the bushmeat trade is channelled, but with a limited engagement with villages. This arrangement pits wildlife managers and resource-users against one another, a conflict exacerbated by a hunting law that is poorly suited to the economic and social reality of Congo basin livelihoods. Three other stakeholders concerned with wildlife, the FAO, the forestry ministry (MEFDD), and African Parks were found in separate group, despite not appearing to form a cohesive group in the real world. A CSO subnetwork had little influence, and was largely excluded from wildlife management, despite its members independently undertaking some activities concerning wildlife. A final subnetwork of state actors which includes local government, judiciary, and police, wielded influence due to their authority, but had a much more limited role in wildlife.

Various collaborations and platforms have been introduced to try to increase coordination in order to address these challenges. The most cross representative platform, the LDF, has thus far been relatively ineffective in its mission to provide alternative livelihoods (Rickenbach and

Garcia 2015), but is nevertheless the largest venue for the largest number and range of stakeholders to meet on wildlife issues. The FAO CBNRM project, the first to place communities at the core of a cross-representative wildlife management platform, was still nascent at the time of this survey and its effectiveness impossible to judge. Stakeholder organisations perceived factors related to network function to be less important than the lack of alternative livelihoods, the commercial nature of hunting, and inappropriate wildlife laws. The fundamental constraints imposed by poverty and a lack of alternative livelihood options means increased involvement of stakeholders may represent increased costs with few benefits. In addition, a lack of funding and capacity constraints probably limits the ability of a number of stakeholders to participate effectively. This appears to be the situation of the numerous CSOs present, on which a number of stakeholders including the CSOs themselves, commented.

If the CSOs present capacity for involvement in wildlife management is questionable, the FAO CBNRM bushmeat project's case is more compelling. The prevailing wildlife management strategy at Ngombé of policing wildlife crime cannot be fully implemented. Policing a concession area over 10,000km^2 is not technically feasible when hunting per se is not illegal, and prosecuting hunters to the full extent of the law is extremely difficult because to do so would result in serious conflict with the local population (Ampolo et al. 2017). Current education programs from PROGEP serve the role of informing communities about wildlife laws, including close seasons, gear, and species restrictions, but unless the economic reliance on hunting can be reduced, education alone will not significantly alter behaviour. The alternative, the inclusion of communities and resource users into wildlife management poses a very different set of challenges that are explicitly challenges of governance, rather than of management.

Finding ways to resolve the governance challenges posed by the inclusion of communities is the goal of the CBNRM pilot project implemented by the FAO. The project's goals include addressing hunting legality, determining the conditions required for sustainable commercial hunting, and supporting villages in taking collective action to better manage wildlife. An earlier survey (Mavah 2011) found that a lack of collective action was considered the most important factor in wildlife management by village groups. While it has been argued that granting resource-users management rights and recognising the livelihood importance of bushmeat may be necessities if wildlife is to be sustainably managed (Brown 2008), the effectiveness of CBNRM systems in the Congo basin forests has not yet been demonstrated. If the FAO project is able to provide evidence that CBNRM is feasible, the contribution of the project to the future of wildlife management in the Congo basin could far outweigh its current impacts.

In doing so it will have to negotiate the same difficult economic and legal context faced by PROGEP. A meaningful transfer of management rights to villages would require an explicit admission that the de facto and de jure rules are not the same. Some stakeholders expressed fear that authorising the commercialisation of hunting will exacerbate it, and so CBNRM represents a challenge to the protectionist status quo of wildlife management in forestry concessions (Conrelis et al. 2017). If hunting occurs clandestinely because it is illegal, it will remain difficult to manage (Brown, 2008), but because most hunters routinely disobey wildlife regulations, expecting hunters to create and abide by community laws that do not contravene national wildlife law is not realistic. CBNRM, facilitated by third parties, and in a more flexible legal framework could present a more pragmatic alternative. In this scenario, the current system in which ecoguards police hunters by attempting to catch them in rule breaking, could be replaced by one in which they assist communities to enforce rules they have set themselves, i.e. when someone hunts in contravention of locally established rules.

At the national and regional level, management of forestry concessions has rarely reached the level of sophistication seen at Ngombé, and many remain under-regulated. However, a number of the initiatives seen at Ngombé have spread and increased their impact, and seem likely to continue to do so, whereas the influence of others may have reached their limits. There are fears that declining demand for tropical timber in Europe and increasing demand in Asia, where concerns for forestry sustainability are lower, combined with lax forestry enforcement, may stall or reverse progress in forestry management (Karsenty and Ferron, 2017). Market saturation of FSC timber means the FSC's direct influence is unlikely to extend further than the concessions it already certifies. However, the government of Congo voluntarily pledged under its voluntary partnership with the EU to have all timber exports meet FLEGT standards, including that destined for non-EU markets. However, FLEGT has faced major problems in Central Africa due to being overambitious (Karsenty and Ferron 2017). An independent forest monitoring unit established in Congo in 2006, funded by the EC, DFID, and others, now allows for some limited monitoring of concessions, including those not subject to FSC audits. Forest management plans have also begun to demand the establishment and funding of ecoguard units (ACFAP 2018), and between 2008 and 2018 the number of concessions with active units grew from two to seven.

Conclusion

This case study illustrates the growth in the number and kind of stakeholders in wildlife management that has occurred in Congo over recent decades, and some of the local, national, and international factors that contributed to them emphasising the special role of forestry. Examining the historical and institutional context of forestry governance can inform our understanding of how wildlife is managed today. Because of the limitations of a case study, constituting a single data point, inference is necessarily limited. However, the range of forestry and wildlife management regimes across the Congo basin, including industrial concessions and community forest, and the presence or absence of FSC, management plans, co-management schemes, and ecoguard units, as well as origin of owners and markets in Asia and Europe, could allow for comparative studies of wildlife stakeholder networks, a first step towards identifying the most successful arrangements. This would increase our understanding of how wildlife is managed over vast areas, and beyond a handful of more well studied, and better managed, concessions in Congo.

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Chapter 3: Environmental Uncertainty and Self-monitoring in the Commons: A Common-pool Resource Experiment Framed Around Bushmeat Hunting in the Republic of Congo

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Abstract

Bushmeat is often a common pool resource issue and is a major threat to wildlife in west and central Africa. Participatory monitoring systems have been proposed to both better monitor natural resources and to engage resource users in Community Based Natural Resource Management systems, in a variety of social-ecological systems. However, studies of self-monitoring schemes in bushmeat hunting systems are scarce, and there are no empirical studies of the impact of self-monitoring on bushmeat hunting. We used a lab-in-the-field common pool resource experiment framed around a bushmeat hunting system, in which participants made individual decisions on time allocation between hunting and farming under three different conditions: without communication between group members, with communication, and with communication and a self-monitoring system. We found that self-monitoring was associated with a lower level of hunting and lower rate of resource decline. However, contrary to expectations, communication alone was not enough to lower hunting levels. We draw on behavioural economic and psychological research on environmental and social uncertainty and self-perception to explore how the act of self-monitoring could have changed behaviour by changing how participants perceived the resource, each other, and themselves. Our results support the notion that hunter self-monitoring could be a useful tool to initiate behaviour change, as well as providing estimates of resource trends

Keywords: Common-pool resource; Self-monitoring; Wildlife conservation; Bushmeat; CBNRM; Experimental economics; Dictator game; Demand effects

1. Introduction

The hunting of wildlife for meat, or "bushmeat", is one of the most urgent threats to wildlife in the tropics, driving many species towards extinction (Ripple et al., 2016). Bushmeat hunting is a Common Pool Resource (CPR) dilemma, although rarely explicitly treated as such (but see Mavah, 2011 and Rickenbach, 2015). CPRs are natural or manmade resources in which yield is subtractable (i.e. the resource can be depleted through overexploitation) and exclusion is difficult but nontrivial (i.e. restricting people's access to it is difficult, but not impossible. Ostrom et al., 1992). Tropical forest lands are often the property of the state, which almost always lacks the means to enforce the law (Wilkie and Carpenter, 1999) while traditional means of management have been undermined by loss of customary land rights (Mavah, 2011; Walters et al., 2015), or overwhelmed by economic, demographic, and technological changes, in many cases leaving bushmeat a de facto open access resource with limited enforcement of restrictions on hunting (Bennett et al., 2007).

Community Based Natural Resource Management (CBNRM) has been proposed as a means to meet these governance challenges (FAO, 2011). According to Nelson et al. (2008), interest in CBNRM "is rooted in the empirical failures of strictly centralized natural resource management policies and practices, broader trends in favour of decentralization in rural development and economic policy, and the desire to create stronger synergies between local economic interests and global conservation objectives". Self-monitoring is a form of locally based monitoring (Danielsen et al., 2009), in which estimates of resource use and/or trends are produced using records of resource harvesting as data. Self-monitoring is one possible component of CBNRM that has received significant attention in the bushmeat literature, with a number of documented implementations (e.g. Sirén et al., 2004; Noss et al., 2005; Rist et al., 2010). Monitoring, specifically involving monitors who are, or are accountable to, resource-users, appears to be critical to successful CBNRM and is included in Ostrom's (1990:94) design principles for successful management of commons, derived primarily from the extensive literature on the governance of fisheries, community forestry, and irrigation systems.

Evidence from resource systems other than bushmeat suggest that participatory monitoring can be both a cost-effective method for producing information on resources, and a platform for strengthening governance systems through the processes of empowerment and integration of resource users into decision making (Danielsen et al., 2005a, 2005b). A recent review of 35 studies of volunteer environmental monitoring (Stepenuck and Green, 2015) found an array of positive effects, including increased social capital (i.e. economic and social benefits), influence on natural resource management policies and practices, and increased community awareness. However, changes in attitudes and behaviour were only observed in five of these studies. Changes resulting from participatory monitoring schemes have included an increase in the number of locally initiated interventions aimed at conserving natural resources (Topp-Jørgensen et al., 2005), an increase in compliance with rules relating to resource use, and increased trust between stakeholders (Rijsoort and Jinfeng, 2005). Noss et al. (2005) note the usefulness of self-monitoring schemes in wildlife management, and propose that participatory methods can provide the "inputs and framework" for community level discussions about wildlife management, even when they do not provide highly accurate assessments of short-term changes in wildlife resources.

Despite this interest there are no empirical studies of the impact of self-monitoring on wildlife management performance. Economic experiments can provide a means of investigation (Ostrom, 2006), and framed field experiments, in which resource users participate in a representation of their own real-world resource system, have been used to explore human behaviour in a number of CPR systems (Cardenas and Carpenter, 2008). Because they include the resource users themselves as subjects, they have the potential to reveal behaviour in response to a broad range of factors specific to the case in question (van Vugt, 2009), which may diverge from those predicted (Ostrom, 2006).

Uncertainty is inherent to many CPR systems (Hine and Gifford, 1996) and social and environmental uncertainty are the major sources, including in bushmeat hunting systems. Each raises different problems. Environmental uncertainty is mainly a problem of optimality or efficiency, whereas social uncertainty is mainly a coordination problem (Messick et al., 1988). People must not only try to understand what is the best way to harvest a resource (i.e. find extraction rates that are profitable but do not destroy the resource), but also whether or not other people will cooperate in this strategy, and if not, how this in turn changes the optimal harvesting solution.

Most research on CPR dilemmas has been conducted under some social uncertainty, in which the intentions and actions of others are imperfectly known, usually by concealing the harvesting behaviour of individuals and only reporting aggregate group harvest. In general, reducing social uncertainty seems to increase cooperation, i.e. Sell and Wilson (1991), while a common social identity, reduction in group size, commitment, and feed-back on others behaviour can also increase cooperation (Van Dijk et al., 2004). The majority of CPR experiments provide a context of very low environmental uncertainty i.e. the size and rate of replenishment of the resource is known at all times, and group harvest level is reported (Cardenas, 2004; Janssen, 2013). Experimental research into the effect of uncertainty has found that when faced with uncertainty in CPR experiments, people tend to increase harvest rates (Hine and Gifford, 1996). Several reasons for this effect have been posited (Van Lange et al., 2013), including over-optimism or over-estimation of resource size (Gustafsson, 1999; Rapoport et al., 1992), the undermining of efficient cooperation (De Kwaadsteniet et al., 2006), and providing an excuse for non-cooperative behaviour (Van Dijk et al., 2004).

A number of studies have also tested social and environmental uncertainty simultaneously. Messick et al. (1988) found that allowing communication between players made decision making more optimal in a task with both social and environmental uncertainty. In a game setup somewhat close to a real natural resource situation, Janssen (2013) found that when players in a spatially explicit CPR experiment had complete information about resource size and players' harvest rates, their own harvest rates were higher than when they had only incomplete information. In this case it appears that being aware that others are harvesting at a high rate spurs people to do the same, and so the effect of combined social and environmental uncertainty may be unpredictable.

This paper aims to investigate the effect of self-monitoring on wildlife hunting, one of the most commonly proposed CBNRM approaches for wildlife management, using an experimental behavioural economics approach. Specifically, we tested how resource extraction rate in a CPR experiment (henceforth "game") differed under three conditions: (i) without communication, (ii) with communication between rounds, and (iii) with communication between rounds and a Self-Monitoring system (henceforth SM, and 'SM with communication'), in which participants (henceforth 'players') could voluntarily produce a public visual record of their hunting effort, success and failure at the end of each round. To do this, we modified an existing CPR game to more closely approximate a wildlife harvest system. We did this through the addition of environmental uncertainty, about resource size and regeneration rate, and by making the probability of harvesting success dependent on the size of the resource. In this manner, players could only learn about the resource through the process of harvesting, a situation analogous to most bushmeat harvest systems. We are not aware of any other study that has tested the effect of SM experimentally, or that has carried out a common pool resource experiment with bushmeat hunting communities.

2. Hypotheses

We considered hunting at a low level to reflect cooperative behaviour, because it supports the group-level objective of maintaining a productive resource, which is ultimately most profitable to the group. Conversely, hunting at a high level was considered to reflect uncooperative behaviour, because it risks resource collapse in an attempt to maximise personal profit at the expense of the group. The experiment was guided by the following hypotheses, H1: Communication would increase cooperation, and H2: SM would further increase cooperation. We expected players to hunt the least in this condition. We hypothesised that hunting would occur at a lower rate in the two conditions where communication was permitted as there is substantial evidence finding communication reduces harvesting in CPR games (Ostrom, 2006). Increased cooperation was expected to result in higher group earnings. However, due to a number of factors, including empirical findings elsewhere (i.e. Janssen, 2013), and the fact that SM was voluntary and open to abuse as players could intentionally use it to try to manipulate competitors, the alternative was also feasible, i.e. H3: SM would not improve cooperation. In addition to our central question, we further hypothesised that socioeconomic characteristics of players and psychological factors would influence behaviour.

3. Methods

3.1. Study Location and Socio-economic Context

The game was played in 10 villages within Forest Management Unit (FMU) Ngombé in the Northern Republic of Congo. The rural population is mostly made up of several Bantu and Bayaka ethnic groups, living in settlements on roads or major rivers. Bayaka includes a number of ethnic groups often referred to as Pygmies (Lewis, 2002), although it is now illegal to use the term in Congo. Unlike elsewhere in the region, Bayaka live in permanent settlements alongside Bantu, rather than as hunter-gatherers as they did in the past and as is often the case when Bayaka populations are described in the literature (Fa et al., 2016). Livelihoods in this area generally consist of a mix of farming, hunting, fishing, and casual labour. For many people hunting remains both a major source of protein and one of the only immediate means of earning cash income. Although Bayaka can still be seen using traditional hunting tools, the vast majority of hunting is carried out using modern methods. Bayaka tend to use snares rather than shotguns, while Bantu tend to use shotguns more. Bushmeat is consumed in the villages, but much of it is sold to traders who transport it to markets in urban areas (Hennessey and Rogers, 2008). While some forms of hunting are allowed in Congo, hunters routinely disobey regulations, by hunting at night with torches, using metal snares, hunting in the closed season, hunting protected species, and hunting without a license. However, despite the presence of ecoguard patrols, full enforcement of hunting regulations is technically challenging, politically complicated, and would place extreme hardship on communities. At the same time, management of hunting at the village level is virtually non-existent. Mavah (2016) argues that traditional modes of wildlife management were undermined, and new ones prevented from developing, by the abolition of customary land rights in the Public Land Law of 1983. Because of these factors, hunting in this area, as in much of the Congo basin, is largely a de facto open access resource.

3.2. Study Design

We carried out a Common Pool Resource experiment, in the form of a game framed around bushmeat harvesting. We use the following terminology to describe it:

- **Game:** The standardized experimental set-up, including instructions, which did not change between sessions, aside from the experimental condition.
- **Condition:** The three experimental conditions (Table 1).
- **Session:** The game played once. Each session had five players.
- **Group:** The five players in one session.
- **Round:** Each session comprised 10 consecutive rounds (described as "years").
- **Turn:** During each round, every player took a turn, one at a time, in which they anonymously chose to divide 12 units of effort (described as "months") between hunting and farming.

We played 30 sessions, with a total of 150 forest dwelling people from 10 different villages, the majority of whom were currently hunters, and all of whom had some experience of hunting. All sessions were played between May 2015 and January 2016. In our game, players independently and anonymously chose how much effort to expend on either hunting from a shared animal population, or farming. Players did not have difficulty understanding this set-up, because hunting and agriculture are two of the most important livelihood activities in this region. This framed field experiment was based on a forest harvest game (Gatiso et al., 2015; Janssen et al., 2013) but with the resource and harvesting modified to better represent wildlife population and hunting dynamics.

Table 1. Experimental conditions with number of sessions and players.

Condition	Rules
10 sessions, each with 5 players: No communication	No communication was permitted between players at any point during the game.
10 sessions, each with 5 players: Communication	Players had 2 min between rounds in which they could discuss whatever they wanted.
10 sessions, each with 5 players: SM and communication	Players had the option of reporting their hunting effort and success/failure using a board and counters between rounds, and had 2 min in which they could discuss whatever they wanted.

The order in which the different conditions was played in each village was randomised. No individual participated in more than one game. Players were chosen randomly when possible and opportunistically when it was not; i.e. when a player dropped out, or when villages were small and it was necessary to involve everyone available. Before playing the game, a village meeting was held in which the project objectives were explained. Potential players were told they would play a game about hunting, that it would take 3 to 4 h to play, that they would earn a participation fee of 1500 CFA (~2.70 EURO), and that they would earn more money depending on how they played the game.

Before playing, each group received training on how to play the game. The instructors followed a script, so all training sessions were as similar as possible (Appendix 1A). Efforts were made to reduce all elements of the game to simple concepts, to make the game as intuitive and easy to understand as possible, without requiring difficult calculations. Players played two practice rounds during training, and had to demonstrate understanding of the game to progress to the next part of the training. During the practice rounds, players made decisions publicly, and so were able to see and understand how all parts of the game functioned. At the end of each training session, players were asked questions to assess and demonstrate their understanding of the game's key concepts. Players who could not answer the questions correctly were replaced (two players out of 150).

While playing practice sessions we noticed that even slight modifications of the instructions could result in very different behaviour during the game. We thought this could be due to a demand effect, whereby players used the game instructions as a cue to how they were "supposed" to behave, and played the game accordingly, and that this desire to behave “correctly” was caused by the presence of a white European researcher (Cilliers et al., 2015). We tested this possibility by playing the Dictator Game 20 times in one village (10 men and 10 women, Appendix 1B). The Dictator Game is a simple economic experiment commonly used to measure altruism (Cardenas and Carpenter, 2008), in which one anonymous player is given a sum of money (in this case 4000 CFA = 6.15 EURO), and must choose what proportion to gift to a second anonymous player. Gifts approaching 50% are thought to indicate altruism, while those approaching 0% indicate selfishness. We found significantly larger gifts in the presence of a white researcher and Bantu assistant than in the presence of two Bantu assistants (40% of stake given to an anonymous member of their community with a white man present versus 8.9% with only Bantu present ($F = 39.013$, $P \leq 0.001$, $N = 20$). We therefore removed the white researcher from all phases of the game, although he was still present in the village during the experiments.

The game was played over 10 rounds (or “years”), and all five players took a turn in every round. We informed players that there would be 10 rounds. Players chose to expend 0 to 12 units of effort (“months”) to hunting in each round, with the remaining effort dedicated to farming. Hunting was not always successful, and the likelihood of success depended on the number of animals remaining. Farming was always successful. Although in reality farming success is also likely to fluctuate, we chose this set-up because in this area farming success is not affected by prior farming activity in the same way that hunting is, nor is one person's success dependent on the farming behaviour of others. A successful hunt was worth 50 CFA (0.08 EURO), an unsuccessful hunt 0 CFA, and each month of farming was always worth 10 CFA (0.02 EURO). Players hunted by drawing at random from a sack, which always contained 100 marbles. Red marbles signified a “kill”, and black marbles signified a failed hunt. There were 80 red marbles at the beginning of the game, and the maximum possible was 100. Players were made aware of this during instruction. The total number of marbles remained constant, but the ratio of red to black marbles changed as a function of number of animals killed and regeneration at the end of each round. The ratio of red to black marbles drawn by players is analogous to Catch Per Unit Effort (CPUE) often used in natural resource monitoring (Rist et al., 2010); e.g. if a player dedicates 10 months to hunting and draws 8 red marbles (“kills”), then he might infer that there are still a lot of animals left (~80% of maximum), but if he only draws 2 he might infer there are few left (~20%). We began with 80 marbles so hunting always had an element of chance, even at the beginning of the game. Decisions were made in private and earnings told to the player at the end of his turn. The player then returned to join the others in the waiting area where a researcher was also waiting to ensure players did not communicate, except during allotted communication phases. Although players took turns to harvest the resource sequentially, they knew that in each round all players faced the same conditions.

In the Communication and SM with communication treatments, players had a two-minute period in which they could discuss anything they wanted. We restricted communication to 2 min based on practice runs, in which communication typically did not last this long. We used Pearson correlations to test whether players communicated more about resource decline as the animals remaining became fewer. In the SM with communication treatment, they also had access to a board divided into strips, and black and red counters corresponding to the black and red marbles. In each round, they were able to place the counters on the board and so publicly record their hunting success and failure e.g. If they went hunting four times and were successful twice and unsuccessful twice, they could place two red counters and two black counters on the board. They were shown this during instruction, and had to demonstrate their understanding by accurately reporting one practice turn, and also reporting a turn inaccurately, to demonstrate that they understood they could also use the system dishonestly.

At the end of each round the total number of animals killed by the group was deducted from the number remaining, and a number of new animals added based on the number of animals remaining. Regeneration was calculated using a density dependent logistic growth model, as is often used in simple population models (e.g. Robinson and Redford, 1991), rounded to the nearest whole number:

$$Growth = r * \frac{(K - N)}{K} * N$$

where growth is the number of new animals added to the resource, r is a constant growth rate, N is population size, and K is carrying capacity. We used a growth rate of 0.4 and a carrying capacity of 100 animals. Maximum regeneration was set at 10, amounting to a maximum sustainable harvest of 2 animals per hunter per turn, and occurred at 50% of the maximum population (50 animals. Fig. 1). Therefore, growth was highest when it was near 50% of maximum, and was lowest when the resource was near zero or 100%. All players faced the same growth function. This was explained to the players (with reference to ecological processes), but the numerical growth rate was not, as we reasoned that in the real-world information about the state of wildlife resources is always uncertain. Players were also never certain of the number of animals left, and were not informed of the number of animals killed by others or by the group combined, and so players could only infer the resource state via their hunting success. At the end of the game, every animal left was shared equally between players (50 CFA per animal = 10 CFA per player per animal), representing the potential future value of the resource. We chose to share the remaining animals between players because in real life people value a healthy resource after they retire from hunting, either as a source of food, family income, or nontangible benefits, for themselves and for their descendants. This is an incentive for cooperation, provided other players also cooperate. Therefore, maintaining an animal population size of around 50% had three benefits to players:

1. A high rate of regeneration, and so an increase in the total number of new animals added to the resource over the game and hence a higher total value of the resource.
2. A higher success rate when hunting than when the resource is depleted (but not when it is above 50%), and hence a higher income for a given time spent hunting.
3. A higher payoff at the end of the game, as all remaining animals are shared between players.

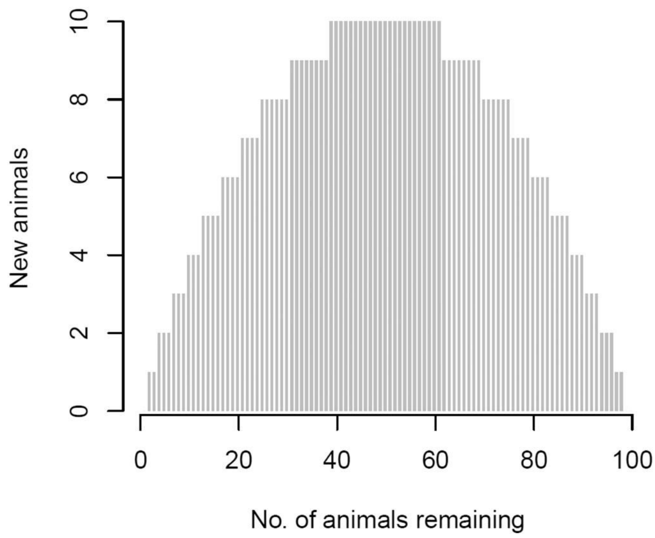

Fig. 1. The stock regeneration in our game, calculate with a growth rate of 0.4 and carrying capacity of 100 animals.

Players answered questions to ensure that they understood these benefits. Players also had to demonstrate that they understood that the maximum payoff would accrue to the group if all players kept hunting to a sustainable level, but that each individual player could earn more by increasing his own hunting; i.e. they were facing a CPR problem.

At the end of the game each player's earnings were calculated as income from every successful hunt, every month spent farming, and the share of all remaining animals. After the game each player completed a questionnaire, which included questions about ethnicity, education, time they had lived in the village, age, income from different activities, value of livestock owned, the combined value of all household assets worth 20,000 CFA or more, area farmed, and familial relationships to other players.

3.3. Subject Characteristics

All participants were male. Our 150 subjects were Bayaka (51%), Bantu (46%), and other (4%). Bayaka tended to have lower incomes, livestock assets, and household assets, be more dependent on hunting than Bantu (Table 2). Mean schooling was 4.7 years, with 41% having three or less years, below which people tend to be illiterate.

Table 2. Subject characteristics. Monetary values reported as CFA (1 euro =~655 CFA).

	Bantu	Bayaka	Other
N	69	77	4
Mean years in village	14.0	19.6	12.8
Mean age	33.1	31.4	34.8
Mean years of schooling	6.7	2.9	6.0
Median hunting income	160,000	189,000	0
Median total income	332,500	250,000	452,500
Median livestock assets	15,000	0	2500
Median items assets	55,000	15,000	58,750
Median field area (m2)	675	0	450
Primary source of income			
Hunting	49%	80%	25%
Agriculture	14%	4%	25%
Fishing	22%	8%	50%
Other	15%	8%	0%

3.4. Statistical Analysis

We used Generalized Linear Mixed Models (GLMM. Baayen, 2008) with different response variables and predictors in different models. We tested for serial correlation using the Wooldridge test (Wooldridge, 2002), and found positive serial correlation (chisq = 303.33, df =

10, p-value < 0.001), meaning OLS estimates of standard errors would be smaller than true standard errors. We therefore opted to use a single mean value for each player across the whole game, rather than one for each round of the game. The full list of predictor variables is presented in Table 3, and the model specifications in Tables 4 and 5. Our response variables for each of the three models were:

- Time spent hunting versus time spent farming over the course of the game (binomial distribution and logit link). This is possible in R using a two-column matrix of hunting and farming per turn as the response. Less time hunting indicated more cooperative behaviour.
- Time spent hunting versus time spent farming in the last turn minus the previous three turns, to test for an end game effect.
- Total game earnings of each player (Poisson distribution and log link). Players in more cooperative groups expected to earn more.

To test for an end game effect we ran a model on a subset of data, which included only the last four rounds of the game. For the response variable we subtracted last round behaviour from the mean behaviour of the previous three rounds, to yield a single normally distributed response variable. This model structure was otherwise identical to model 1.

We log or square root transformed skewed covariates, and then z-transformed all co-variates to a mean of zero and a standard deviation of one (Aiken and West, 1991). We included observation (player) nested within session, and village as random effects. The sample size for this model was a total of 150 players. We used Pearson correlations to test whether hunting effort in the different rule conditions was correlated within a village; i.e. if villages that hunted at a higher level in one condition also hunted at a higher level in the other conditions, and to test whether players communicated more about resource decline as the animals remaining became fewer.

The models were fitted in R (R Core Team, 2016) using the function glmer of the R package lme4 (Bates et al., 2015). To test the significance of our models we used likelihood ratio tests (Dobson and Barnett, 2008), comparing the fit (deviance) of a full model with the fit of a reduced model (Forstmeier and Schielzeth, 2011), comprising only the control variables and the random effects (including the random slopes). We checked for influential cases by excluding cases one at a time from the data and comparing the model estimates derived for these data with those derived for the full data set. We found no overly influential cases. Variance Inflation Factors were derived using the function vif of the Rpackage car (Forstmeier and Schielzeth,

2011) applied to a standard linear model excluding the random effects and random slopes. This did not indicate collinearity to be an issue (maximum VIF: model 1 = 1.41, model 2 = 1.43, and model 3 = 1.44).

Table 3. The predictor variables included in the generalized linear mixed models. Not all variables were included in every model. See model results tables for which variables were included in each model.

Variable	Description
Predictors	
Condition	The three experimental conditions, discussed at length in Sections 1 and 3.2 of the main text: 1) No communication. 2) Communication. 3) SM with communication
First round hunting effort	The player's number of months dedicated to hunting in the first round of the game. First round behaviour is less constrained by factors that are internal to the experiment, and so a truer indicator of a players innate propensity to cooperate or not.
Number of animals killed by the rest of the group	The total number of animals killed over the course of the game by the rest of the group. When considering earnings, the most important factor effecting an individual's outcome is the behaviour of the rest of the group. This is fundamental to CPR dilemmas.
Age	The age of the player. Age has been linked to cognitive traits such as risk aversion and patience.
Education	Number of years of school attendance of the individual. Years of education is linked to cognitive capacity and numeracy, and so may influence performance
Ethnicity	The ethnic group that the individual identified with.
Hunting income	The player's absolute annual income from hunting.
Hunting dependence	A variable constructed from non-hunting cash income, area farmed, and value of livestock owned. All hunters were ranked for each variable, and all three rankings were summed. Ranks were summed, so that hunters who earned money from other sources, farmed large areas of land, and kept livestock had the lowest scores.
Size of household	The number of people living in a household, defined as a group of people sharing meals and residing together.
Items Assets	The total value of all household items worth 20,000 FCFA or more.
Participation in cooperative	Whether the player had contributed either money or time towards a cooperative, such as an agricultural project.
Time in village	Number of years living in village.
Relatedness	The relatedness of a player to other members of their group. We used reported relatedness as a proxy, and assigned each relationship a value based on expected average genetic relatedness for that relationship; i.e. 0.5 for siblings or parent/son relationships, 0.25 for uncles/nephews or cousins. Relatedness is predicted by evolutionary psychology to be a major determinant of cooperation.
Extended family	The number of non-blood familial relationships in the group for each, i.e. relationships such as "little brother" or "uncle" where no blood line could be established. These relationships are common in the study area and in many parts of Africa, and might be expected to represent a stronger association, and hence cooperativeness, than other kinds of non-blood relations.
Controls	
Game order	The order in which the session was played. We played three sessions in a village on consecutive days. We included this variable to control for learning between games, for example if later groups benefited from hearing about the game from individuals who had already played.
Turn order	The order in which the player took his turn within the group. In each round, each took a turn. The order that each player took their turn was the same in every round. We included this variable in case there was an influence of turn order on hunting level.
Random effects	Individual, session, village were included to control for the hierarchical structure of observations.

4. Results

4.1. The Effect of Condition on Hunting Effort

We used effort invested in hunting versus effort invested in farming by each player over the course of the game as the response variable in the first model. As each is the inverse of the other, we will refer only to time invested in hunting, as "hunting effort", for the purposes of discussion. A low time investment in hunting indicates cooperative behaviour. Players dedicated between 0 and 12 effort units ("months") to hunting per round (Fig. 2). Individual hunting effort ranged from 13 to 90 months over the course of an entire game, from a potential maximum of 120, and group hunting effort ranged from 135 to 394 months, from a potential maximum of 600. The full model was highly significant (likelihood ratio test comparing full and null model: $\chi^2 = 54.01$, df = 15, P < 0.001. Table 4). SM with communication reduced the likelihood of choosing hunting over farming by 43% (estimate = −0.43, SE ± 0.07, $\chi^2 = 16.526$, P < 0.001. Post-hoc test: z = −5.76, P < 0.001), but hunting effort in the No communication and Communication conditions were not significantly different from each other. We found a significant effect for real world hunting income, which had a small positive effect on hunting level, a highly significant effect but small negative effect of relatedness, and a highly significant and small to moderate positive effect of first round hunting level. The model testing for the presence of an end game effect found no significant effect (likelihood ratio test comparing full and null model: $\chi^2 = 15.703$, df = 16, P = 0.47).

4.2. Village Level Correlations

There was no relationship at the village level between mean hunting effort in the No communication condition and the Communication (r (8) = 0.02, p = 0.95) or SM with communication conditions (r (8) = 0.06, p = 0.87. Fig. 3). However, there was a very strong correlation between hunting effort in the Communication and SM with communication conditions (r(8) = 0.98, p < 0.001), suggesting a strong effect of village, aside from those variables included in the model, that mediated how individuals played the game.

Table 4. Results of the GLMM in which the response variable was a two-column matrix of time spent hunting versus time spent farming, with a positive response indicating an increase in hunting.

Response variable: time hunting versus time farming

Predictor variable	Estimate	SE	χ^2	df	p
Intercept	0.18	0.09	NA[a]	NA	NA
Communication	−0.09	0.08	16.526[b]	2	< 0.001***
Self-monitoring	−0.43	0.07			
Ethnicity: other	0.37	0.24	1.821[c]	2	0.40
Ethnicity: Bayaka	−0.02	0.07			
Age[d,f]	−0.01	0.03	0.114	1	0.72
Size of household[d,f]	−0.03	0.03	0.764	1	0.37
Years in school[f]	−0.02	0.04	0.112	1	0.72
Hunting income	0.08	0.03	6.592	1	0.01**
Hunting dependence	−0.05	0.03	1.755	1	0.15
Value of assets[d,f]	0.02	0.05	0.183	1	0.65
Time living in village[e,f]	0.02	0.04	0.377	1	0.49
Relatedness[d,f]	−0.11	0.03	11.047	1	< 0.001***
No. friends in group[f]	−0.01	0.03	0.06	1	0.80
Hunting in first round[f]	0.28	0.04	17.468	1	< 0.001***
Experience in a co-op	0.10	0.08	1.444	1	0.20
Game order[f]	0.09	0.04	3.763	1	0.02*
Turn order[f]	0.01	0.03	0.111	1	0.73

* = $p < 0.05$.
** = $p < 0.01$.
*** = $p < 0.001$.
[a] Not shown because of having a very limited interpretation.
[b] The test refers to the overall effect of rule condition as obtained from comparing the full model with a reduced model lacking it.
[c] The test refers to the overall effect of Ethnicity as obtained from comparing the full model with a reduced model lacking it.
[d] log transformed.
[e] Square root transformed.
[f] z-Transformed.

Table 5. Number of animals remaining and earnings at the end of the game, and their increase over the No communication condition.

	No. animals remaining			Earnings		
Rule	Mean	SE	Increase (%)	Mean	SE	Increase (%)
No communication	14.7	4.5	.	2119	563	.[a]
Communication	19.6	4.7	33%	2247	561	6%[b]
SM with communication	31.6	6.8	115%	2563	630	21%[ab]

Although within condition earnings were significantly higher in SM with communication than in No communication, the variable as a whole did not contribute significantly to the model, and should be treated with caution.

[a] Indicate significant differences in earnings.

[b] Indicate significant differences in earnings.

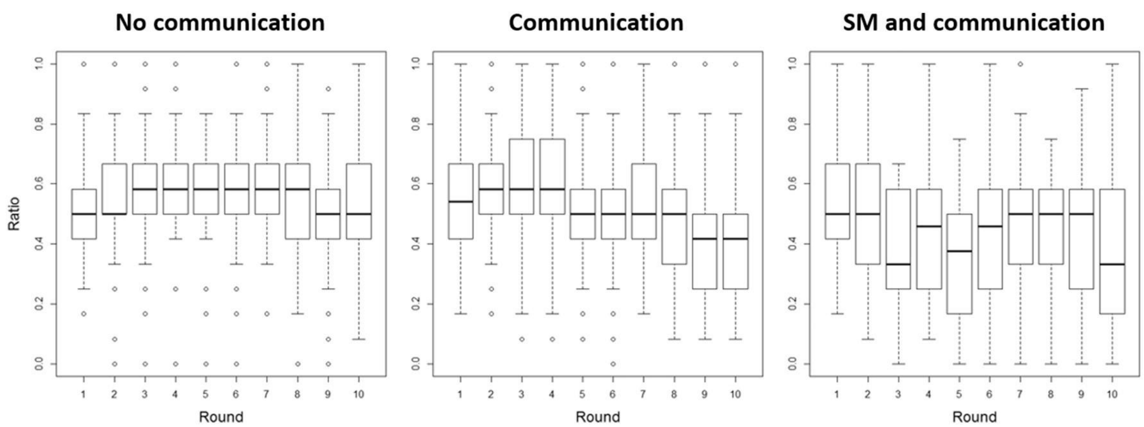

Fig. 2. The ratio of hunting to farming in each round in each condition. Median shown as solid line, top and bottom of box upper and lower quartiles. Whiskers indicate 1.5 times the inter-quartile distance, outliers shown as points.

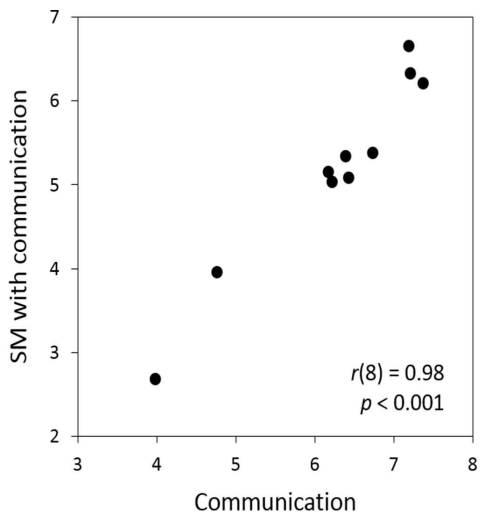

Fig. 3. Mean hunting effort in each village was correlated in the Communication and SM with communication conditions but neither were correlated with the No communication condition.

4.3. Accuracy of Self-monitoring Reporting

Inspection of the data suggested that dishonesty when reporting catches to the group was rare, with most reporting being accurate. The presence of both under and over-reporting suggests that error rather than dishonesty may have accounted for some of the under-reporting. Catch was reported correctly in 81.4% of turns, under-reported in 12% and over-reported in 6.6%. By comparing transcripts of discussions during the games and records of reporting, we noticed only one occasion in which a player intentionally misled their group by hunting at a high level, while reporting a low level and strongly advocating for reducing the group hunting level.

4.4. Resource Depletion and Earnings

The resource declined over the course of the game in all conditions (Fig. 4 and Table 5). In all conditions, resource decline was fastest at the beginning of the game, and appeared to have reached an equilibrium by the end of the game. At the end of the game, the remaining resource ranged from zero to 81 animals, and the number of new animals generated over the course of the game ranged from 29 to 100. Mean group earnings ranged from 1534 FCFA to 2600 FCFA, meaning that players in the most cooperative group earned 70% more than in the least cooperative group. Individual earnings ranged from 1390 FCFA to 3250 FCFA, the highest earning individual earning 134% more than the lowest. The highest individual earnings accrued to a player in the game with the largest range in earnings, who defected while the rest of the group was generally cooperative. This happened in the SM with communication condition and the player used the monitoring system to manipulate other players. The defecting player earned 85% more than the player in the group who earned the least. The model using individual earnings as the response variable was highly significant (likelihood ratio test comparing full and null model: $\chi^2 = 57.352$, df = 16, $P < 0.001$. Table 6).

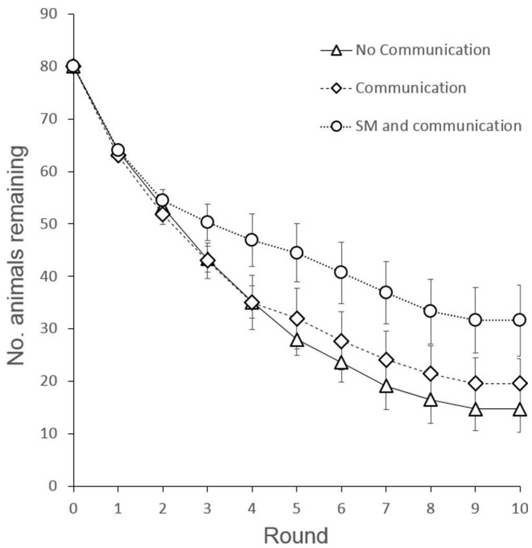

Fig. 4. Resource size at the end of each round in each condition, with standard error bars.

Although individual earnings were 21% higher in the SM with communication condition, and 6% higher in the Communication condition, than in the communication condition, this term was not significant. This may be due to an insufficient sample size, or number of rounds played. However, the hunting level of other players had a large impact on an individual's earnings with an increase of one SD in animals killed by others resulting in a fall in earnings of around 20% for the individual (estimate ± SE = −0.206 ± 0.010, z = −20.248, $P < 0.001$), and condition did predict hunting level in the previous models. Increasing relatedness to other players increased earnings (estimate ± SE = 0.011 ± 0.003, z = 3.376, P = 0.001), while individual hunting effort in the first round decreased earnings (estimate ± SE = −0.012 ± 0.005, z = −2.505, P = 0.012).

Table 6. Results of the GLMM in which the response variable was individual player earnings.

Response variable: earnings					
Predictor variable	Estimate	SE	χ^2	df	P
Intercept	7.526	0.057	NA[a]	NA[a]	NA
Communication	0.114	0.125	1.728	2	0.421[b]
SM with communication	0.217	0.059			
Ethnicity: Other	−0.016	0.016	−0.483	2	1.000[c]
Ethnicity: Bayaka	−0.007	0.009			
Age[d,f]	0.006	0.005	0.802	1	0.370
Size of household[d,f]	0.005	0.005	1.070	1	0.301
Years in school[f]	0.001	0.005	−1.481	1	1.000
Animals killed by group	−0.206	0.010	37.800	1	< 0.001***
Hunting income	< 0.001	0.003	−1.570	1	1.000
Hunting dependence	0.002	0.010	0.133	1	0.715
Value of assets[d,f]	−0.002	0.003	0.249	1	0.618
Time living in village[e,f]	−0.004	0.005	−0.479	1	1.000
Relatedness[d,f]	0.011	0.004	5.201	1	0.023*
Extended family[f]	0.001	0.004	−1.615	1	1.000
Hunting in first round[f]	−0.005	0.005	4.041	1	0.044*
Experience in a co-op	−0.001	0.009	−1.322	1	1.000
Game order[f]	−0.06	0.068	−1.492	1	1.000
Turn order[f]	0.00	0.003	−1.476	1	1.000

* = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$.

[a] Not shown because of having a very limited interpretation.

[b] The test refers to the overall effect of condition as obtained from comparing the full model with a reduced model lacking it.

[c] The test refers to the overall effect of Ethnicity as obtained from comparing the full model with a reduced model lacking it.

[d] log transformed.

[e] Square root transformed.

[f] z-Transformed.

4.5. Communication

Players used the communication period to discuss a range of issues (Table 7). Unfortunately, due to small sample it was not possible to test the effect of communication on game outcomes, but we report raw data and broad patterns where possible. All groups that had the option to communicate or monitor did so. 88% of individuals participated in communication in both monitoring and SM and monitoring conditions, and all players with the option to self-monitor did so. Players referred to the natural resource dilemma in the majority of games (i.e. "We will live to see the consequences of our poor management", "We have to cooperate", and "We need a strategy") indicating that they indeed understood the game situation. Discussions about hunting in the game sometimes concerned the mechanics of the game i.e. "hunting is a waste of time when the resource is depleted" and "we should reduce hunting so the resource can recover", but also often referred to factors relevant to real hunting, but not hunting in the game i.e. "The government and NGOs are right to tell us to reduce hunting" and "We need to rest, because we don't go to the forest every day [in real life]". No players shared how many animals they had taken verbally (aside from when they caught zero), meaning that in Communication treatment players could only infer the resource state form their own hunting success rate, and through other's estimation of resource state, such as "the animals are few now" or "hunting is hard now". Players were aware when the resource declined, and spoke more about resource decline as the remaining resource became lower in both treatments allowing communication (Communication: $r(98) = -4.0$, $p < 0.001$. SM with communication: $r(98) = -4.5$, $p < 0.001$).

Table 7. Topics players communicated about. Numbers indicate the number of times that subject was spoken about by a single player

	Communication			Communication and monitoring		
	% games (*n* = 10)	Mean per game	SD	% games (n = 10)	Mean per game	SD
Speakers per round		1.8	0.22		1.9	0.33
Signal agreement	80	2.6	1.41	90	1.6	0.75
Signal disagreement	10	0.1	0.45	40	0.7	0.85
Attempt to give order to another player	20	0.3	0.70	30	0.4	0.57
Signal individual intent	70	1.2	0.84	80	1.1	0.66
Signal respect for another player	90	2.2	1.42	100	2.6	1.49
State game is going well	0	0.0	0.00	20	0.5	1.03
State resource is overexploited	90	2.8	1.14	80	2.6	2.26
Call to reduce hunting - vague	100	4.0	1.86	100	5.6	2.38
Call to reduce hunting - specific	90	2.4	1.90	90	2.2	1.28
Call to reduce hunting - total	100	6.4	2.65	100	7.8	3.74
Call to Increase hunting - vague	40	0.4	0.24	40	0.9	1.11
Call to increase hunting - specific	50	0.6	0.43	0	0.0	0.00
Call to increase hunting - total	70	1.0	0.77	40	0.9	1.30
Reference to game mechanics	90	4.3	2.14	100	5.6	2.38
Reference to real world e.g. "we need to reduce hunting so our children will know the animals"	80	2.0	0.83	80	2.0	1.50

5. Discussion

We did not find support for H1 (communication alone would increase cooperation), and found support for H2 over H3 (Self-monitoring increased cooperation, rather than decreasing it). Most socioeconomic and psychological variables were either non-significant or had only small effect sizes, aside from first round hunting level.

5.1. Why Did Self-Monitoring Reduce Hunting?

Players mostly used the self-monitoring system relatively honestly, and it enabled them to reduce their hunting level to the benefit of the group. This requires explanation, given that there were no direct negative consequences for players who played dishonestly. In fact, players could very easily hunt at a high level, while reporting a low level of hunting, with no risk of being caught. They could even leverage their reputation as a responsible hunter, created through false reporting, to try to manipulate others into reducing their hunting, so allowing them to claim more of the resource themselves. However, this happened egregiously on only one occasion, when one player did exactly that. The ability to sanction non-co-operators typically stabilizes group cooperation at a high level, whereas cooperation typically collapses in the absence of the ability to sanction (Gürerk et al., 2006). Although our experimental set-up did not allow for the imposition of penalties, other studies have found that when able to do so, people are generally willing to engage in costly sanctioning, with the proximate cause being strong negative emotional responses to free-riders (Fehr and Gächter, 2002). There is therefore a clear social pressure to hunt at a low level in the context of our game, but in the absence of a means of detecting freeriding, this becomes only a reason to report hunting less, rather than actually hunting less. A second finding that requires explanation is that SM with communication increased cooperation, while communication alone did not. Which mechanisms determine this behaviour is an empirical question, but several authors have proposed potential explanations for similar phenomena in CPR and other economic experiments. These explanations fall into three categories, related to how individuals perceive the resource, their group, and themselves, and are discussed below.

The first category encompasses environmental uncertainty; i.e. how people perceive the resource. Environmental uncertainty concerns both resource size and regeneration rate. Experimental studies of environmental uncertainty have found that when resource size is uncertain, there is a general tendency to over-estimate the amount of resource available for harvesting, and to increase harvests (Van Dijk et al., 2004). Uncertainty may undermine normative pressures that might otherwise promote restraint, by depriving players of the information required to operationalise a norm, even if all agree to it, thus obstructing the translation of an abstract goal (cooperation) into a certain one, harvesting less (Hine and Gifford, 1996). Self-monitoring reduces uncertainty by combining information from all players in a group, and by leaving a physical record over time. Reducing uncertainty may make it a less credible excuse for selfish behaviour. Reducing environmental uncertainty may therefore reduce selfish behaviour, even when improved estimates are not explicitly used as a basis for decisions about harvest rates.

The second category relates to social uncertainty; i.e. how people understand the behaviour and intentions of others (Jager et al., 2002). Relevant factors may include group identification, as well as communication with and social pressure exerted by other group members. Communication may help by reducing perceived uncertainty through the creation of group identity, or by eliciting social norms (Bicchieri, 2002). Individuals differ in their predispositions to cooperation. More self-centred individuals tend to defect more, because they see cooperation as offering an uncertain gain (or certain loss), and defection as a certain gain (or uncertain loss. Biel and Gärling, 1995). Reducing social uncertainty may change the perception of this balance for self-centred individuals, making gains from cooperation and losses from defection more certain. While communication is often enough to increase cooperation, this was not the case in this study, where an effect was only seen when self-monitoring also occurred. Perhaps uncertainty reduced the ability of communication alone to overcome the CPR dilemma.

The third category relates to how a person perceives themselves. Humans are social animals, and much of their evolved and learned moral psychology relates to how people should interact within groups (Cosmides, 2004). In the context of our game, players are subject to two contradictory motivations: to maintain a positive view of themselves, and to gain from cheating (Mazar et al., 2008). The act of self-monitoring entails reporting behaviour in a way that is precise rather than vague (as in the Communication condition), and so dishonesty becomes an active decision. This may draw the players attention to the moral dimension of resource use (i.e. free riding), through mechanisms such as the Self-concept threat (in which immoral behaviour threatens one's ability to consider themselves as moral individuals), Categorization (in which situational factors force one to reconstrue an action as more morally deviant than before), and Attention to Standards (in which being reminded of one's moral standards makes failing to meet them more salient and so more damaging to self-concept (Mazar et al., 2008).

Many of these explanations function by activating moral and social norms, drawing attention to them, and reducing the uncertainty that makes it easier to shirk them. We did not find that greater information about other's behaviour resulted in higher levels of resource extraction, as it has in other CPR experiments in which monitoring was externally imposed rather than carried out voluntarily by the players (Janssen, 2013). This may be related to framing, with our scheme more likely to be seen as a platform to facilitate collective action and information about resource size, rather than as a way to detect free-riders. Alternatively, inaccurate, even sometimes dishonest reporting may provide a space for trust and cooperation, even while it enables selfish behaviour, whereas complete information may serve to undermine trust because selfish behaviour is apparent to all. In such a scenario, an intermediate level information facilitates cooperation, while too much or too little information undermines it.

5.2. Psychological and Socio-economic Factors

Socioeconomic factors were mostly not significant determinants of the outcome in any of the models. We found no significant effect of ethnicity on our response variables. This is interesting, given the large differences seen between the two populations, including differences in income, hunting dependence, years in school, livestock assets, and area of agricultural land, suggesting that whatever influenced behaviour was independent of these contextual factors. Focusing on the first model using hunting effort, the most direct measure of individual behaviour, as a response variable (Table 5), we found no effect of age, size of household, or value of assets owned. Although years in school predicts literacy and numeracy, and hence many aspects of cognitive capacity, we found no effect of years of schooling. This is often the case in CPR experiments, and may reflect the fact that CPR problems are social dilemmas, as well as economic ones, and that solutions hence often are social (Kollock, 1998). We found no effect of experience with real world cooperatives although others have done so (Cilliers et al., 2015), but experience with cooperatives was generally very low in our location, and our sample size was also small compared to other studies.

Hunting effort in the game increased with real world hunting income (estimate ± 0.091 SE ± 0.03, $\chi^2 = 6.71$, $P = 0.01$), but not with degree of hunting dependence. Whether real world hunting income predicted hunting effort because of underlying psychological traits, such as hunters being less cooperative or more likely to discount the future, or a heuristic (i.e. frequent hunting is a strategy that in the real world, so it could work in the game) is unknown. First round hunting level significantly predicted subsequent hunting level (estimate ± 0.28 SE ± 0.04, $\chi^2 = 17.468$, $P < 0.001$), and reflects the effect of individual differences between players. In the same study area, Rickenbach et al. (2015) found that Bayaka tended to discount the future more steeply than Bantu, while nearby Salali and Migliano (2015) found that Bayaka discounted the future more heavily when they lived in remote villages, and less so when they lived alongside Bantu in a logging town. Our experiment may not be suitable for evaluating differences between these populations. Alternatively, the differences between populations may not be that large, because Bayaka lived in permanent settlements in all cases. We found no relationship between experience of other livelihoods, including farming, and hunting effort. The genetic relatedness of players had a small significant effect on hunting (estimate ± −0.11 SE ± 0.03, $\chi^2 = 10.78$, $P < 0.001$), with more closely related players spending less time hunting, but extended familial relationships ("little-brother", "uncle", etc.) did not, and neither did years living in the village.

The strong correlation between outcomes in the Communication and SM with communication with communication in games played in the same village is puzzling, and we are not aware of another study finding such a strong effect. There are three more plausible explanations: Collusion, chance, and an unobserved village level characteristic. We noticed no evidence of a shared strategy that would suggest players had colluded before the game began. Indeed, harvesting rates were diverse in most rounds of all games. The probability that this correlation was simply a chance occurrence was less than one in a thousand. It is possible to envisage some village level characteristic, such as trust or cooperativeness, that mediated behaviour, and that only had an effect when players could communicate, but was not captured by the individual level socio-economic variables measured. However, more obvious and measurable factors such as village size and market integration do correspond to the observed behaviour.

6. Limitations and Applications

A limitation specific to our experiment is that hunter self-monitoring systems in the real world will mostly have input from wildlife managers, who would be able to analyse data and make recommendations about extraction levels. However, wildlife resources have a number of characteristics that make quota based harvesting systems inappropriate (e.g. complexity, stochasticity, and uncertainty), and participatory, adaptive management approaches a more realistic option (van Vliet et al., 2015). Consequently, the role of wildlife professionals is less relevant to our game, in which the resource is simple so that depletion is relatively easy to detect. Real world complexities such as multi-species harvesting, spatial distribution and quality difference of patches, and the need to convert raw catch data into abundances indices, would potentially make bushmeat monitoring schemes more reliant on external support than those in other systems, such as community forestry schemes.

This experiment is also limited in the number of treatments tested. We chose treatments considered most relevant to the context: treatments that mirrored the current situation (communication without monitoring), likely real-world implementations (self-monitoring with communication), and a baseline. Other relevant questions can be envisaged with relevance to hunting e.g. different types of self-monitoring. Adding an additional treatment of Self-monitoring without communication would have enabled us to test whether the reduction in hunting was due to self-monitoring assisting with coordination, or altering perceptions of the resource i.e. by reducing social or environmental uncertainty.

More generally, two major criticisms directed towards economic experiments are that they lack realism and therefore are not generalizable, and that they are susceptible to demand effects (Levitt and List, 2007), meaning that researchers are not measuring the variables they think they are measuring. In combination, these criticisms would indicate that economic experiments are not useful for understanding the "real world". The long-term cumulative effects of dishonesty, corruption, and dissatisfaction that can undermine CBNRM for example (Nielsen and Lund, 2012), are not considered here. However, there is growing evidence that prosocial behaviour in experimental settings is correlated with real world behaviour ((Benz and Meier, 2008; Cilliers et al., 2015; Fehr and Gächter, 2002; Rustagi et al., 2010), pointing towards the existence of general across-situational traits.

Furthermore, generalizability is not solely a problem of experimental economics (Falk and Heckman, 2009), and in the case of self-monitoring, generalizing the findings of one implementation of a scheme (even across several villages) may be problematic, given the variation in social-ecological systems. It is therefore necessary to recognize the potential of economic experiments, which is to allow the testing and formulation of hypotheses in a controlled setting, with human subjects. This is particularly relevant to the governance of bushmeat harvest systems, which is understudied relative to other CPRs, and where there is an absence of CPR experiments.

In the case of hunter self-monitoring, observational data are rare, real world schemes tend to involve small numbers of villages and be short term, and measuring outcomes is difficult. Many questions are also made difficult or impossible to study in a natural setting, as bushmeat hunting is often criminalized. Theory concerning decision making in wildlife management is spread across several disciplines, including psychology, economics, and sociology (Keane et al., 2008), and the experimental method has been used extensively in addressing questions of importance in each of these disciplines. The game presented here could easily be adapted with simple rule changes, in order to study the impacts of a range of factors of interest to bushmeat researchers and wildlife management practitioners, including social (e.g. number of players, leadership, social norms, and multi-generationality), environmental (e.g. size and regeneration rate of resource, multi-species communities, and spatial management), and economic (e.g. value of resource and different forms of sanctioning) factors.

7. Conclusions

The act of self-monitoring reduced hunting effort, increased earnings, and reduce the rate of resource decline in a CPR experiment framed around bushmeat hunting. Although we can only speculate on the mechanisms by which this worked, it appears that the activity of self-monitoring encourages pro-social behaviour, supporting the notion that self-monitoring can assist in management of bushmeat hunting CPR systems by changing the behaviour of hunters (Noss et al., 2005). While studies of real-world schemes have often sought to test accuracy (Rist et al., 2008) or to describe various aspects of the scheme, such as wildlife offtake or participation rates, self-monitoring may be just as valuable for its normative effects, and its potential to facilitate community level collective action, as one component of CBNRM in bushmeat harvest systems. Although largely absent from the bushmeat literature, economic experiments have the potential to generate and test hypotheses related to wildlife governance, providing insights, which would be extremely difficult to obtain through alternative methodologies. We also highlight the importance of demand effects using the Dictator Game, and recommends that researchers undertaking experimental studies should consider carefully how to avoid these when planning their experiments.

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Chapter 4: Using wildlife indicators to facilitate wildlife monitoring in hunter-self monitoring schemes

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Abstract

Wildlife populations in tropical forests are difficult to monitor. Hunter self-monitoring schemes hold promise, but their accuracy in estimating populations has not been verified and obtaining useful wildlife estimates from generally low-quality data remains a challenge. We tested whether wildlife indicators could be useful for wildlife monitoring in such schemes, because they might eliminate the need to estimate effort in hunter surveys, and reduce records of many species into a single informative variable. We implemented a hunter self-monitoring scheme in eight villages in the northern Republic of Congo, collecting shotgun, snare, and camera trap records in "zones" within each village's hunting territory (shotguns = 83 zones, snares = 50 zones, cameras = 21 zones). Using each of these three survey methods, we calculated for each zone three different indicators used in wildlife studies: mean body mass, the mean intrinsic rate of increase (*r*max), and a duiker index (small duikers as a percentage of total duikers). Survey effort could be estimated for both snares and cameras and was used to estimate species relative abundances (Catch Per Unit Effort, CPUE). Mean body mass was the most effective indicator, followed by the duiker index. Both were correlated between survey methods and changed with increasing hunting pressure regardless of survey method used. They also predicted total CPUE in kg for zones, and often the CPUE of individual. They also gave the most precise estimates of the three indicators, and snare estimates were more precise than shotgun. In contrast, mean *r*max generally performed poorly, and was often not correlated with the other indicators, or with hunting pressure. Our findings suggest that some indicators can produce useful wildlife estimates from

hunter self-monitoring schemes, that are also easy to implement and comprehend for hunters and wildlife managers.

Keywords: Bushmeat hunting, tropical wildlife, self-monitoring, wildlife indicators, Community Based Natural Resource Management (CBNRM).

Introduction

Hunting of wild animals for protein and income is important to the livelihoods of many rural people in tropical forest areas (Nielsen et al. 2017; 2018), where it is often referred to as bushmeat hunting. Across the tropics, 27.5% of forests are designated as protected areas, but in the Congo Basin, the proportion falls to only 16.3% (Nelson and Chomitz 2011). In the remainder of the ~2 million km^2 Congo Basin forest estate, hunting is permitted under various restrictions (Ziegler et al. 2016). However, legislation regulating wildlife hunting is poorly aligned with local realities, and enforcement is technically and politically difficult. As a result, overexploitation will likely result in the collapse of many wildlife populations.
Unless measures are taken to halt this trend, it has been predicted that 81% of available bushmeat sources in Congo Basin countries will be lost by 2050 (Fa et al. 2003).

Population monitoring is critical to wildlife management but remains a challenge for ecologists and conservationists (Burton et al. 2015). Monitoring is costly and difficult, particularly for species that are mobile, scarce, and cryptic, as is common among forest-dwelling species. Wildlife surveys in tropical forests are usually carried out using line transects, which are poorly suited to surveying game species as many game species are rarely encountered and hence likely to be underestimated (Fragoso et al. 2016). Furthermore, records of animal signs are difficult to identify (van Vliet et al. 2008) and decay rapidly, which makes conversion into abundance estimates problematic (Schwarz and Seber 1999). Camera trapping is increasingly used and tends to generate many records of target species, but estimates are prone to bias due to detection rates that differ by species and across time and space (Sollmann et al. 2013, but see Howe et al. 2017), and data processing requires substantial effort and specialised skills.

Hunter self-monitoring has been proposed to improve wildlife monitoring (Noss, 2004; Rist et al. 2010). Hunter self-monitoring schemes are examples of locally based monitoring in which hunting provides encounter data for wildlife abundance estimates. Self-monitoring schemes are thought to have advantages over more conventional methods, including generating large amounts of data at low cost (Rist et al. 2010) and focusing on spatial scales and species relevant to hunters. Locally based monitoring schemes may also have benefits of importance in resource management, including integrating resource users into the resource management process, greater acceptance of wildlife estimates by local people involved in producing them (Danielsen et al. 2005), and catalysing locally initiated action to protect resources (Danielsen et al. 2010). Hence, self-monitoring has the potential to be beneficial to both resource users and resource managers and improve the sustainability of bushmeat harvest systems.

Hunter records have been used to produce various kinds of estimates, including absolute and relative hunting offtake (Noss et al. 2004), modelling species occurrences (Parry and Peres 2015), and relative abundance estimates including Catch Per Unit Effort (CPUE) (Kümpel et al. 2010). However, the accuracy of such estimates produced by hunter self-monitoring schemes or hunter interview surveys have only rarely been examined, and the majority of studies do not attempt to verify their accuracy by comparison with other methods (e.g. Noss et al. 2004, de Mattos Vieira et al. 2015, Paula et al. 2017). Studies that attempt verification do so only indirectly (Peres et al. 2006, Parry and Peres 2015), or assess how accurate hunters are in reporting hunting activity instead (Rist et al. 2010). Rist (2007) attempted to verify a self-monitoring scheme by comparing CPUE to density estimates from transect surveys. CPUE is a relative abundance estimate that divides catch by effort (e.g. three monkeys killed in two hours of hunting gives an estimate of 1.5 monkeys per hour of hunting). The higher this number, the easier it is to catch an animal, and this number should be positively correlated with the absolute abundance of a given species or group of species in a location. However, of five species for which Rist (2007) obtained sufficient data two had significant positive relationships, two non-significant relationships, and a fifth a significant *negative* relationship. That is, as this species became more abundant according to transect records, it became less abundant according to hunting records. These contradictory results likely stem from a challenge facing validation of self-monitoring schemes accuracy: both self-monitoring schemes and any survey method to which they are compared are usually relatively inaccurate.

CPUE in hunter self-monitoring schemes is generally inaccurate because estimating the effort component of hunting is not trivial. It is particularly difficult for a hunter using firearms to estimate how much time he has spent actively hunting (e.g. stalking as opposed to in transit to the hunting ground) during a hunting trip, and this may not be consistent between trips. Accurately reporting the location of hunts is not as difficult, but still a source of error (Rist et al. 2009), and technological solutions are likely to add significant expense to schemes and require a certain level of expertise to use and to analyse. As a consequence, while CPUE has a number of attributes that make it attractive for self-monitoring schemes (i.e. simple to calculate, correlated with absolute abundance, and easy to interpret), the challenge of obtaining reliable estimates of hunting effort mean that relative abundance estimates such as CPUE are less than ideal for schemes staffed mainly by local members of hunting communities, especially where snare hunting is rare or prohibited.

Ecological indicators may provide a pragmatic alternative to CPUE. Indicators can be calculated from a subsample of animals caught or encountered, and do not require any estimation of hunting effort, but only a record of the animal encounter and its location. Researchers have made use of several indicators to monitor wildlife in the Congo basin, including mean body mass (Ingram et al. 2015), the intrinsic rate of increase (Fa et al. 2015), and recently a duiker index (Yasuoaka et al. 2015). Each of these indicators depends upon on the empirically supported assumption that large and slowly reproducing species are more vulnerable to hunting and will decline or disappear faster with increasing hunting pressure (Wright 2003). At the same time, small species that reproduce quickly are more resistant to hunting will decline more slowly, or even increase under hunting pressure through competitive release. These indicators are simple, and therefore easy to calculate, and potentially easily communicated and understood by wildlife managers and hunters, which are essential features if they are to be widely used.

We tested the ability of a hunter self-monitoring scheme to provide accurate estimates of the status of wildlife populations, using three indicators: mean body mass; mean intrinsic rate of increase; and the duiker index (cf. above). First, we attempt to validate the accuracy of hunter records by comparing them to camera trap records. We reasoned that if the estimates of a single indicator were correlated across hunting zones regardless of the survey method, that indicator was reliable. Second, in the absence of unbiased wildlife estimates to compare our estimates to, we compared the indicators to a proxy of hunting pressure, estimated from human population density, to determine if they could be useful for wildlife monitoring. Third, we assessed the response of individual species to each indicator, by comparing CPUE estimates from camera and snare surveys where effort could be estimated to indicator estimates in each zone. Fourth, we assessed the precision of each method and indicator under different sampling effort, to evaluate reasonable sample sizes and variation in the ability to detect change.

Methodology

Study Location

This study was undertaken in the Forestry Management Unit (FMU) Ngombé, in the Northern Republic of Congo, in collaboration with the forestry company operating the concession, *Industrie Forestière de Ouesso* (1° 7' 27.8256" N16° 0' 19.1808" E). Hunting takes place with shotguns and snares. Although only around 5000 people live within the FMU, much of the bushmeat is traded to the adjacent large town of Ouesso (Hennessey and Rogers, 2008. Population ~30,000). Hunters live in villages along the major roads in the concession, and the major rivers that form its Northern and Eastern borders. Hunting takes place on short trips from the village or multi-day hunting trips from semi-permanent hunting camps in the forest. Aside from hunting, the main livelihood activities are agriculture and fishing. Between June 2015 and August 2017, we implemented a hunter self-monitoring scheme in 8 villages in the concession (Table 1), with a minimum of six months of monitoring in each village. We selected villages that were spread across the entire concession and covered a gradient of human population density.

The Participants

Participants in the scheme totalled 227 hunters and eight village monitors (henceforth monitors). Monitors were trained to record information on data sheets during our initial visit to a village. All participating hunters were given an ID number to provide anonymity, and the only record combining names and ID numbers was kept by the monitor. Once a month we visited villages to pay monitors and collect data sheets. At the same time, we checked data sheets for errors, re-trained monitors when necessary, and called a meeting in which we presented project results, received feedback, and paid participating hunters. Monitors were

Table 1. Village sample characteristics: village population size, hunting pressure proxy (described in the section Geographic and habitat data), number of participating hunters, number of animal records used in the analysis, number of zones surveyed, and survey effort estimated for each village.

				Shotguns		Snares			Cameras		
ID	Village population	Hunting pressure	Number of hunters	Animal records	Number of zones	Animal records	Number of zones	Effort (trap days)	Animal records	Number of zones	Effort (m^2 days)
1	19	49	12	361	4	77	4	4,688	196	2	13,421
2	141	228	47	796	17	322	12	31,684	418	4	30,431
3	38	558	23	585	12	326	9	26,290	189	3	18,899
4	194	622	42	297	11	206	7	33,690	90	3	13,125
5	128	3,068	26	426	15	318	10	47,021	280	3	20,401
6	93	9,907	10	364	4	226	4	21,356	112	3	12,601
7	27	119	23	643	11	31	2	3,200	.	.	.
8	157	221	44	584	9	15	2	380	198	3	15,610
		Total	227	4,056	83	1,521	50	168,309	1,483	21	390,498

paid 40,000 FCFA (~€60) on a monthly basis, and individual hunters were paid a fee per record. Fees were negotiated in each village and ranged from 250 to 500 FCFA (€0.38 to €0.76), enough to encourage their participation but small relative hunters' overheads (i.e. shotgun shells cost 800 FCFA each in this area) and therefore were unlikely to affect the level of hunting. During each visit, to minimise fabrication we randomly tested one to three hunters, by asking them to recount hunts they had previously described to the monitor. We did not detect any fabrication.

The Hunt Recording System

We pilot tested the scheme to determine a suitable recording system, including hunter follows and comparing hunt records made by hunters and monitors to records made by researchers (Appendix 2A). We found hunters reported what they killed accurately (93%, n = 41) but animals they only saw inaccurately (34%, n = 49), and were only able to consistently record the location of animal kills in relation to which hunting camp they thought was nearest (48% reports were of correct zone, 49% incorrect but adjacent zone, n = 90). Therefore, we used animal kills linked to hunting camps to calculate indices. As hunting with metal snares is not allowed by Congolese legislation, we did not accompany hunters to check snares. We chose not to do so for the same reason that we did not ask them to provide information about hunting of protected species: to minimise the real and perceived risk of participation for hunters, and to eliminate as much as possible any incentive for dishonesty. Hence, the reporting accuracy of snares is not known.

To map the territory, we first asked hunters to describe the hunting trails and camps around the village, sketching a map. We then visited these trails and camps with hunters and georeferenced them using GPS units. We used this data to produce maps that monitors and hunters could use to register hunting trips (Appendix 2B: Fig. B.1). We developed separate data sheets for shotgun and snare hunting (Appendix 2C: Figs C.1 and C.2). Both data sheets recorded the village, the hunter's anonymous ID, and the date of the record. The shotgun sheet also included which zones the hunter was in, whether they hunted at night and/or during the day for each 24-hour period they were in the forest, and any animals encountered. The snare sheet recorded the number of snares used, the location of snare line, the time between checking the snare, and any animals caught. For every animal recorded, hunters were asked to report the type of encounter (i.e. animal was killed, seen, or heard), the species, the location, and the age and sex of the animal.

Camera Trapping Survey

We surveyed 21 zones over seven village territories with camera traps (two zones in one of the villages, three zones in five villages, and four zones in one village). We tried to select zones that were actively hunted during the study period to allow direct comparison between hunting and camera records, and at different distances from each village where possible in order to cover a gradient of hunting activity. We used Bushnell camera traps (models 119435, 119476, and 119678). We used GIS to define a triangular grid centred on the hunting camp, so that camera placement was random in relation to habitat, with cameras spaced 300 meters apart. This approach was a compromise allowing a high number of cameras to be fitted inside each hunting zone, while still being larger than the home range of many of the smaller game species (although large species often range farther than this distance). In each zone, we placed up to 20 cameras for a minimum period of 14 days. However, effort was not constant due to human error and camera malfunction (Table 1). Cameras were placed 45 cm above the ground to increase the number of small mammals recorded. We estimated camera coverage as a triangular area in front of the cameras where the sensor could be triggered by movement. We did this by moving in front of the cameras while they were in their survey position, using their setup mode which responds to movement with an LED alert. All videos were viewed, and the time the video was recorded and number of animals of each species in each video were recorded. To eliminate multiple records of the same individual, we considered any video of a species recorded within one hour by the same camera to be the same individual, following Hegerl et al. (2015).

Estimating wildlife indicators and Catch Per Unit Effort (CPUE)

Because of the difference between species caught at night and during the day in shotgun hunting, and because primates are rarely caught with snares or by cameras, we used only shotgun records from night hunts. We excluded elephants from the camera data, because these species are rarely killed by subsistence hunters, and could skew indicator values substantially due to their large mass and tendency to move in groups.

For each zone we calculated three indicators: Mean body mass, mean intrinsic rate of increase (*r*max), and the duiker index, separately for each of the three survey methods: snare, shotgun, and camera trapping. We applied *r*max and mean body mass estimates for each species based on estimates in relevant literature (Appendix 2D: Table D.1). Mean body mass, here measured in kg, is used extensively in fisheries but is also proposed for large-scale analyses of bushmeat catch records in the OFFTAKE database (Ingram et al. 2015). Mean body mass (henceforth MEAN-KG) is the mean body size of all individuals recorded in a survey, and is calculated as:

$$\text{MEAN} - \text{KG} = \frac{\sum_{i=1}^{S}(\text{mass}_i * n_i)}{N}$$

Where *n* is the number of individuals of a given species, and *N* is the total number of individuals of all species (see Appendix 2D: Figure D1 for a visual representation of the indices). MEAN-KG is expected to fall under hunting pressure. The second indicator was calculated in the same way, but instead of mass was based on an estimate of the maximum per capita population growth rate (*r*max) for each species (henceforth MEAN-RMAX):

$$\text{MEAN} - \text{RMAX} = \frac{\sum_{i=1}^{S}(r\text{max}_i * n_i)}{N}$$

Species such as rodents tend to have a high *r*max, while species such as large duikers tend to have a low *r*max, so MEAN-RMAX is expected to rise under hunting pressure. Fa and colleagues have made extensive use of *r*max in long-term surveys of bushmeat markets in Africa (Fa et al. 2015), and it has also been used in the Neotropics (Antunes et al. 2016).

More recently, Yasuoka et al. (2015) proposed a duiker index, calculated as the ratio of blue duikers *Philantomba monticola* to red duikers, a group consisting of three medium-sized *Cephalophus* species (*C. callipygus*, *C. dorsalis*, and *C. nigrifrons*). Here we used a percentage instead of a ratio (henceforth DUIKER%), to account for zero values which cannot be calculated as ratios:

$$DUIKER\% = \frac{n_{blue\ duikers}}{n_{blue\ duikers} + n_{red\ duikers}}$$

As hunting pressure increases, the number of red duikers, which are larger species vulnerable to hunting, is expected to fall, while the number of blue duikers, a smaller, more hunting resilient species, remains relatively constant. Hence, DUIKER% is expected to increase under hunting pressure, as the small blue duiker makes up a higher percentage of the catch.

Because survey effort can be estimated relatively accurately for snares and camera traps, we calculated CPUE for several species for which we had sufficient data as:

$$Snare\ CPUE = \frac{No.of\ animals\ caught\ in\ snares}{No.of\ snares * No.of\ days\ between\ visits}$$

$$Camera\ CPUE = \frac{No.of\ animals\ recorded\ by\ cameras}{No.of\ minutes\ camera\ running * camera\ coverage}$$

We also calculated an aggregate CPUE for species weighing greater than 15kg, less than 15kg, and a total CPUE in kg for each zone, which was the mass of all animal carcasses of all species divided by the total effort.

Geographical and habitat data

We recorded the number of people over 16 years of age in each study village through a direct census and used population data from the forestry company for other villages in the concession. All geographical analysis was carried out in QGIS 2.18.1 (QGIS Development Team 2017).

For an indicator to be useful, it should have different values at sites that are differentially impacted by hunting. However, it was not possible to estimate this impact of hunting on wildlife directly from our data, because hunting pressure is a function of many difficult to quantify processes. These include how hunting intensity has varied historically, the rate of wildlife dispersal and reproduction, and the exponential increase in wildlife habitat area with a linear increase in distance from a village (i.e. doubling the radius of a circle quadruples its area), in addition to the current amount of hunting that occurs at a specific site. However, many studies report that wildlife is more depleted nearer to settlements (Benítez-López et al. 2017), and a wildlife survey in Gabon (Koerner et al. 2016) suggests that the impact of hunting may decline in a linear or linear-like manner with distance from the nearest village. Therefore, we calculated a rough proxy of impact (henceforth ‘hunting pressure’) from village population data, with hunting pressure at 0 km from the village being equal to the population of the village, declining linearly to zero at a distance of 25 km from the village. We chose 25km because hunters very rarely travel more than this distance from their village during hunting trips. We did not take the effect of rivers into account, because they were generally small and easy to cross, through the use of fallen trees that had become part of the route. Where hunters were based in villages on the large Sangha river, they used boats if they wanted to hunt on the opposite side.

We used vegetation data from a remote sensing survey undertaken for the forest management plan before forestry operation began (IFO et al. 2007), extracting vegetation cover for each zone. Over 99% of vegetation type was dense forest (63%) or swamp forest (36%). Hence, we excluded other vegetation types and included percentage cover of dense forest in each zone as a habitat variable.

Statistical analysis

First, we made comparisons between indicators within and between survey methods, to determine if indicators were correlated with each other (i.e. MEAN-RMAX vs MEAN-KG vs DUIKER% from snare data) and whether different survey methods were correlated with each other (i.e. MEAN-KG from snare vs shotgun vs camera records). We then compared the indices with hunting pressure, to test responsiveness to different levels of hunting. We compared the indicators with CPUE of individual species and all species combined, to test whether the indicators related to relative abundances of wildlife. We pooled records by zone, and so do not include individual hunts or cameras. All data were log or square root transformed for normality where appropriate. We used weighted linear regressions to compare all indicators calculated from all three survey methods with one another, for a total of 36 pairwise comparisons, using the R function 'lm' in the statistics package R (version 3.2.5, R Core Team 2016). We included the number of records as weights, and where we compared two methods, we used whichever number of records was lower (i.e. a zone with 50 shotgun records but only five snare records would have a weight of five).

We checked validity and stability with various diagnostic tests (Cook's distance, DFBetas, DFFits, and leverage; distribution of residuals, residuals plotted against fitted values, and heteroskedasticity). In the majority of cases, these did not indicate disproportionally influential cases, nor obvious deviations from the assumptions of normality and homogeneity of residuals (Quinn & Keough 2002; Field, 2013). In instances where residuals did not pass tests for heteroskedasticity, box-cox power transformation was performed on the response variable (Osborne, 2010), using the caret package in R (Kuhn, 2018). In cases where residuals still did not pass tests for heteroskedasticity, we do not present regressions, but only plots of the data.

We compared all methods with hunting pressure, including the habitat variable. We compared the indicators with CPUE for individual species and for all species combined, calculated from snare and camera data, in order to test whether the indicators predicted animal abundances. We used general linear models with poisson and negative binomial distributions, using the 'glm' function in the statistics package R, and the glm.nb function from the MASS package in R (Venables and Ripley 2002). We chose models passing a dispersion factor threshold of 1.2 where possible, and adjusted p-values to account for over-dispersion when it occurred (Gelman and Hill, 2007). We used trapping effort (log transformed) as an offset term in the model.

To evaluate the influence on sampling effort (number of animals killed), on the precision of indicators for each hunting method we simulated samples increasing in increments of ten from 10 to 100. We selected three zones with a high number of hunting records and with mean indicator values in the upper, middle, and lower end of the indicator range. These were also zones with high, intermediate and low levels of hunting pressure. From the hunting records for each of these zones, we randomly sampled animals with replacement, 100 times at each sample size, and estimated the intercept, 95% CIs and maximum CIs of the estimate using GLM or GLMM where we had an appropriate number of levels in the random effects. In those cases, we included hunter ID and hunt ID as random effects.

Results

Animal records

Survey methods varied substantially in the frequencies with which different species were recorded (Fig. 2, Appendix 2E: Table E.1). Camera traps recorded 24 species with 88% of records accounted for by seven species: blue duiker (27%), red duiker (17%), giant pouched rat (*Cricetomys emini*) (13%), African brush-tailed porcupine (*Atherurus africanus*) (9%), marsh mongoose (*Atilax paludinosus*) (5%), yellow-backed duiker (*Cephalophus silvicultor*) (5%), and central chimpanzee (*Pan t. troglodytes*) (5%). Snares caught 21 species, but five species accounted for 86% of the catch: blue duiker (24%), Peter's duiker (*Cephalophus callipygus*) (23%), African brush-tailed porcupine (19%), giant pouched rat (12%), and bay duiker (*Cephalophus dorsalis*) (4%). Shotgun records recorded 22 species, but 86% of records were accounted for by only three species: blue duiker (52%), African brush-tailed porcupine (25%), and Peter's duiker (10%). While cameras regularly recorded a broad range of species, including the very largest and smallest species, the hunting records were truncated at both ends of the mass and *r*max distributions, and shotgun records particularly so. Hunters generally do not target the smaller, less profitable species, while larger species may be more difficult to catch for technical or behavioural reasons, despite being more valuable.

Survey methods and indicator comparisons

Of 36 comparisons between methods (a single indicator calculated from a single survey method), 24 comparisons were significant, 9 were not, and a further three did not meet model assumptions for homoskedasticity (Table 2, and Appendix 2F: Fig. F.1). MEAN-KG was always correlated between camera, shotgun, and snare survey methods, as was DUIKER%. Eight of nine non-significant relationships and all three models that failed to meet assumptions for heteroskedasticity included both estimates from shotgun data and MEAN-RMAX (from shotguns or snares). This pattern appears to be due to the restricted range of species targeted by hunters using shotguns which means that faster reproducing species such as rats and squirrels that may increase under hunting pressure are simply not hunted. This interpretation is supported by the fact that MEAN-RMAX calculated from both cameras and snares were each linearly related in four of eight comparisons, whereas MEAN-RMAX from shotguns were in only two comparisons. Furthermore, two of the most commonly recorded species in all three surveys, blue and red duikers (for >62% of shotgun catches, >47% of snare catches, and >44% of camera records), are assigned very similar values in the MEAN-RMAX indicator. Therefore, changes in their relative abundances results in much larger changes to the MEAN-KG and DUIKER% indicators.

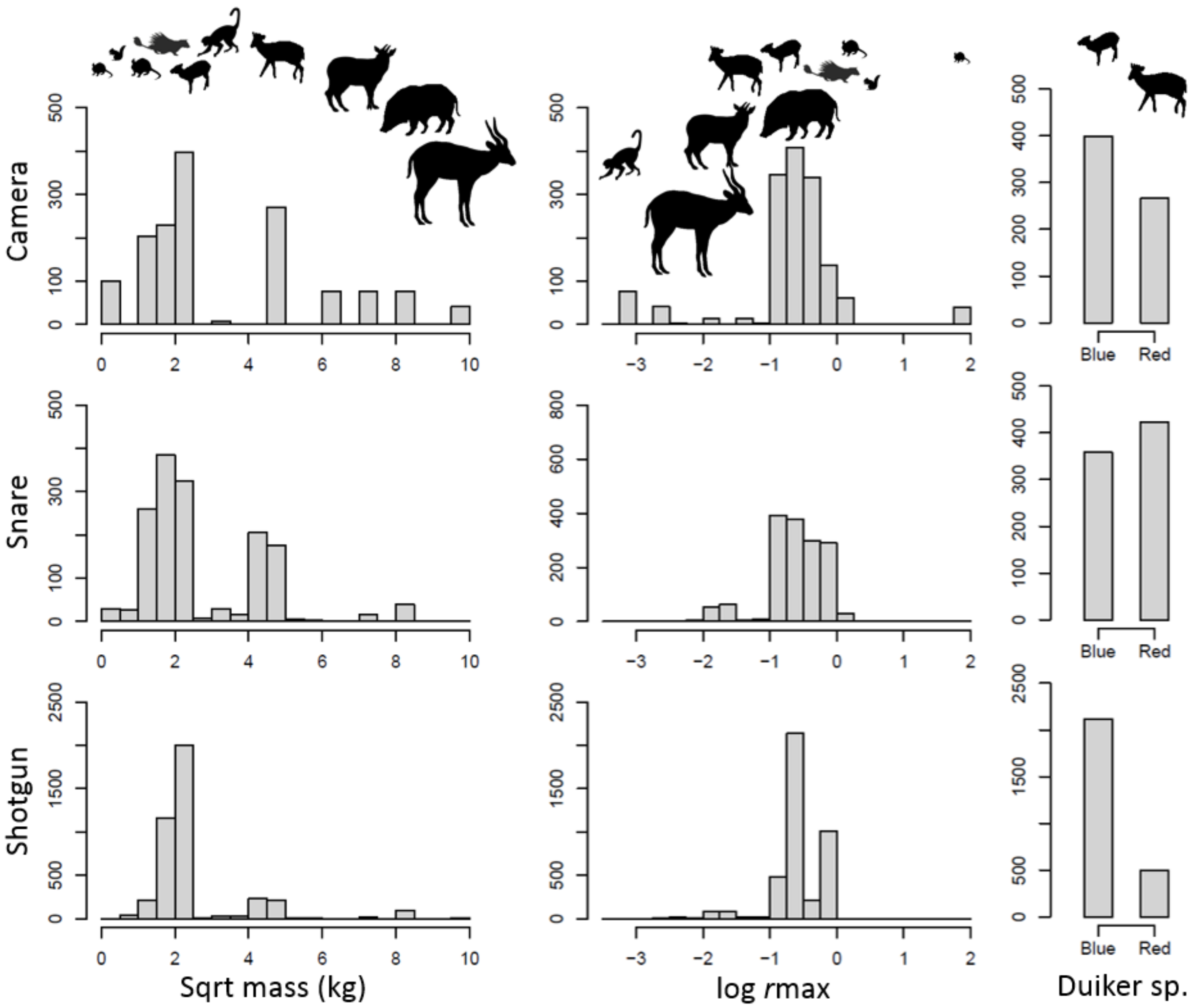

Figure 1. Histograms of animal mass and *r*max, and bar plots of red and blue duikers recorded by each survey method. Reference species are shown in ascending order of mass: mouse sp., squirrel sp., giant pouched rat, African brush-tailed porcupine, blue duiker, greater spot-nosed monkey (*Cercopithecus nictitans*), Peter's duiker, yellow-backed duiker, red river hog (*Potamochoerus porcus*), and sitatunga (*Tragelaphus spekei*).

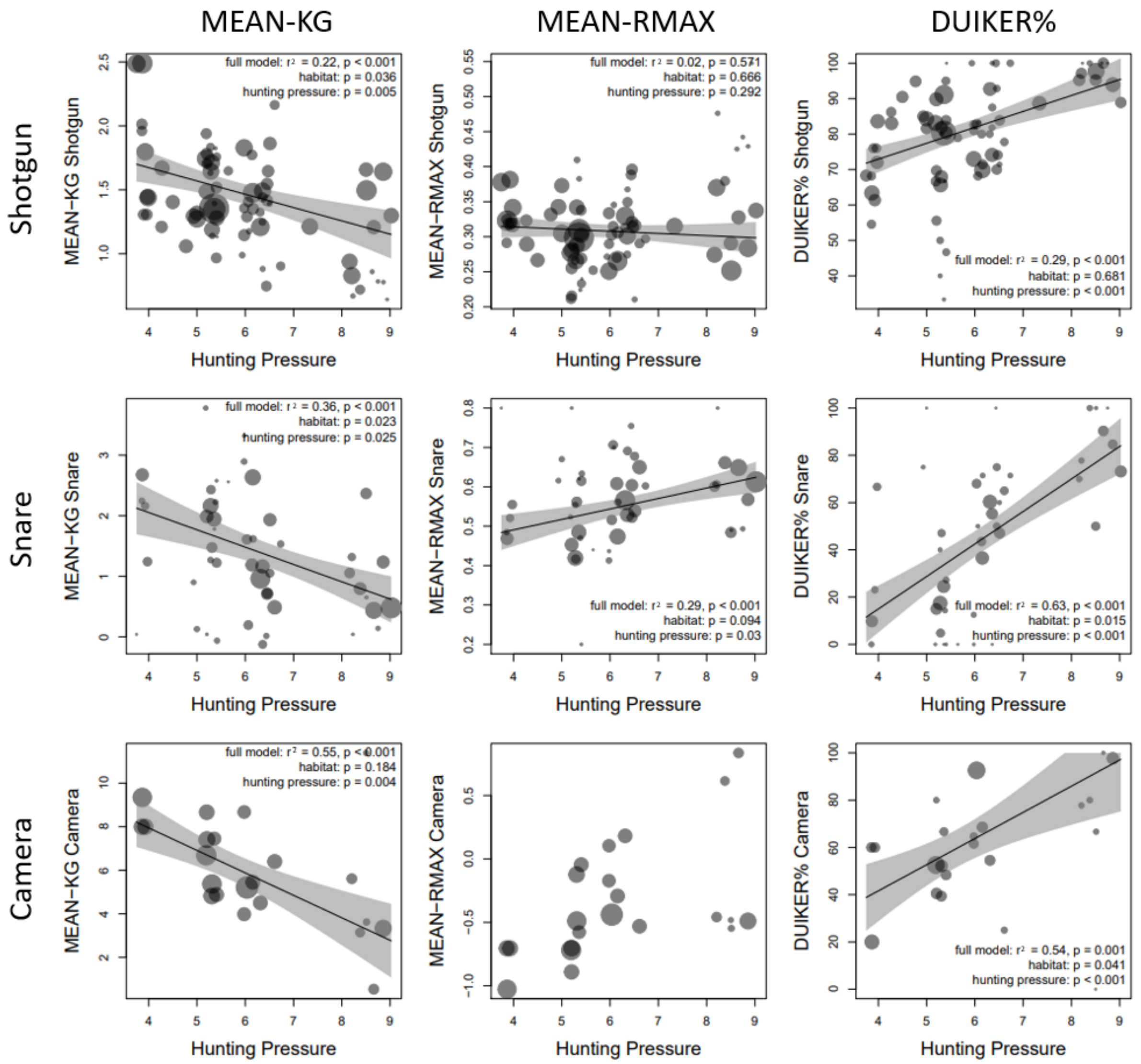

Figure 2. The response of the three indicators to hunting pressure, for shotgun, snare and camera surveys. Regression lines and 95% Cis shown, except where models failed assumptions for heteroskedasticity. The area of circles corresponds to the number of records (max = 293)

Hunting pressure predicted DUIKER% and MEAN-KG indicator estimates as expected, with DUIKER% increasing with hunting pressure and MEAN-KG falling (Fig. 2 and Appendix 2F: Table F.1). r^2 values were higher for snare and camera estimates than for shotguns, while r^2 values were higher for DUIKER% than for MEAN-KG for both shotgun and snare records. MEAN-RMAX did not perform as well, was not significant for shotgun records, was significant for snare data ($p = 0.03$), while the model for camera records did not meet assumptions for heteroskedasticity. The range in values of indicator estimates under hunting pressure was larger in snare and camera data than shotgun data, resulting in steeper regression slopes and larger r^2 estimates. Again, this appeared to be due to the restricted range of species caught by shotguns, but also the high proportion of blue duikers found in shotgun records. Where the habitat variable was significant, effect size appeared small relative to hunting pressure (Appendix 2F: Table F.1).

Catch Per Unit Effort (CPUE)

We were able to estimate effort for snares and cameras, and so could estimate CPUE which we compared to the indicators. Although we could not estimate CPUE from shotgun data, we compared indicator estimates from shotgun records with CPUE calculated from snares and cameras. We found that in most cases the indicators significantly predicted the combined CPUE of species >15 kg in both snare and camera data (table 3, Appendix 2G: Figs. G.1), while small changes for species <15kg were found in the snare data, but not the camera data. As MEAN-KG declined and as DUIKER% and MEAN-RMAX increased the CPUE of large species declined, while the CPUE of small species increased slightly or did not change. Indicators based on shotgun records predicted the CPUE of all species >15kg from snare records but not camera records, and did not predict CPUE of species <15kg (table 3, Appendix 2G: Fig. G.2).

At the species level, we found a similar pattern, with individual species >15kg often declining as the indicator changed under increasing hunting pressure, and species <15kg not changing or increasing (Fig. 4, Appendix 2G: Fig. G.3). We found that MEAN-KG most often predicted species CPUE in snare data (100% of species tested) whereas the DUIKER% and MEAN-RMAX were less effective. Indicators predicted species CPUE much more often in snare data than in camera. Indicators were a significant predictor of CPUE more often than our hunting pressure proxy. In addition to the important game species shown in Fig. 4, there was also evidence in some cases that the indicators predict the CPUE of species mainly only recorded by cameras, including small species such as squirrels (MEAN-KG) and mice (MEAN-RMAX and DUIKER%), and large species such as gorillas and chimpanzees (MEAN-KG and DUIKER%. Appendix 2G: Fig G.4).

Table 2. p values and R^2 of weighted regressions between all survey method and indicator pairings

		Shotgun						Snare						Camera			
		MEAN-KG		MEAN-RMAX		DUIKER%		MEAN-KG		MEAN-RMAX		DUIKER%		MEAN-KG		MEAN-RMAX	
		r^2	p	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p	r^2	p
Shotgun	MEAN-RMAX	†		.	.	.	.	.	.	.	.	.	.	.	.	.	.
	DUIKER%	**0.38**	**<0.001**	0.05	0.051	.	.	.	.	.	.	.	.	.	.	.	.
Snare	MEAN-KG	**0.22**	**0.001**	**0.25**	**0.001**	**0.20**	**0.001**	.	.	.	.	.	.	.	.	.	.
	MEAN-RMAX	†		**0.19**	**0.002**	**0.21**	**0.001**	**0.73**	**<0.001**	.	.	.	.	.	.	.	.
	DUIKER%	**0.34**	**<0.001**	**0.18**	**0.005**	**0.57**	**<0.001**	**0.62**	**<0.001**	**0.59**	**0.001**	.	.	.	.	.	.
Camera	MEAN-KG	**0.24**	**0.022**	0.02	0.526	**0.19**	**0.044**	**0.30**	**0.027**	0.22	0.065	**0.59**	**0.001**	.	.	.	.
	MEAN-RMAX	**0.34**	**0.004**	0.02	0.564	†		**0.26**	**0.042**	0.24	0.055	**0.42**	**0.009**	**0.46**	**0.001**	.	.
	DUIKER%	**0.38**	**0.002**	0.00	0.912	**0.39**	**0.002**	0.13	0.178	0.07	0.333	**0.45**	**0.006**	**0.35**	**0.003**	0.08	0.205

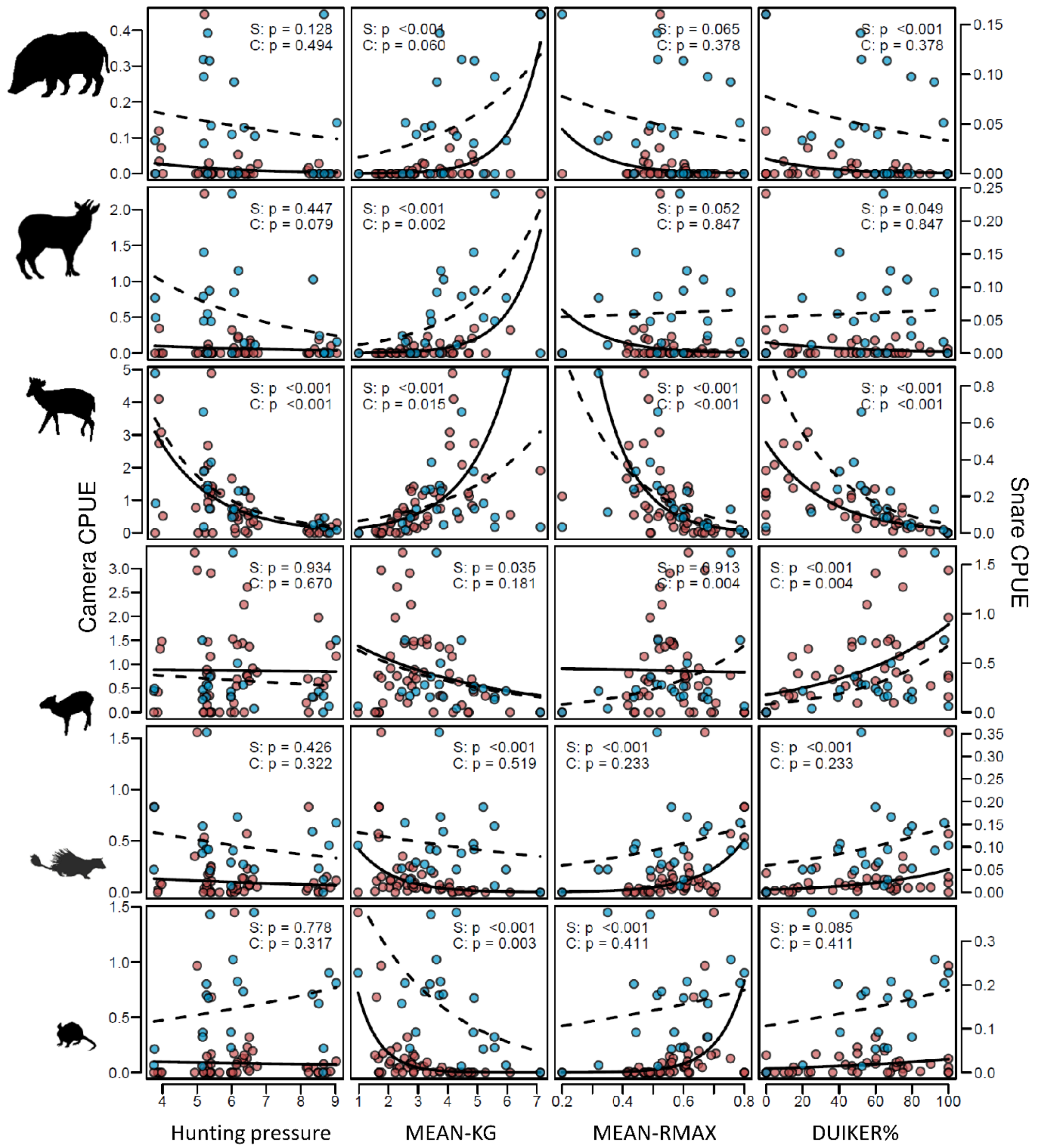

Figure 3. The relationship of indicators to Catch Per Unit Effort (CPUE) based on snare and camera data. Each point represents a zone. Snares are shown as red points and solid regression lines, cameras as blue points and dashed lines. Species shown are, from top to bottom, red river hog, yellow-backed duiker, red duiker, blue duiker, African brush-tailed porcupine, and giant pouched rat.

Table 3 Logistic regression results for comparisons of indicators from each survey method with CPUE estimated from snare and camera data.

		Under 10kg					Over 10kg				
Predictor	Response	Estimate	SE	*z*	*p*		Estimate	SE	*z*	*p*	
MEAN-KG Snares	CPUE Snares	-0.521	0.140	-3.729	<0.001	***	1.727	0.137	12.597	<0.001	***
MEAN-RMAX Snares		3.234	1.063	3.043	0.002	**	-11.582	2.028	-5.711	<0.001	***
DUIKER% Snares		0.013	0.004	3.460	0.001	**	-0.033	0.004	-8.495	<0.001	***
MEAN-KG Cameras	CPUE Cameras	-0.238	0.139	-1.711	0.087		0.747	0.136	5.479	<0.001	***
MEAN-RMAX Cameras		-0.410	0.378	-1.084	0.278		-1.571	0.388	-4.051	<0.001	***
DUIKER% Cameras		0.018	0.006	3.007	0.003	**	-0.012	0.009	-1.350	0.177	
MEAN-KG Shotguns	CPUE Snares	-0.032	0.049	-0.659	0.51		0.309	0.055	5.6	<0.001	***
MEAN-RMAX Shotguns		5.405	2.414	2.239	0.025	*	-10.069	3.948	-2.55	0.011	*
DUIKER% Shotguns		0.008	0.009	0.918	0.358		-0.04	0.018	-2.192	0.028	*
MEAN-KG Shotguns	CPUE Cameras	-0.026	0.054	-0.474	0.636		0.106	0.063	1.669	0.095	
MEAN-RMAX Shotguns		-3.074	4.446	-0.691	0.489		-3.2	5.566	-0.575	0.565	
DUIKER% Shotguns		-0.012	0.014	-0.865	0.387		-0.041	0.016	-2.608	0.009	**

Precision

A good wildlife indicator should provide different values for zones that have different wildlife communities due to different hunting pressure. The smaller the confidence interval of an indicator estimate relative to the range of that indicator's estimates, the more precise is the indicator, and the more it can differentiate between zones with different wildlife communities. Simulation of hunting offtake from high, intermediate, and low-pressure hunting zones revealed that snares were much more precise for any given survey effort (Fig. 6). This higher precision is likely due to the more restricted range of species caught with shotguns, which includes fewer very small and large animals and produced smaller absolute differences between mean zone estimates for all indicators. Snare estimates of MEAN-KG and DUIKER% were able to differentiate (no overlap of 95% confidence intervals) between high, intermediate, and low-pressure zones with few records (MEAN-KG = 20, DUIKER% = 30), while MEAN-RMAX required 90 records. For shotguns, only MEAN-KG (80 records) and MEAN-RMAX (100 records) were able to differentiate between zones in the simulations.

Discussion

This study adds to the growing body of literature on self-monitoring schemes, by demonstrating that hunting records can in some circumstances be used to produce indicator estimates that are correlated predictably with the state of terrestrial wildlife populations. However, these findings require some qualification, because not all indicators were effective,

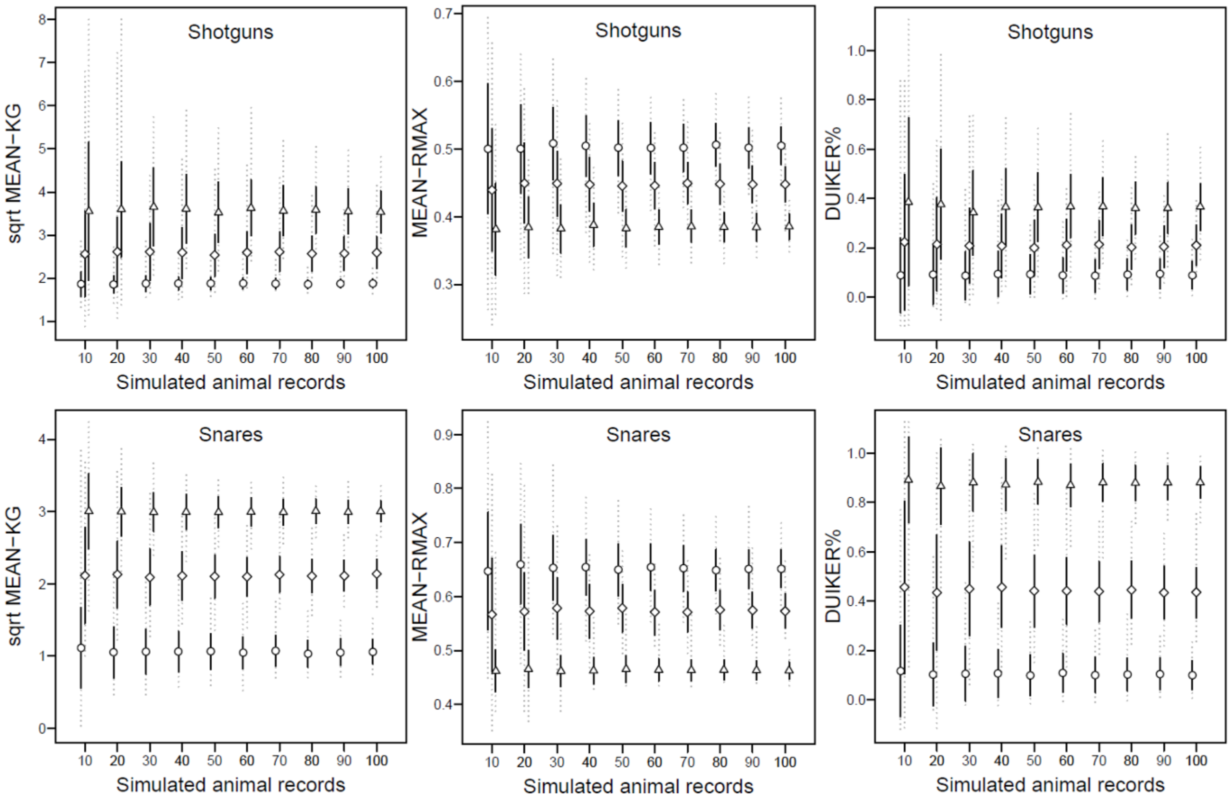

Figure 4. Indicator estimates associated with increasing sample size, in zones under high (✦), intermediate (+), and low (⬬) hunting pressure, showing mean, 95% confidence intervals as solid whiskers, and complete range as dotted whiskers. Where 95% confidence intervals do not overlap, estimates for zones can be considered different from each other. The upper panel show shotguns and the lower panel snares. Points are staggered slightly on the x-axis to prevent overlap.

and indicators performed better with some survey methods than others. In both cases, the likely cause is the selectivity of hunting, which means that only some of the changes that occur in wildlife communities are reflected in hunting offtake. Nevertheless, wildlife indicators represent a viable solution to a number of challenges in wildlife monitoring. The validity of this conclusion rests on the consistency of many separate results across different survey methods, which we summarise here.

First, we tested the reliability of the indicators by comparing estimates for different hunting zones across a hunting pressure gradient. We found that pairings including the MEAN-KG and DUIKER% indicators were almost always correlated, except in cases where the pairing included the MEAN-RMAX indicator, and especially when the comparisons also included estimates from shotgun records.

Next, we compared our indicators to a proxy for hunting pressure. We found that two of our selected indicators changed in the expected manner in wildlife populations subject to increasing hunting pressure, with MEAN-KG falling and DUIKER% increasing as large species are depleted, and smaller species remain stable or even increase, regardless of the survey method used. However, models that used shotgun records explained only around half of the variance explained by those using cameras or snares. In contrast, MEAN-RMAX did not respond predictably to hunting pressure.

When compared to CPUE, we again found MEAN-KG and DUIKER% to be better indicators than MEAN-RMAX, as they predicted the CPUE of individual species more frequently in the snare survey (MEAN-KG = 9, DUIKER% = 6, MEAN-RMAX = 4, out of 9 species). Indicators worked better for snares than cameras. Although we could not test the ability of shotgun records to predict CPUE directly, shotgun indicators did track the CPUE of species >15kg calculated from snare records, but not from camera records.
Finally, we tested the precision of each of the two hunting methods and three indicators. We again found MEAN-KG and DUIKER% to be superior to MEAN-RMAX, and again found that estimates from snares were superior to those from shotguns, in this case being more precise, and requiring a much smaller sample size to detect change.

Therefore, MEAN-KG and DUIKER% appear to be good indicators for monitoring wildlife populations at out study site, because they consistently behaved in a way that was expected, responded to hunting pressure, were correlated with CPUE of many species, and had a high precision. In contrast, MEAN-RMAX was a poor indicator, because it did not respond to hunting pressure, was often not correlated with the CPUE of species, and had a low precision, and this was especially the case with shotgun records. MEAN-KG and DUIKER% were also almost always correlated, whereas MEAN-RMAX was not. This requires explanation, because the *r*max and mass of a species are correlated. Red and blue duikers are the two species used to calculate the DUIKER% indicator, and comprised more than 44% of all records in each survey method. The mass of these species is very different (~15kg and 5kg respectively, or ~2,2, 0.8, and 1.3 standard deviations of the mass in our shotgun, snare, and camera samples), but their *r*max estimates are very similar (~0.44 and 0.49 respectively, or ~0.2 standard deviations in all survey samples). This problem is compounded by the limited range of species caught by hunters, especially when using shotguns, because these species make up the majority of the catch.

This truncation of the mass and *r*max range of animals occurs because hunting is conducted selectively. Hunters avoid the smallest species by choosing not to shoot animals whose market value is less than that of a shotgun cartridge and by setting snares that are too large to catch the smallest animals, e.g. squirrels. The larger range of species caught by snares, including higher numbers of small and large animals, results in a wider range of estimates, a higher precision in indicator estimates, making snares technically superior for wildlife monitoring. In addition, the ease of estimating effort from snare catch records allows for relatively easy direct estimation of CPUE and hence are technically better for estimating wildlife population trends. However, shotgun hunting is generally more compatible with existing wildlife laws in many countries and so may be preferable. A further implication of our findings is that despite being correlated, shotguns and snare catch data provide quite different estimates of indicators, and aggregation of the two methods is likely to confound estimates.

The consistency of these trends across three separate survey methods lends confidence in the validity of these results. Camera trapping enabled us to independently verify the results of the hunting surveys, despite the method being subject to a different set of biases. Our camera survey had better spatial resolution (we knew exactly where the records came from), and recorded small and large species that were scarce or absent in the hunting records. However, unlike the hunting methods, animals could, and likely were, recorded multiple times, particularly species that are territorial and have relatively small territories, such as the red and blue duikers. Despite this, we found correlations between the different survey methods (for MEAN-KG and DUIKER%), a similar response to hunting pressure, and correlation with declines in large mammals, and changes to many individual small and large species.
MEAN-RMAX appeared to be more effective for cameras than the other surveys, because of the range of species caught, but less effective for DUIKER%, perhaps because of multiple recordings of the same species.

Taken together, these results are evidence that some indicators can be useful for wildlife monitoring, by capturing key information about wildlife communities in a single, easy to calculate variable. The advantages of using effective indicators can be added to those already associated with self-monitoring, which include abundant, low-cost data (Rist et al. 2010) and relevance to resource users. Although this paper considers only the wildlife monitoring aspect of self-monitoring schemes, the secondary outcomes of self-monitoring are perhaps even more important. These include the integration of resource users into management (Noss et al. 2004), and potential behaviour (Noss et al. 2005) and policy change at a scale suited to community level management (Danielsen et al. 2010). A monitoring system involving or being directly accountable to resource users is one of Ostrom's design principles for effective natural resource management (Ostrom 1990:94), derived from the study of successful common pool resource systems.

Indicators may also be suitable for the spatial management of hunting systems in tropical forests, where quota systems and closed seasons devised for hunting systems in the global north are inappropriate (Milner-Gulland et al. 2003). Some have made a case for zonal management systems (Mockrin and Redford 2011), in which areas of forest are closed for hunting for wildlife populations to recover through reproduction or migration. The self-monitoring scheme described here seems well suited as a component of such spatial management strategies.

As a wildlife monitoring tool, the main alternatives to hunter self-monitoring are transect surveys and camera trapping studies. However, transect surveys are not well suited for monitoring of many tropical forest game species (Fragoso et al. 2016), particularly at a spatial scale and precision of relevance to hunting management. Use of camera traps, at present, is limited by high cost, the complexity of data processing required, and vulnerability to bias. This is changing, however, with new techniques allowing the use of cameras as point transects (Howe et al. 2017) in some circumstances, and technological advances producing algorithms enabling automated identification of animals. Cameras, or other remote sensing devices, will most likely become the best tool for accurately assessing wildlife populations in the future, if not already, but will require sophisticated data processing and technical expertise. However, self-monitoring schemes will always be relevant for wildlife management in the vast areas of forests outside protected areas where hunting is permitted and management resources are scarce. This includes the majority of tropical forests.

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Chapter 5: Synthesis

This study sought to address questions related to wildlife governance in an industrial forestry concession, and if hunter self-monitoring could assist through behaviour change or producing wildlife population estimates.

Chapter 1. Wildlife governance

In chapter two, we describe the network of actors participating in wildlife governance in an industrial forestry concession in the Republic of Congo. This chapter contributes to a growing literature on natural-resource network governance, but one that has thus far had little overlap with the bushmeat and wildlife literature. Using the methods of network analysis in combination with an historical overview, this chapter brought together information from disparate sources and fields, including: interviews, satellite imagery, organizational reports, policy documents, and scientific literature on wildlife, forestry, sociology and history. This broad focus can contribute to our understanding of the reality “on the ground”, which can often remain hazy to academics in distant labs, whose research focus may be narrower. But for those interested in the conservation of wildlife, this is where ideas ultimately succeed or fail. When we as conservationists try to identify solutions, this is the context in which we do so, and it is one which we are often relatively ignorant of. Studies of this kind can provide academics with insight into wildlife management as it really happens, by describing how things are, and why they came to be.

This chapter focused specifically on a single concession: forestry management unit (FMU) Ngombé. In Congo, a single forestry concession encompasses the many hunting territories of the communities who live there. A forestry concession is itself one of many, each of which is subject to regulations and agreements, both formal and informal, spanning local, national, and international scales. As such, the concession offers a useful intermediate position from which to view many relevant processes. FMU Ngombé saw an increase in the number of actors, types of actors, and a shift in the role of the forestry company. Over this period, wildlife came to be managed far more actively, with the establishment of ecoguard units, who undertake patrols in the forest and control the movement of bushmeat on the roads. These units are a state sanctioned component of a powerful wildlife co-management arrangement, which also includes the forestry company and an international wildlife NGO. Despite this decentralisation of wildlife management to include a broader range of non-state actors, communities themselves do not play an active role in wildlife management. Factors leading to this situation include the criminalisation of hunting in national law, lack of organisation at the village level, and extreme poverty, with hunting being one of the few livelihoods generating immediate cash income.

The most recent development is a sustainable bushmeat project implemented by the FAO, which is concerned with all three of these concerns. It is a wildlife-focused Community Based Natural Resource Management (CBNRM) scheme that seeks to legitimise the bushmeat trade and in the process make it more manageable, to serve as a platform supporting greater community level organisation, and to secure the sustainable harvest of wildlife. CBNRM is an alternative to top-down down management systems, but in the Congo basin, the concept remains largely aspirational, and in need of rigorous field testing. Hunter self-monitoring is a common component of community wildlife management, and is a component of the CBNRM project at FMU Ngombé. This chapter establishes the lack of community involvement in wildlife management, and the current interest in CBNRM, including hunter self-monitoring, as a possible solution. The following two chapters sought to test two aspects of hunter self-monitoring that have been assumed, but not empirically established. The first is its ability to change the behaviour of hunters towards more sustainable hunting practices. The second is in producing estimates of wildlife abundance from raw hunter records that can be used for monitoring wildlife.

Chapter 2. Hunter self-monitoring for promoting sustainable hunting

In chapter in three, we tested the ability of self-monitoring to change harvesting behaviour. To test this directly would is a major challenge, far beyond the scope of a PhD. As a more feasible alternative, we used a behavioural economic experiment in the form of a Common Pool Resource (CPR) game, framed around the bushmeat harvest, using Congolese villagers as the subjects. Groups of five subjects chose the amount of effort they dedicated to hunting in a shared wildlife resource, under three experimental conditions. The resource changed dynamically, and declined if hunting removed more animals than were created through reproduction, and the fewer animals remaining, the more difficult hunting became. In a baseline condition, players could not communicate with each other, and so their only information about the state of the resource and the hunting behaviour of others was their own hunting success or failure. In many CPR experiments, resources are quickly depleted under this condition (Ostrom et al., 1992). In the second condition, subjects could communicate with each other, better representing the real situation of village hunters in Africa, and potentially allowing for information sharing and coordinating hunting behaviour. In most CPR experiments, communication alone allows subjects to reduce the level of resource extraction and to use the resource more effectively (Ostrom et al., 1992). In the third condition, subjects could again communicate but also had a voluntary, visual means of reporting their hunting behaviour to the group. This final condition was meant to simulate a hunter self-monitoring scheme, and provided a formal means for sharing the information that each subject had access to.

In this experimental set-up, voluntary self-monitoring in combination with communication was sufficient to reduce hunting compared to the baseline (no communication) condition, whereas communication alone was not. The implications this research has for real world hunter self-monitoring depends on the extent to which they can be generalised from the experimental to natural settings. Taken at face value, the result suggests that implementing self-monitoring schemes may be a relatively easy way of encouraging hunters to hunt more sustainably. This has been seen in some real-world natural resource systems, in which the act of self-monitoring was enough to catalyse resource users into initiating actions to try to conserve resources (Danielsen et al., 2007). There is also some evidence that behaviour in experimental settings is correlated with behaviour in real-world settings. In one experiment, forest user groups in Ethiopia that contained a larger share of individuals who acted as conditional cooperators[4] in an experimental setting were more successful in real-world forest commons management (Rustagi et al., 2010).

However, the relevance of both of these findings depends on their generalisability to bushmeat harvesting systems in Congo, and there are numerous reasons be cautious. We may not be able to generalise experimental results to the real-world because the salient factors may be missing from the experiment. We may not be able to generalise results from other resource systems, because hunting is a particularly difficult resource to manage due to biophysical factors such as wildlife being difficult to monitor, mobile, living at low densities, with low reproductive rates, as well as socio-economic factors such as lack of alternatives, the criminalisation of hunting, and a scarcity of pre-existing institutions for wildlife management (Inamdar et al., 1999). So, while it may indeed be the case that self-monitoring could motivate hunters to hunt more sustainably, within a finite range of contexts, this does not necessarily follow from an experiment, played over the

[4] Conditional co-operators are willing to cooperate when others are, but will switch to more selfish strategies when others are not cooperative

course of an hour, with specific payoffs and rules, just as findings that self-monitoring in other resource systems can increase locally initiated conservation actions does either. Only long-term implementations in real communities are can satisfactorily answer the question of whether self-monitoring can contribute to wildlife management through behaviour change, but this experiment at least gives additional support to the idea that it could.

Chapter 3. Hunter self-monitoring for monitoring wildlife populations

Chapter five addressed a conceptually much simpler, but still technically difficult challenge in hunter self-monitoring: using raw hunter records to produce useful estimates that can be used for wildlife monitoring. To try overcome this challenge we used three methodological differences compared to previous studies: a survey area containing a relatively large number of hunting territories and zones over a large area, a camera trapping survey as the validation data, and converting animal records to indices. The first allowed us to survey over a much larger gradient of hunting pressure than can be seen over a single village territory, or a few nearby villages sharing similar sizes and histories. The second gave us validation data that consisted of many records of the same species that hunters typically catch in snares or during shotgun hunting at night, at a similar spatial scale to that provided by hunters, as opposed to transect data which people have tried to use in the past (e.g. Rist, 2007). The third allowed us to avoid the difficulties associated with estimating survey effort during hunting, especially when using shotguns, and exploited the tendency of wildlife communities to shift from large to small bodied species under hunting pressure.

Using this method, we were able to find correlations between wildlife indicators derived from hunter and camera trapping methods for two of the indicators we tested, indicating that hunting methods can be used to monitor changes to wildlife communities. The same two indicators responded predictably to hunting pressure and correlated with the abundance of many species, estimated from snare hunting and camera trap surveys. Together, these results suggest that indicators can provide means of producing useful estimates of wildlife community state, of relevance to both hunters and managers. However, we also found that hunting selectivity, especially in shotgun hunting, mean that hunting offtake fails to capture the full magnitude of change, because they underestimate the presence of the largest and smallest species.

Conclusion

In conclusion, this thesis helps to elucidate the current state and history of wildlife governance in forestry concessions, and tests ways that hunter self-monitoring might improve wildlife management. It is apparent that even in one of the best run forestry concessions in the tropics, where there is both a will for wildlife management and financial and technical resources not found elsewhere, communities are still unable to play an active role in wildlife management. In part this is because it is difficult for traditional managers to include villages, and in part because questions remain as to whether it would lead to improved wildlife conservation. These questions are now being tested in the forests of the Congo basin by the FAO and others. The experiments presented here suggest that they are worth testing, and that hunter self-monitoring may be a useful part of any CBNRM, to both promote more sustainable behaviour and to monitor local wildlife populations. Even where CBNRM is not pursued, short-term hunter self-monitoring could add valuable, low cost data on game species that is currently of low quality in the large-scale transect surveys, as part of the large-scale biomonitoring surveys of the kind conducted by international wildlife NGOs across huge swathes of Central Africa's forestry concessions today (e.g. N'Goran, 2017).

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Appendices

Appendix 1. Environmental Uncertainty and Self-monitoring in the Commons: A Common-pool Resource Experiment Framed Around Bushmeat Hunting in the Republic of Congo

Appendix 1A – Game Script

First of all, we would like to thank you for accepting our invitation, and for coming to participate in this experiment.

In this experiment today you can earn a considerable amount of money that you are permitted to keep and take home. You must understand that this is not our private money but given to us by our university for research. If you listen to the following instructions carefully, you can, depending on how you play the game, earn a considerable amount of money. This, therefore, requires you to follow the instructions very attentively. The objective of the experiment is to get data for our research project. It does not have any political or religious objectives. We are interested in your decisions during the game. However, there is no "right" or "wrong" answer or way to play the game. The decisions you make will not be shared with anybody, not even the other players, and we will not take your names so the answers can never be traced back to you.

You will be paid 1500 FCFA for just participating in the experiment (as an appearance fee) plus the additional earnings that you have earned during the game. The game has 10 rounds. You can earn money in each round depending on the number of animals you decide to harvest, the amount of time you decide to spend farming, and how many animals are left at the end of the game. Your earnings will be paid out to you in private so that nobody will know your decisions in the experiment. If you play well you can earn up to 3000 or more in the game, if you play poorly you can earn 1200 or less.

Some important remarks before we can start:

1. The game will take about three hours, including waiting time. If you find that this experiment is something that you do not wish to participate in for any reason, **or you already know that you will not be able to stay for the three hours,** please let us know immediately so that we can replace you with somebody else.

2. You are not allowed to talk during the game (except where permitted i.e. the communication part in rules 2 and 3).

3. It is very important that you understand the game. Therefore, we will check your understanding by asking each of you "test questions" about the procedures of the game. If you do not understand the rules you may always ask the assistants to explain them. **But if you cannot answer the test questions after explaining them again, we will have to exclude you from the experiment.**

4. If you have questions, always raise your hand and wait until the assistant comes to you. Then you can ask your question and the assistant will answer it. You are not allowed to leave the game area without permission.

In this game you are making decisions about the management of a forest from which you and 4 other people can hunt animals. You can earn money by hunting animals, and by farming. Each round is a year, and you can choose to go hunting or farming 12 months a year. At the beginning of each round you will go hunting, and you stop when you have caught enough animals. The rest of the time you will go farming. But the more the group hunts, the smaller the number of animals in the forest becomes and the less easy it will be to catch animals in the remaining rounds. You will be paid 50 CFA for every animal you hunt, and 10 CFA for every time you farm. Hunting is worth more money, but you won't always catch animals. Farming is worth less, but you always get paid for it. In addition to this, you will share all the remaining animals between you.

Here, let me show you how it works….

This is the hunting part of the game. The red marbles are animals, red for blood, the black marbles mean you didn't catch anything, black for an empty pot. When we play the game, the marbles will be hidden in a bag so you can't see which are which. The hunting game will take place in the "forest", over there, one at a time, so nobody else can know what happened. So, imagine you are playing (select one player). These green marbles represent months spent farming, green for the agricultural fields. You can exchange months spent farming to go hunting, so give me one marble and you can go hunting. (allow player to play until he is finished. Then play with other players. If all players are exchanging all 12 green marbles, explain that they have the option of keeping some).

At the end of the turn, more animals will be born, and we add more animals to the forest. The number of new animals added depends on the number that are left in the forest. When there are only a small number of animals, then the number of animals added is small, because there aren't many mothers to have babies. When the number of animals in the forest is very high, there is only a small number of animals added because there isn't a lot of food for the new animals. When there are about half the animals left, the number of new animals added is highest, because there are a lot of mothers and lots of food. So, it's good to keep the number of animals at about half or more. At the start of the turn there were 80 animals out of a possible 100. You can all see that it is very easy to catch animals when there are lots of animals in the forest. As the number decreases, it gets harder to catch animals, which you will see next turn.

As a group, you killed ___ animals. That leaves ___ in the forest. When there are this many animals in the forest, there are this many new animals born (show them, but don't tell them the number). So, after the new animals are added, the number of animals in the forest has fallen by this much (show them the marbles, and the replacement of the red marbles with the black marbles). So now there are only ___ in the forest.

*** Only for rules 3: Now we will play again, but we will add a new rule. You can use these boards and these counters to show what you've done in the forest. Red tokens correspond to red marbles, and black tokens correspond to black marbles. Here, let me show you (show a turn, and then show how you can report it). This is optional, and you can also show something else if you want to. For example, you can go hunting ten times, and say you only went hunting once ***

Now we will play another turn (only two practice rounds are played in all three conditions). To prove that you understand, I want every one of you to spend some time farming this turn, and not go hunting for all twelve months (play another round. If any players go hunting for all 12 months, make sure they understand that it is optional, and that they demonstrate this by retaking their turn).

You can see now that there are fewer animals in the forest, it is more difficult to hunt, and you get more black marbles than you did last time. So, when the number of animals falls, you lose money, because there are less new animals born each year, and you spend more time hunting unsuccessfully when you could be farming instead.

*** Players report what they did in the forest. On the last player, if nobody else has "cheated", tell them to cheat so everybody can see and understand ***

So, if the whole group hunts a little bit instead of a lot, the whole group wins and everybody gets the more money. But players that hunt more than the others get more money than the others, look (show the difference between the highest earning player and the lowest earning player from the last round played). But if everybody hunts a lot, then the number of animals gets low and everybody loses.

At the end of the game you will be paid the amount of money in CFA equal to:

- the amount you earned from hunting
- the amount you earned from farming
- your share from the number of animals in the forest at the end of the game
- the appearance fees

Appendix 1B – The Dictator Game

We played the dictator game with a total of 20 people divided into two sessions. We played one session with a group of 10 men, and one with a group of 10 women. At the beginning of each session we explained the rules of the game and how it would be played, and that they would decide how much of a stake (4000 CFA) they would keep, and how much would go to a randomly selected member of their community. There were four members of the research team present for the instruction (henceforth "researchers"), three Congolese, and one white European. In both sessions the same Congolese researcher explained the rules according to a script, whilst the three remaining researchers remained silent. Once the game had been explained, one Congolese researcher remained with the group to facilitate and to ensure they didn't communicate with each other, whilst the other three researchers went to a nearby building where the participants would play the dictator game (the "game area"). The Congolese researcher who remained with the participants sent them to the game area one at a time, when called.

Although three researchers (one white European, two Congolese) had entered the game area, there were only ever two present when a participant arrived to play the game, with the third exiting another door. One Congolese researcher was always present (always the same researcher throughout both sessions) and administered the game, whilst a second researcher sat silently alongside them, without making eye contact. For half of each session this second researcher was a Congolese researcher, and for half it was the white European. We switched the order in the second session. So, in the first session, the first five participants took their decision in the presence of a Congolese researcher and a white European, and the second five participants in the presence of two Congolese researchers. This order was reversed in the second session. So, for all participants, the only change in the entire set up was the presence of a white researcher or a Congolese researcher during the decision-making part of the game.

When the participants arrived at the game area, they had the rules explained to them again briefly and were then asked to draw from a pile of notes in front of them (the stake). They could keep the money they wanted and place the rest in an empty envelope in front of them and seal it, before leaving the game area. The researchers turned away during the decision-making phase.

Appendix 2. Using wildlife indicators to facilitate wildlife monitoring in hunter-self monitoring schemes

2A. Accuracy of hunter reporting and recording

In order to test the accuracy of the records, we accompanied hunters on 15 hunting trips of different duration, totalling 25 days of hunting. Observers carried a GPS unit, and recorded the route, all animals encountered, type of encounter, and when hunters were actively hunting and when they were not. Returning from the hunting trip, the hunters reported to the village monitor as usual. We then compared the record of the trip produced by the monitor with that produced by the observer, evaluating whether or not an animal encounter was reported and if the details of the encounter were correctly reported. We also compared whether or not the animal was reported in the correct zone, i.e. if the hunter correctly reported the nearest camp. If not, we recorded if the zone reported was adjacent to the correct zone, or further away. We accompanied hunters on 15 hunting expeditions, of one night (n = 8), two nights (n = 4), and three nights duration (n = 3). Overall, hunters reported a low percentage (47%) of animal encounters (including killed, seen, and heard), but a high percentage of animals they had killed (93%. table 1). Hunters seemed to report only the first encounter with a species that they hadn't killed, rather than every encounter. The age and sex of 36 of 42 (86%) killed animals were correctly reported by hunters. Hunters were able to report the correct zone in which they encountered an animal in only about half of the cases (48%, table 2). However, almost all of the incorrectly reported locations were reported as the adjacent zone. The magnitude of the error in terms of km varied as zones differed in size, but the discrepancy was generally between 2 and 4 kilometres. Overall, hunters seemed to report killed animals reliably, and locations somewhat reliably. As a result, we only include killed animals in the subsequent analyses.

Table A1. The rate at which hunters reported animals they had killed or otherwise encountered compared to those reported by an observer accompanying the hunters on a hunting trip.

Encounter type	Reported		Not reported		Total
All encounters	90	(47%)	98	(52%)	188
Killed animals only	41	(93%)	3	(6%)	44
Animals not killed	49	(34%)	95	(65%)	144

Table A2. Hunter accuracy in reporting encounter location in terms of where hunters reported they had encountered an animal and where the animal was actually encountered.

Reported location of animal encounter	N	%
Correct Zone	43	(48%)
Adjacent Zone	44	(49%)
Further	3	(3%)
Total	90	(100%)

2B. Maps

We held participatory mapping sessions, and then walked hunting paths with hunters and a GPS for georeferencing. We then produced simple maps (figure 1), which included the names of hunting camps and a letter ID which could be used by hunters and monitors to record where animals were encountered.

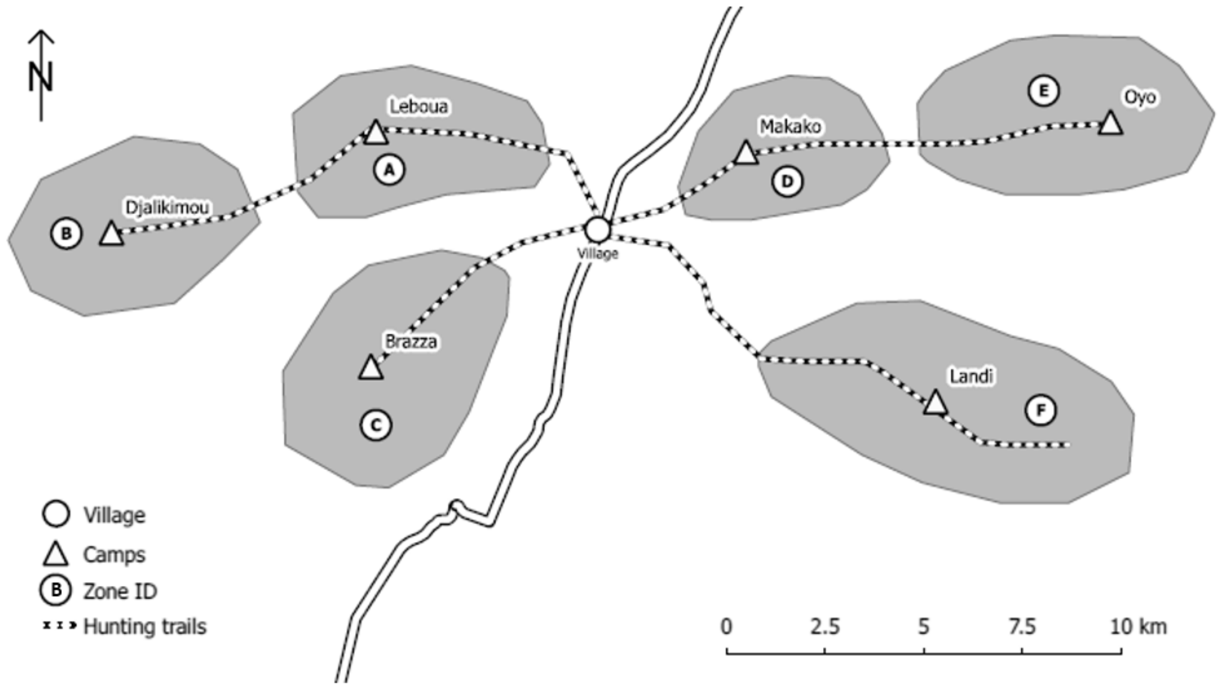

Figure B1. An example of a map provided to monitors, depicting major hunting paths, camps, and both local name and an identification letter used in data recording

2C. Record sheets

Figure C1. Snare record sheet

Village:		ID:	

Date of record	

Days between visits?	

Number of snares	Large	
	Small	

Location	

Animal No.	Species	Bonne (B)/ Pourrir (P)/ Enfuir (E)	sex (M/F)	Adult/ Infant (A/I)	Destination of carcass
1					
2					
3					
4					
5					
6					
7					
8					
9					
10					

Consumed by hunter or family = 1, Sold for consumption in the village = 2, Sold to a restaurant in the village = 3, Sold in the village for resale = 4, Sold on the river towards Ouesso = 5, Sold on the river towards Pokoloa = 6, Sold on the river towards Molanda = 7

Figure C2. Shotgun record sheet. Data collected but not used in analyses presented are shown in red box.

Village:		ID:		Date of recording:	

When did you leave the village?	Date		Hour		When did you return to the village?	Date		Hour	

Where were you on the 1st day?		Did you hunt during the day?	night?	
Where were you on the 2nd day?		Did you hunt during the day?	night?	
Where were you on the 3rd day?		Did you hunt during the day?	night?	
Where were you on the 4th day?		Did you hunt during the day?	night?	

If the trip lasted longer than this, please use a second sheet

	Species	Seen/ Heard/ Killed/ Missed	Location	Day/ Night (D/N)	Adult/ Infant (A/I)	Sex (M/F)	Destination of carcass
1							
2							
3							
4							
5							
6							

Consumed by hunter or family = 1, Sold for consumption in the village = 2, Sold to a restaurant in the village = 3, Sold in the village for resale = 4, Sold on the river towards Ouesso = 5, Sold on the river towards Pokoloa = 6, Sold on the river towards Molanda = 7

2D. KG and RMAX values used

Table D1. Mass and *r*max values used in analyses

Species	*r*max	Citation	Male	Female	Infant	Citation
Aonyx capensis congicus	0.6	Allebone-Webb 2009	22	22	11	Jacques et al., 2009
Atherurus africanus	0.8	Fa et al. 2015	2.831	2.831	1.4155	Fa and Purvis, 2001
Atilax paludinosus	0.63	Fa et al. 2015	3.075	3.075	1.5375	Fa and Purvis, 2000
Caracal aurata	0.46	*	10	10.0	5	Kingdon, 1997
Cephalophus callipygus	0.44	Fa et al. 2015	19.6	21.9	10.95	Kingdon and Hoffmann 2013
Cephalophus dorsalis	0.2	Fa et al. 2015	19	22.2	11.1	Kingdon and Hoffmann 2013
Cephalophus nigrifrons	0.44	Fa et al. 2015	13.8	14	7	Kingdon and Hoffmann 2013
Cephalophus silvicultor	0.43	Fa et al. 2015	70	70	35	Kingdon and Hoffmann 2013
Cercocebus agilis	4.6	Allebone-Webb 2009	8.3	6.5	3.25	Weckerly, 1998
Cercocebus albigena	0.14	Fa et al. 2015	8.3	6.5	3.25	Weckerly, 1998
Cercopithecus ascanius	0.08	van Schaik and Isler, 2012	4.1	2.9	1.45	Bateman, 1984
Cercopithecus neglectus	0.137	van Schaik and Isler, 2012	7	4	2	Weckerly, 1998
Cercopithecus nictitans	0.11	Fa et al. 2015	6.6	4.2	2.1	Weckerly, 1998
Cercopithecus pogonias	0.1	Fa et al. 2015	3.8	3.8	1.9	Fa and Purvis, 1997
Colobus guereza	0.21	Fa et al. 2015	10.7	9	4.5	Weckerly, 1998
Cricetomys emini	0.7	Fa et al. 2015	1.14	1.14	0.57	Fa and Purvis, 1998
Funisciurus spp.	1.06	Fa et al. 2015	0.112	0.112	0.056	Hayssen, 2008 †
Galago sp.	0.968	van Schaik and Isler, 2012	1.51	1.258	0.629	Dixson, 1998
Genetta sp.	0.67	Allebone-Webb 2009	2.5	2.5	1.25	Fa and Purvis, 2002
Gorilla g. gorilla	0.07	Fa et al. 2015	160	93	46.5	Weckerly, 1998
Hyemoschus aquaticus	0.48	Fa et al. 2015	0.7	12	6	Kingdon, 1997
Manis spp.	0.15	Fa et al. 2015	1.5	1.5	0.75	Fa and Purvis, 1997
Mouse spp.	7.07	Brown and Singleton 1999	0.02	0.02	0.01	Brown and Singleton 1999
Nandinia binotata	0.68	Fa et al. 2015	2.95	2.95	1.475	Fa and Purvis, 1999
Osteolaemus tetraspis	0.57	Allebone-Webb 2009	6	6	3	Allebone-Webb 2009
Pan t. troglodytes	0.04	Fa et al. 2015	49	41	20.5	Weckerly, 1998
Panthera pardus	0.31	Fa et al. 2015	55	55	27.5	Fa et al. 2015
Perodicticus potto	0.34	Fa et al. 2015	1.5	1.57	0.785	Dixson, 1998
Philantomba monticola	0.49	Fa et al. 2015	4.8	5.3	2.65	Kingdon and Hoffmann 2013
Potamochoerus porcus	0.7	Fa et al. 2015	67	55	27.5	Leslie and Huffman, 2015
Red duiker spp.	0.44	Fa et al. 2015	19.6	21.9	10.95	Kingdon and Hoffmann 2013
Smutsia gigantea	0.1	Fa et al. 2015	32.5	32.5	16.25	Kingdon, 1997
Thryonomys swinderianus	0.57	Fa et al. 2015	4.75	3.5	1.75	Okorafor et al., 2013
Tragelaphus spekei	0.28	Fa et al. 2015	100	53	26.5	Weckerly, 1998

* extropolated from felid of similar mass
† assumed Funisciurus congicus.

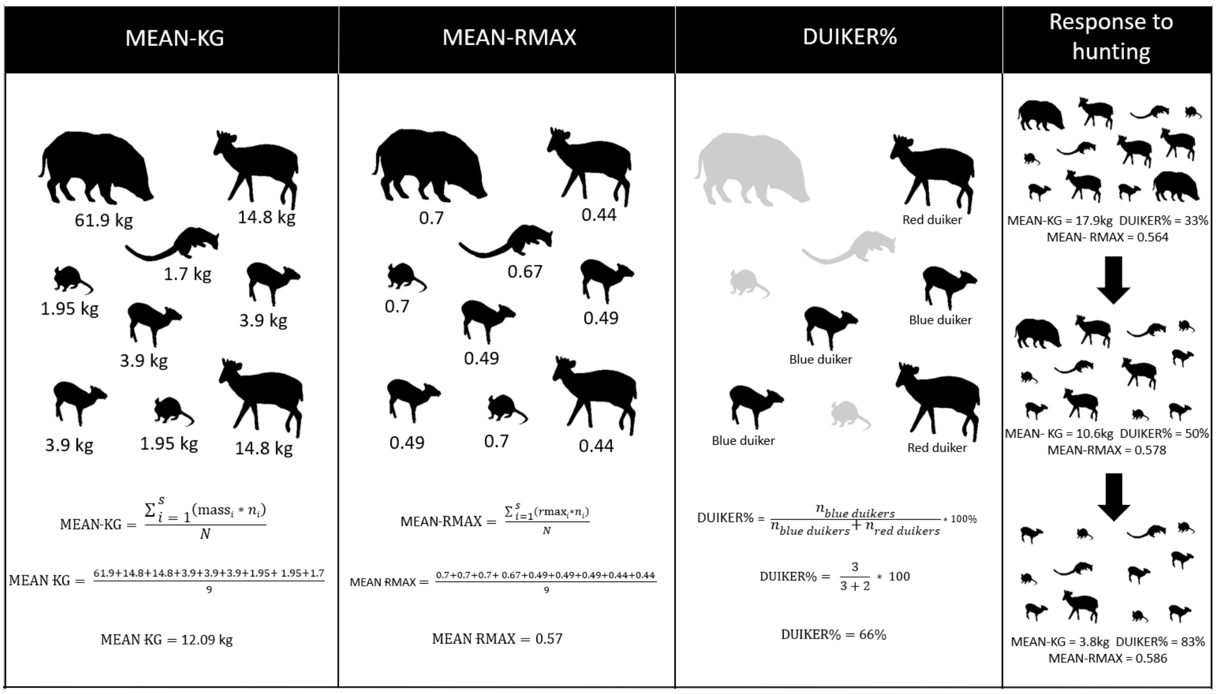

Figure D1. The indicators assessed and their expected response to hunting pressure. Each indicator is estimated for the same sample, and the predicted response of each indicator under increasing hunting pressure is shown in right column. *n* is the number of individuals of a species, *N* is the total number of individuals of all species.

2E. Survey records. Table E1. Records based on each survey method used in analyses, after excluding unidentified species. Percentages given for species are within phylogenetic class, whilst subtotal percentage is of entire sample of survey method

Species	Common name	Shotgun		Snare		Camera	
Carnivores							
Aonyx capensis congicus	Congo clawless otter					5	(5%)
Atilax paludinosus	Marsh mongoose	10	(6%)	38	(45%)	68	(67%)
Caracal aurata	African golden cat					3	(3%)
Genetta sp.	Unidentified genet sp.	7	(4%)	3	(4%)	23	(23%)
Nandinia binotata	African palm civet	139	(89%)	43	(51%)	1	(1%)
Panthera pardus	Leopard					2	(2%)
Sub-total		**156**	**(4%)**	**84**	**(6%)**	**102**	**(7%)**
Pangolins							
Manis spp.	Unidentified pangolin sp.	78	(92%)	49	(98%)	12	(92%)
Smutsia gigantea	Giant pangolin	7	(8%)	1	(2%)	1	(8%)
Sub-total		**85**	**(2%)**	**50**	**(3%)**	**13**	**(1%)**
Primates							
Cercocebus albigena	Grey-cheeked mangabey	4	(11%)	3	(18%)		
Cercopithecus ascanius	Red-tailed monkey	4	(11%)				
Cercopithecus neglectus	De Brazza's monkey					2	(2%)
Cercopithecus nictitans	Greater spot-nosed monkey	8	(23%)	5	(29%)		
Cercopithecus pogonias	Gray's crowned monkey	5	(14%)			2	(2%)
Galago sp.	Unidentified galago sp.					3	(2%)
Gorilla g. gorilla	Western lowland gorilla					41	(33%)
Pan t. troglodytes	Western chimpanzee					76	(61%)
Perodicticus potto	Potto	14	(40%)	9	(53%)	1	(1%)
Sub-total		**35**	**(1%)**	**17**	**(1%)**	**125**	**(8%)**
Reptiles							
Osteolaemus tetraspis	Dwarf crocodile	2	(100%)	3	(100%)		
Sub-total		**2**	**(0%)**	**3**	**(0%)**		
Rodents							
Atherurus africanus	Brush-tailed porcupine	1006	(98%)	290	(56%)	134	(32%)
Cricetomys emini	Forest giant pouched rat	8	(1%)	187	(36%)	188	(44%)
Funisciurus spp.	Unidentified squirrel sp.			29	(6%)	62	(15%)
Mouse spp.	Unidentified mouse sp.					39	(9%)
Thryonomys gregorianus	Lesser cane rat	9	(1%)	11	(2%)		
Sub-total		**1023**	**(25%)**	**517**	**(34%)**	**423**	**(29%)**
Ungulates							
Cephalophus callipygus	Peter's duiker	387	(16%)	343	(68%)		
Cephalophus dorsalis	Bay duiker	83	(4%)	63	(12%)		
Cephalophus nigrifrons	Black-fronted duiker	29	(1%)	17	(3%)		
Red duiker spp.	Red duiker species					266	(32%)
Cephalophus silvicultor	Yellow-backed duiker	68	(3%)	31	(6%)	77	(9%)
Hyemoschus aquaticus	Aquatic chevrotain	11	(0%)	5	(1%)	5	(1%)
Philantomba monticola	Blue duiker	2114	(89%)	358	(71%)	398	(49%)
Potamochoerus porcus	Red river hog	42	(2%)	27	(5%)	60	(7%)
Tragelaphus spekei	Sitatunga	21	(1%)	6	(1%)	14	(2%)
Sub-total		**2755**	**(68%)**	**850**	**(56%)**	**820**	**(55%)**
Grand Total		**4056**	**(100%)**	**1521**	**(100%)**	**1483**	**(100%)**

2F: Regressions between all indicators and survey methods

Table F1. Test statistics of regressions of human population and habitat type on indicator. MEAN-RMAX camera model did not meet model assumptions for heteroskedasticity

		Hunting pressure				Habitat				Model			
Indicator	Method	Estimate	SE	t	p	Estimate	SE	t	p	F	r^2	p	
MEAN-KG	Shotgun	**-0.082**	**0.028**	**-2.931**	**0.005**	**-0.002**	**0.001**	**-2.139**	**0.036**	10.241	0.22	<0.001	***
	Snare	**-0.18**	**0.078**	**-2.319**	**0.025**	**-0.087**	**0.004**	**-2.363**	**0.023**	12.887	0.36	<0.001	***
	Camera	**-0.853**	**0.257**	**-3.317**	**0.004**	-0.012	0.009	-1.379	0.184	11.701	0.55	<0.001	***
DUIKER%	Shotgun	**754.828**	**135.4**	**5.576**	**<0.001**	-1.932	4.93	-0.392	0.696	16.548	0.32	<0.001	***
	Snare	**10.364**	**2.197**	**4.717**	**<0.001**	**0.258**	**0.101**	**2.55**	**0.015**	34.25	0.63	<0.001	***
	Camera	**14.158**	**3.027**	**4.677**	**<0.001**	-0.222	0.102	-2.187	0.041	10.935	0.54	0.001	***
MEAN-RMAX	Shotgun	-0.004	0.003	-1.062	0.292	0	0	0.433	0.666	0.565	0.02	0.571	
	Snare	**0.018**	**0.008**	**2.239**	**0.03**	0.001	0	1.711	0.094	9.209	0.29	<0.001	***
	Camera												

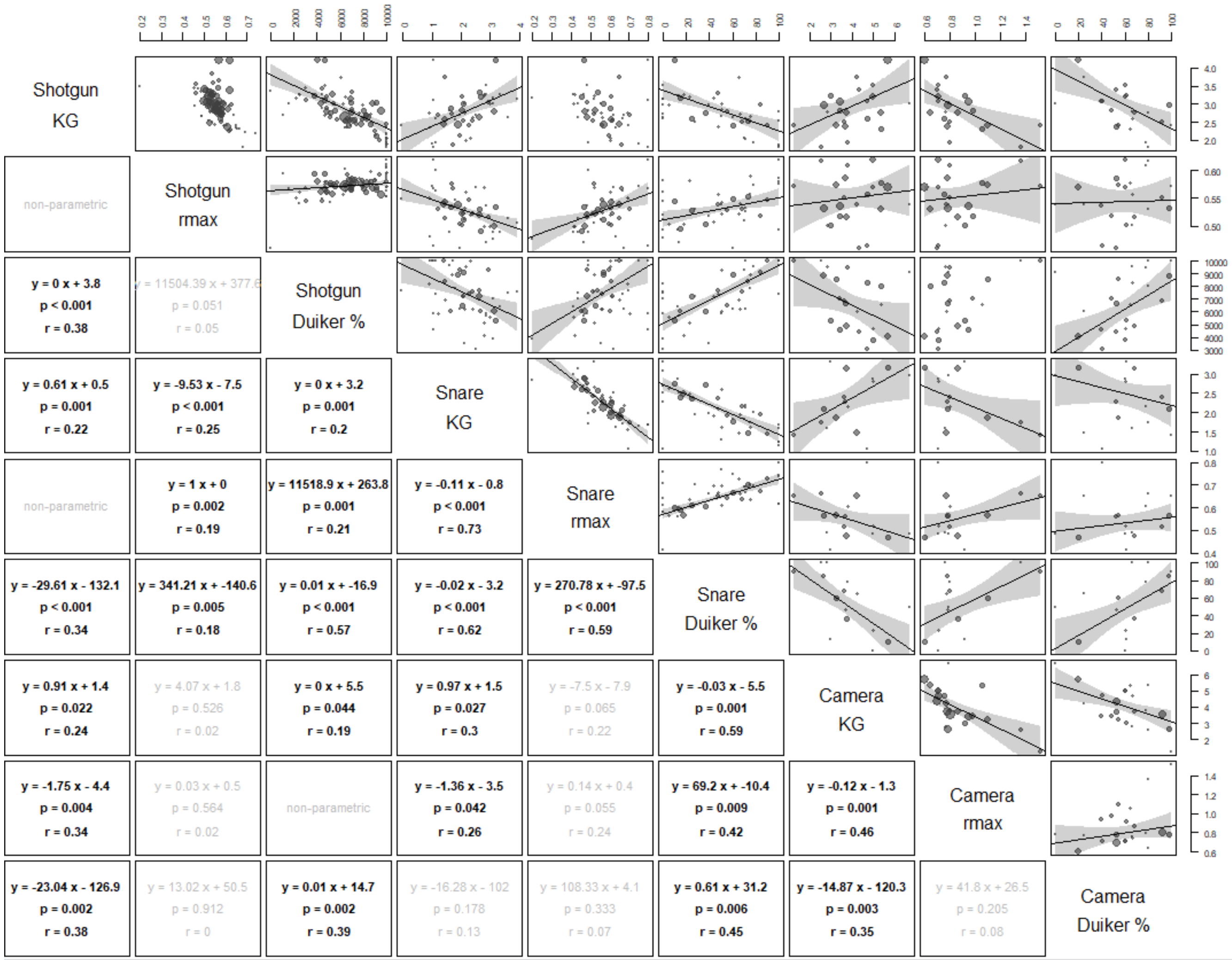

Figure F1. All indicators and survey types regressed against each other. Linear regression fit and test statistics (*p* value and R^2) on the opposite side of diagonal.

2G. CPUE logistic regression plots and statistics tables

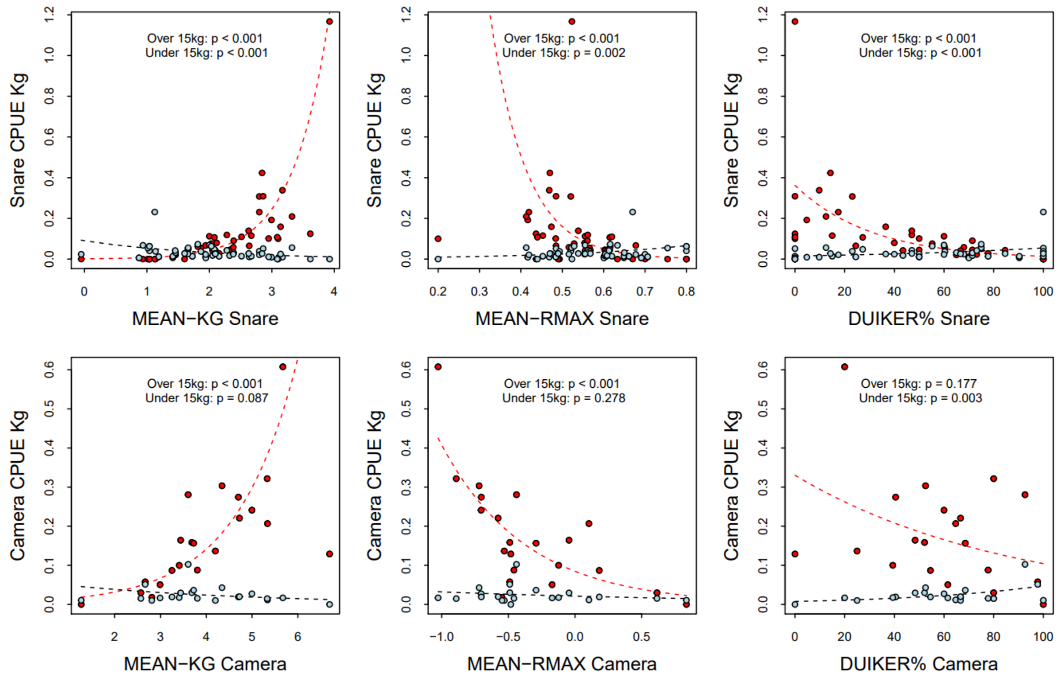

Figure G1. Logistic regression results for comparisons of indicators from each survey method with CPUE estimated from snare and camera data. Species <15kg as blue points and regression line, and species >15kg in red. Indicators generally predict CPUE of species >15kg, with declines expected under increasing hunting pressure. The pattern for species <15kg is much less clear, with indicators predicting no change, or sometimes small increases under increasing hunting pressure.

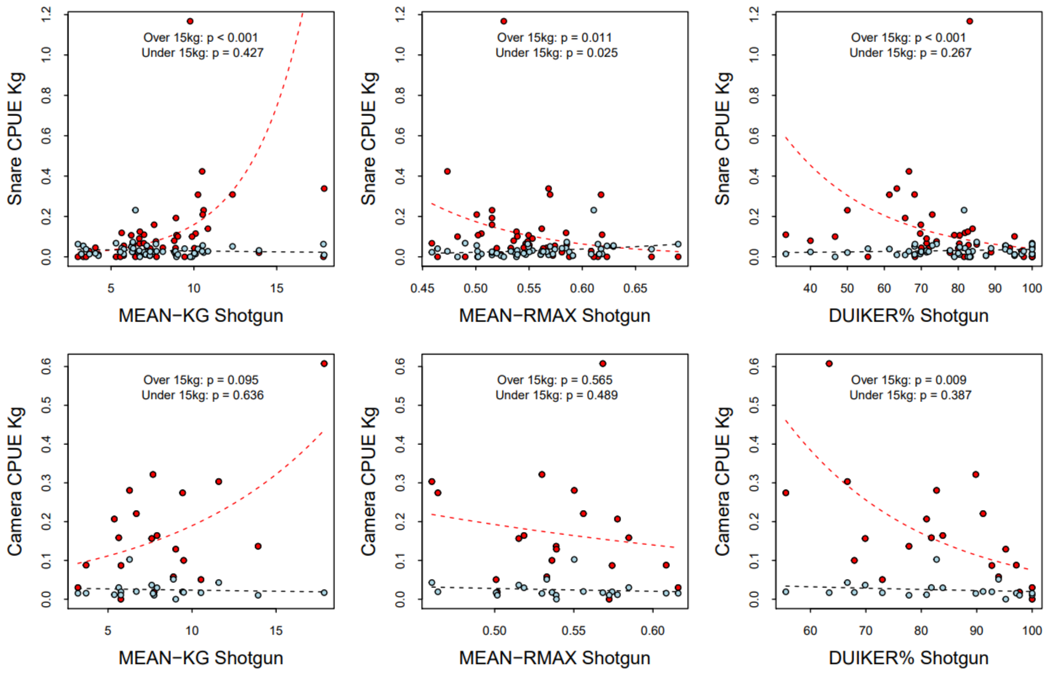

Figure G2. Logistic regression results for comparisons of indicators from each survey method with CPUE estimated from snare and camera data. Species <15kg as blue points and regression line, and species >15kg in red. Indicators generally predict CPUE of species >15kg, with declines expected under increasing hunting pressure, but no changes for species <15kg.

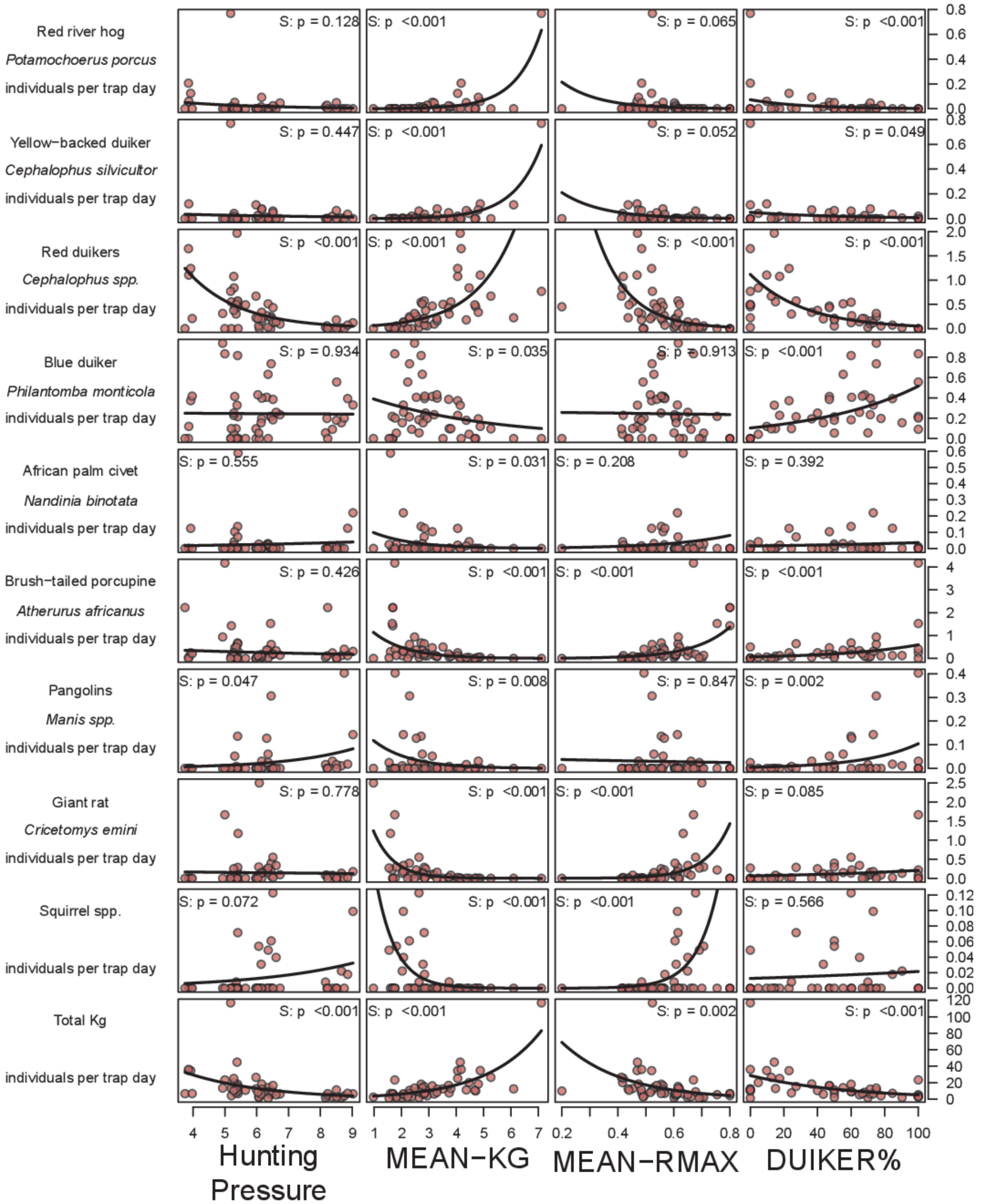

Figure G3. Snare CPUE. Logistic regression models showing species and total Catch Per Unit Effort in snares as a function of human population pressure and wildlife indicators

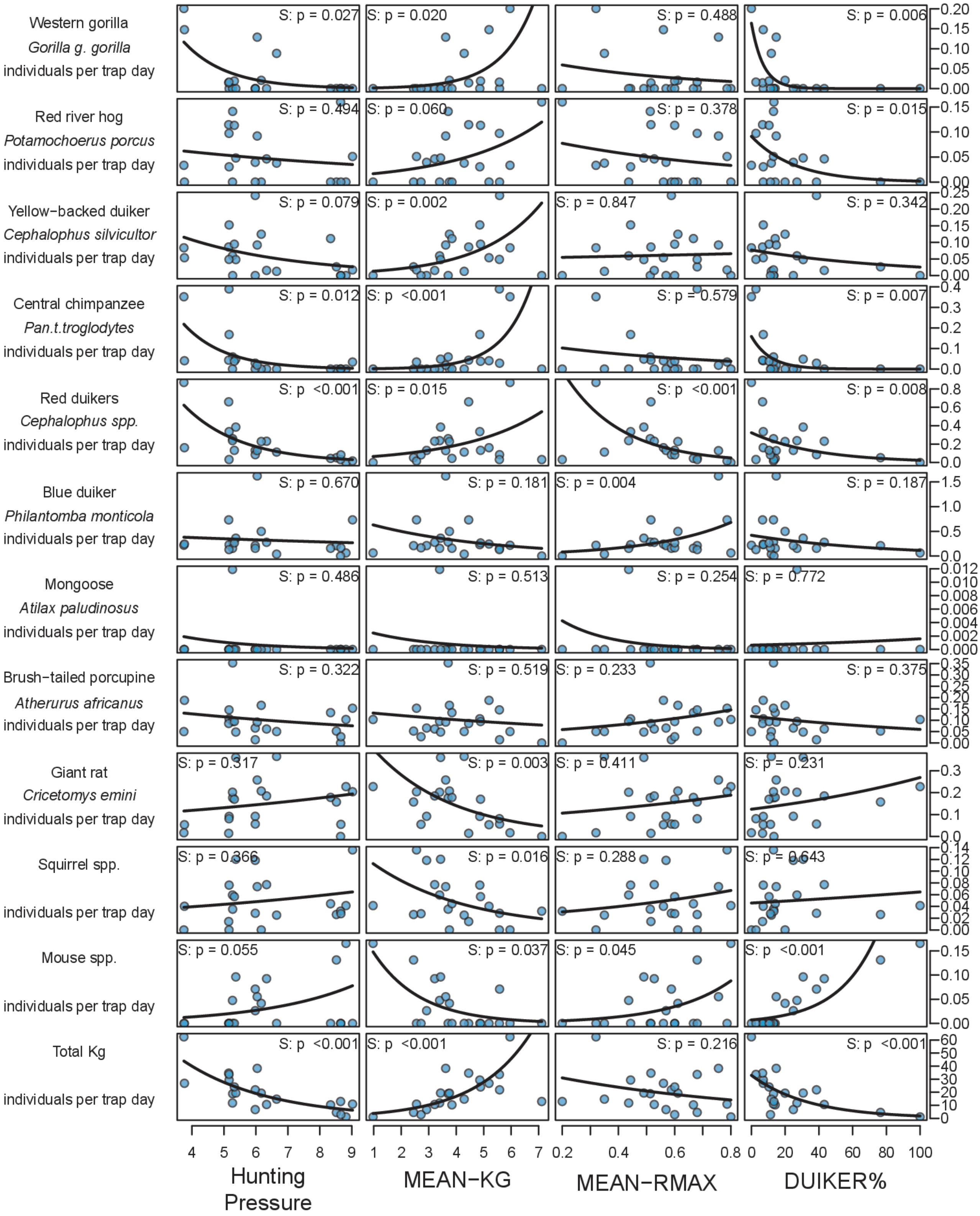

Figure G4. Camera CPUE. Logistic regression models showing species and total Catch Per Unit Effort in cameras as a function of human population pressure and wildlife indicators

Table G1. Logistic regression model results of species and total CPUE against hunting pressure and indicators (snare records). All species with > 25 records shown

	Hunting Pressure				MEAN-KG				MEAN-RMAX				DUIKER%			
Species	Estimate	SE	z	*p*	Estimate	SE	z	*p*	Estimate	SE	z	*P*	Estimate	SE	z	*p*
Potamochoerus.porcus	0	0	-1.18	0.238	**1.052**	**0.131**	**8.019**	**<0.001**	-7.54	4.081	-1.848	0.065	**-0.032**	**0.01**	**-3.323**	**0.001**
Cephalophus.silvicultor	0	0	-0.512	0.609	**0.982**	**0.125**	**7.867**	**<0.001**	-6.951	3.577	-1.943	0.052	**-0.019**	**0.01**	**-1.972**	**0.049**
Red Duikers	**0**	**0**	**-4.415**	**<0.001**	**0.682**	**0.1**	**6.83**	**<0.001**	**-8.66**	**1.325**	**-6.534**	**<0.001**	**-0.031**	**0.004**	**-8.693**	**<0.001**
Philantomba.monticola	0	0	0.005	0.996	**-0.224**	**0.106**	**-2.113**	**0.035**	-0.138	1.271	-0.109	0.913	**0.016**	**0.004**	**4.31**	**<0.001**
Nandinia.binotata	0	0	1.127	0.26	**-0.674**	**0.312**	**-2.162**	**0.031**	4.53	3.595	1.26	0.208	0.009	0.01	0.856	0.392
Atherurus.africanus	0	0	-0.567	0.571	**-0.833**	**0.154**	**-5.394**	**<0.001**	**7.815**	**1.698**	**4.603**	**<0.001**	**0.022**	**0.006**	**3.557**	**<0.001**
Manis.spp.	0	0	1.384	0.166	**-0.769**	**0.29**	**-2.655**	**0.008**	-0.689	3.572	-0.193	0.847	**0.031**	**0.01**	**3.132**	**0.002**
Cricetomys.emini	0	0	-0.889	0.374	**-1.365**	**0.186**	**-7.327**	**<0.001**	**11.722**	**1.917**	**6.116**	**<0.001**	0.012	0.007	1.722	0.085
Funisciurus.spp.	0	0	1.422	0.155	**-1.397**	**0.317**	**-4.41**	**<0.001**	**13.033**	**2.555**	**5.101**	**<0.001**	0.005	0.009	0.574	0.566
Snare Total Mass (kg)	**0**	**0**	**-2.794**	**0.005**	**0.504**	**0.074**	**6.809**	**<0.001**	**-4.604**	**1.453**	**-3.168**	**0.002**	**-0.017**	**0.004**	**-4.554**	**<0.001**

Table G2. Logistic regression model results of species and total CPUE against hunting pressure and indicators (camera records). All species with > 25 records shown

	Hunting Pressure				MEAN-KG				MEAN-RMAX				DUIKER%			
Species	Estimate	SE	z	*p*	Estimate	SE	z	*p*	Estimate	SE	z	*p*	Estimate	SE	z	*p*
Gorilla.g..gorilla	**0.002**	**0**	**3.406**	**0.001**	**0.949**	**0.409**	**2.323**	**0.02**	**-7.33**	**2.656**	**-2.76**	**0.006**	-0.012	0.017	-0.693	0.488
Potamochoerus.porcus	0	0	1.309	0.191	0.364	0.194	1.881	0.06	**-1.991**	**0.822**	**-2.421**	**0.015**	-0.008	0.01	-0.881	0.378
Cephalophus.silvicultor	**0.001**	**0**	**2.151**	**0.031**	**0.521**	**0.168**	**3.104**	**0.002**	-0.569	0.599	-0.95	0.342	0.002	0.009	0.193	0.847
Pan.t..troglodytes	**0.002**	**0**	**4.77**	**<0.001**	**1.081**	**0.255**	**4.235**	**<0.001**	**-4.63**	**1.727**	**-2.681**	**0.007**	-0.01	0.018	-0.555	0.579
Cephalophus.callipygus	**0.001**	**0**	**3.659**	**<0.001**	**0.394**	**0.162**	**2.425**	**0.015**	**-1.327**	**0.5**	**-2.655**	**0.008**	**-0.03**	**0.007**	**-4.296**	**<0.001**
Philantomba.monticola	0	0	1.703	0.089	-0.26	0.194	-1.338	0.181	-0.672	0.509	-1.319	0.187	**0.021**	**0.007**	**2.849**	**0.004**
Atilax.paludinosus	0	0	1.304	0.192	-0.067	0.195	-0.346	0.73	-0.392	0.548	-0.715	0.475	**0.02**	**0.008**	**2.571**	**0.01**
Atherurus.africanus	0	0	0.824	0.41	-0.095	0.147	-0.645	0.519	-0.349	0.393	-0.888	0.375	0.009	0.008	1.192	0.233
Cricetomys.emini	0	0	-2.198	0.028	**-0.392**	**0.133**	**-2.947**	**0.003**	0.396	0.331	1.197	0.231	0.006	0.007	0.822	0.411
Funisciurus.spp.	**0**	**0**	**-2.091**	**0.037**	**-0.328**	**0.137**	**-2.402**	**0.016**	0.177	0.381	0.464	0.643	0.008	0.007	1.063	0.288
Mouse.spp.	**-0.001**	**0**	**-2.428**	**0.015**	**-0.666**	**0.319**	**-2.089**	**0.037**	**2.234**	**0.451**	**4.955**	**<0.001**	**0.028**	**0.014**	**2.001**	**0.045**
CameraTotalMass	**1**	**0**	**60504.029**	**<0.001**	**0.565**	**0.105**	**5.363**	**<0.001**	**-1.497**	**0.244**	**-6.143**	**<0.001**	-0.008	0.006	-1.238	0.216

2H. References

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Appendix 3. Curriculum vitae

SERGIO MARROCOLI

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EDUCATION

PhD Environmental Conservation - Max Planck Institute for Evolutionary Anthropology and iDiv - Leipzig, Germany - 2014 to 2019

Implemented a self-monitoring scheme in eight villages in a forestry concession in the Republic of Congo, in collaboration with rural communities and a forestry company. Worked with over 200 subsistence hunters and eight village monitors to test whether the recording of hunting activity can be used to monitor wildlife and to support more sustainable hunting, using a range of methods including economic experiments, camera trapping, and social network analysis.

MRes Biodiversity and Conservation - University of Leeds – Merit – 2007 to 2008

BSc (hons) Human Ecology - University of Brighton - 2:1 – 2003 to 2006

CONFERENCE TALKS

2017 - iDiv conference, Leipzig, Germany - Using indicators to facilitate wildlife monitoring in hunter-self monitoring schemes

EMPLOYMENT

Consultancy - World Wide Fund for Nature (WWF) - July 2018 to September 2018

Built a tool allowing the production of species population projections for use in project planning and grant proposals. The tool was implemented in R using the Shiny package, allowing users to run statistical models, adjust model parameters, and produce table and chart outputs via a web based GUI.

Field Site Manager - PANAF programme: the cultured chimpanzee, Max Planck Institute for Evolutionary Anthropology - Liberia - 2012 to 2014

Managed one of 40 research sites as part of a continent-wide survey effort. Collected field observations including camera trap videos, wildlife and vegetation surveys, and biological samples, which will be used to determine which behavioural, ecological, social, and genetic factors may have driven the evolution of culture in chimpanzees.

Principle Investigator/Country Coordinator - Frontier Madagascar - 2010 to 2012

Managed a biodiversity research and education project, Including up to 10 research staff and 30 gap-year volunteers at any one time.

Appendix 4. Publication list

First author

- Marrocoli, S. et al. 2018. Environmental Uncertainty and Self-monitoring in the Commons: A Common-pool Resource Experiment Framed Around Bushmeat Hunting in the Republic of Congo. *Ecological Economics*. 149, 274–284. https://doi.org/10.1016/j.ecolecon.2018.03.020

- Marrocoli, S., Nielsen, M.R., Morgan, D., van Loon, T., Kulik, L. and Kühl, H. In review. Using indicators to facilitate wildlife monitoring in hunter-self monitoring schemes. *Ecological Indicators*.

Other peer reviewed articles

- Heinicke, S., et al. 2019. Characteristics of positive deviants in western chimpanzee populations. Front. Ecol. Evol. 7. https://doi.org/10.3389/fevo.2019.00016

- Hoffmann et al. 2017. Persistent anthrax as a major driver of wildlife mortality in a tropical rainforest. *Nature* 548, 82–86. https://doi.org/10.1038/nature23309

- Kalan, A. K., Hohmann, G., Arandjelovic, M., Boesch, C., McCarthy, M., Agbor, A., et al. (in press). Species and individual variation in the novelty response of wild African apes. *Current Biology*.

- Kühl, et al. 2016. Chimpanzee accumulative stone throwing. *Scientific Reports*. 6. https://doi.org/10.1038/srep22219

- Smith, H., Marrocoli, S., Garcia Lozano, A., Basurto, X., 2018. Hunting for common ground between wildlife governance and commons scholarship. *Conservation Biology*. https://doi.org/10.1111/cobi.13200

- Tagg et al. 2018. Nocturnal activity in wild chimpanzees (Pan troglodytes): Evidence for flexible sleeping patterns and insights into human evolution. *Am. J. Phys. Anthropol.* 166, 510–529. https://doi.org/10.1002/ajpa.23478

Appendix 5. Author contributions

Chapter 2. Marrocoli, S., Nielsen, M.R., Morgan, D., and Kühl, H. In preparation. The co-evolution of wildlife and forestry management in tropical forestry concessions: A case study from the Republic of Congo.

Planning: Marrocoli, S. (80%), and Kühl, H. (20%).

Analysis: Marrocoli, S. (100%).

Writing: Marrocoli, S. (75%), Morgan, D. (5%), Nielsen, M.R. (10%), and Kühl, H. (10%).

Chapter 3. Marrocoli, S., Gatiso, T.T., Morgan, D., Nielsen, M.R. and Kühl, H. 2018. Environmental Uncertainty and Self-monitoring in the Commons: A Common-pool Resource Experiment Framed Around Bushmeat Hunting in the Republic of Congo. *Ecological Economics*. 149, 274–284. https://doi.org/10.1016/j.ecolecon.2018.03.020

Planning: Marrocoli, S. (50%), Gatiso, T.T. (20%), Morgan, D. (10%), Kühl, H. (20%).

Analysis: Marrocoli, S. (80%), Gatiso, T.T. (10%), Kühl, H. (10%).

Writing: Marrocoli, S. (70%), Nielsen, M.R. (10%), Gatiso, T.T. (10%), Morgan, D. (5%), Kühl, H. (5%).

Planning: Marrocoli, S. (80%), and Kühl, H. (20%).

Analysis: Marrocoli, S. (100%).

Writing: Marrocoli, S. (75%), Morgan, D. (5%), Nielsen, M.R. (10%), and Kühl, H. (10%).

Chapter 4. Marrocoli, S., Nielsen, M.R., Morgan, D., van Loon, T., Kulik, L. and Kühl, H. In review. Using indicators to facilitate wildlife monitoring in hunter-self monitoring schemes. *Ecological Indicators*.

Planning: Marrocoli, S. (50%), Morgan, D. (10%), van Loon, T. (10%), and Kühl, H (30%).

Analysis: Marrocoli, S. (80%), Kulik, L. (10%), and Kühl, H. (10%).

Writing: Marrocoli, S. (70%), Morgan, D. (5%), Nielsen, M.R. (10%), van Loon, T. (5%), and Kühl, H. (10%).

Appendix 6. Eigenständigkeitserklärung

Hiermit erkläre ich, dass die Arbeit mit dem Titel "Hunter self-monitoring and wildlife governance in an industrial forestry concession in the Republic of Congo: context, behaviour change, and wildlife monitoring" bisher weder bei der Naturwissenschaftlichen Fakultät I Biowissenschaften der Martin-Luther-Universität Halle-Wittenberg noch einer anderen wissenschaftlichen Einrichtung zum Zweck der Promotion vorgelegt wurde. Ferner erkläre ich, dass ich die vorliegende Arbeit selbstständig und ohne fremde Hilfe verfasst sowie keine anderen als die angegebenen Quellen und Hilfsmittel benutzt habe. Die den Werken wörtlich oder inhaltlich entnommenen Stellen wurden als solche von mir kenntlich gemacht. Ich erkläre weiterhin, dass ich mich bisher noch nie um einen Doktorgrad beworben habe.

Newtown (Powys, UK), den__29/06/2020____________

Sergio Marrocoli