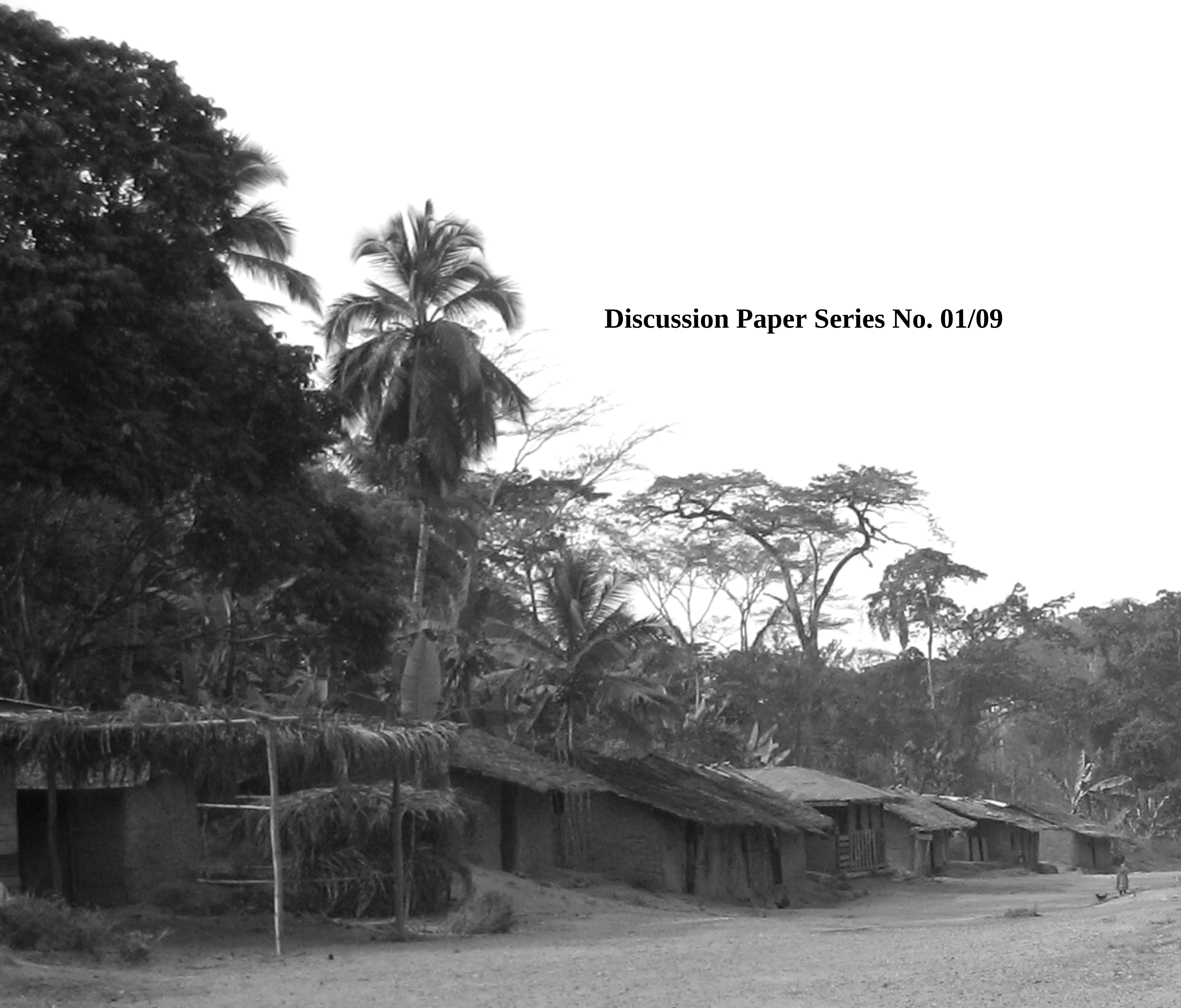
DISCUSSION PAPER SERIES

Managing Forest Wildlife for Human Livelihoods in the Korup-Oban Hills region, West-Central Africa

Deciding on Wildlife Monitoring Schemes Used in Community Based Wildlife Management Models

Janine Kunz



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Address of the author(s):
Janine Kunz
Georg-August-Universität Göttingen
Centre for Nature Conservation
Department of Conservation Biology
Von-Siebold-Str. 2
37075, Göttingen, Germany

e-mail: summsenine@yahoo.de

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Editors:
Matthias Waltert and Christos Astaras, Centre for Nature Conservation,
Georg-August-Univeristät Göttingen, Germany

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Series coordinator:
Matthias Waltert
e-mail: mwalter@gwdg.de

Deciding on Wildlife Monitoring Schemes Used in Community Based Wildlife Management Models

Janine Kunz
Study program: Master of International Nature Conervation (M.I.N.C.)
Department of Conservation Biology
Centre for Nature Conservation
Georg-August-Universität Göttingen

Abstract: Community based wildlife management models require the input of up to date information on a series of parameters that either directly or indirectly reflect the status of local wildlife. This paper briefly introduces parameters that affect sustainable hunting in Central and West Africa, and which would therefore need to be incorporated in monitoring schemes, and presents methods that have been in the past used to monitor wildlife densities at hunted and not-hunted sites. Finally, a small list of recommendations is presented for methods suitable for community based wildlife management projects.

Keywords: line transects, distance sampling, recce walks, wildlife monitoring

1. Introduction

An essential aspect of any wildlife management system, whether administered by local communities or not, is the ability to access whether current exploitation rates by humans are biologically sustainable for wildlife species in the long run. The ability of project managers to make the appropriate decisions depends on the establishment of a monitoring system that will record changes of important parameters over time (such as wildlife harvest rates and wildlife population growth rates). However intuitive this may be, monitoring programs have suffered by discontinuity, diverse methodologies which warrant data incomparable along sites, or where simply absent at the early stages of projects.

This discussion paper attempts in the format of a literature review paper to introduce important factors that wildlife monitoring programs should take into consideration, and attempts to make some recommendations on which monitoring schemes would be appropriate for a community-based wildlife management scheme.

2. Factors Affecting Hunting Sustainability

2.1. Impact of hunting on wildlife populations

Exploitation of animal populations has been identified as one of the main reasons why species are currently threatened with extinction (Mace & Reynolds, 2001). Robinson et al. (1999) reported that exploitation of bushmeat by tropical forest dwellers has increased in recent years, due to growing human populations, greater access to undisturbed forests, changes in hunting technology, scarcity of alternative protein sources, and the fact that bushmeat is often a preferred food. Increase in wild meat consumption occurs at an alarming rate throughout the humid tropics (Robinson & Bodmer, 1999; Fa, Peres & Meeuwig, 2002), affecting particularly mammal populations either hunted for subsistence or commercial purposes (Robinson et al., 1999; Robinson & Bodmer, 1999; Robinson & Bennett, 2000). The pressure is such, that nowadays approximately one third of mammal and bird species worldwide are threatened by overexploitation (Baillie et al., 2004). Local extinctions of hunted species are widespread, with West and Central Africa being especially hard hit. The recent extinction of Miss Waldrons's red colobus (*Procolobus badius waldroni*), a primate subspecies endemic to West Africa, was attributed to bushmeat hunting (Oates et al., 2000).

According to Robinson and Bennett (1999), hunting impacts wildlife populations by:

1) Lowering population densities of hunted species. Redford (1992) estimated that mammal populations are reduced by 70% and 95% under light and heavy hunting respectively.

2) Reducing average body size of hunted species by selecting for large-bodied animals (Bennett 1999).

3) Lowering the mean age of first reproduction of individuals.

4) Increasing, at least theoretically, the mean female fecundity (Caughley, 1977).

5) Altering population demographics, typically by reducing the proportion of animals in older age classes (Robinson, 1999).

6) Decreasing total future reproduction of hunted populations. (Despite potentially lowering the age of sexual maturity and increasing female fecundity, densities of tropical forest species can still decrease sharply due to hunting, as the ratio of breeding individuals to the total population decreases(Robinson and Redford 1991)).

7) Extirpating vulnerable species, such as species with a large-body and low intrinsic population growth rates (Bennett et al., 2007).

8) Changing the biological community of a forest, by removing (totally or ecologically) large-bodied species which are preferred by hunters (Peres, 2000). In addition, hunting can alter the guild and trophic level composition of the biological community, with potentially wide repercussions to the forest structure. Patterns of pollination, seed dispersal, and seed predation can be affected (Redford, 1992). Furthermore, predator numbers can be reduced, as the species they prey on decline.

9) Decreasing the forest’s annual biomass production, by eliminating large-bodie and numerous species (Puri, 1992).

2.2. Human landscape factors affecting sustainability

The spatial distribution of human land uses across the landscape can influence the extent to which wildlife populations are buffered from hunting pressure. For instance, proximity of a hunting area to a protected area (or another area with high wildlife densities which can serve as a “source”) can increase the level of sustainable yield of the area (Bodmer, 1995). On the contrary, increased accessibility of a hunting ground to local population (i.e. via logging roads, paths, etc.) is more likely to lead to unsustainable wildlife harvest levels. Similarly, proximity of hunted areas to market and other commercial centres is prone to increase profitability of hunting and therefore increase the pressure on the wildlife.

2.3. Biological factors affecting hunting sustainability

The level of bushmeat that can be sustainably harvested each year from a certain area depends on growth rate of targeted species and their vulnerability or resilience to human activities in general and hunting specifically.

Biological production

Forests in general sustain less wildlife biomass per km^2, when compared to more productive ecosystems like grasslands (Robinson and Bennett, 1999), and as such hunting in forests is likely easier to become unsustainable. Even within rainforests, there is considerable variation in wildlife densities and biomass production rates.

Hunting usually lowers population densities to levels of less than Maximum Productivity. Population densities of 65% to 90% of carrying capacity (K) have been suggested as maximal for productivity (Robinson and Redford, 1991), yet in tropical forests even “light” hunting reduces populations, on average, to about 30% of carrying capacity (Redford, 1992).

Vulnerability or resilience to harvest

Species with low intrinsic rates of population growth (R_{max}) are less resilient to harvest. For instance, primates and carnivores tend to have low rates compared to their body mass, whereas ungulates and rodents tend to have higher growth rates (Bodmer, 1995). Another category of vulnerable species are those whose nesting, social or anti-predator behaviour makes them easy to harvest. Hunting pressure can cause behavioural changes in certain species (Mitchell and Tilson, 1986), such as changing their activity patterns or their vocalization frequencies. Species that are naturally occurring in low population densities tend, in general, to be more resilient to hunting, as are species that are able to recolonize available habitat. Species that can adapt to disturbed habitats, such as secondary forests and agricultural settings, are less vulnerable than habitat specialists. Although being in closer proximity to humans increases mortality, this seems frequently to be offset by the greater productivity of the species in these habitats (Wilkie, 1999).

The first step needed to make wildlife exploitation sustainable is to evaluate the sustainability of current levels of harvest. Many researchers have carried out assessments of the sustainability of bushmeat hunting, with a particular emphasis on tropical forest mammals. The paucity of available biological data and the difficulty of collecting the data required for a full sustainability assessment is a major challenge for such analysis (Milner-Gulland et al., 2001). There is an urgent need to identify geographical regions where hunting pressure is most acute, identify bushmeat species that are of conservation concern and identify management activities that will be effective under a community based wildlife management model.

3. Composition of bushmeat

Although the composition of bushmeat off take can vary depending on the local fauna, hunting technology used, and local preferences, there are certain patterns that are almost universal across West and Central Africa. According to a large scale bushmeat survey undertaken by Wilkie et al. (1999) in the Congo Basin (Table 1), duikers (*Cephalophus spp.*), bushpigs *(*i.e. *Potamochoerus porcus),* primates and rodents form the majority of the bushmeat recorded in the markets. Duikers accounting both for most biomass and total number of carcasses than any other animal group.

Table 1: Composition of cushmeat harveste in the Congo Basin; adapted from Wilkie et al., 1999

Location	Ungulates %[a]	Primates %	Rodents %	Other %
Ituri forest, DRC	60–95	5–40	1	1
Makokou, Gabon	58	19	14	9
Diba, Congo	70	17	9	4
Ekom, Cameroon	85	4	6	5
Brazzaville, Congo	76	8	6	10
Ouesso, Congo	57	34	5	4
Ndoki and Ngatongo, Congo	81–87	11-16	2-3	-
Dzanga-Sangha, CAR	77–86	0	11-12	2-12
Libreville, Port Gentil, Oyem, and Makokou, Gabon	34–61	20-45	5-27	3-12
Bioko and Rio Muni, Equatorial Guinea	36–43	23-25	31-37	2-4
Dja, Cameroon	88	3	5	4
Ekom, Cameroon	87	1	6	6
Oleme, Congo	62	38	-	-

[a]*primarily the duikers (Cephalophinae) and bushpigs (Potamochoerus porcus)*

Table 2 lists all common bushmeat species in Nigeria and Cameroon, and shows their IUCN Red list status (2007). The status of each species should be a criterion used to select resilient enough species which could be included in a community based wildlife management model. However, the current status of most targeted species listed in Table 2 are under some level of threat of extinction, making any level of exploitation potentially dangerous for their survival.

Table 2: Common bushmeat species of Nigeria and Cameroon and their conservation status

Group	Common name	Scientific name	Red List category and criteria*
Amphibians			
Frogs	African hairy frog	*Trichobatrachus robustus*	LC[1]
Birds	Crested guineafowl	*Guttera pucherani*	LC[1]
	African grey parrot	*Psittacus erithacus*	NT[1]
	Palm nut vulture	*Gypohierax angolensis*	LC[1]
Mammals			
Bats	Strawcoloured fruit bat	*Eidolon helvum*	LC[1]
Carnivores	Leopard	*Panthera pardus*	LC[1]
Primates	Crowned monkey	*Cercopithecus pogonias*	LR/lc[2]
	Mona monkey	*Cercopithecus mona*	LR/lc[2]
	Preuss's monkey	*Cercopithecus preussi*	EN A1cd+2cd[2]
	Putty-nosed monkey	*Cercopithecus nictitans ludio*	-
	Red-capped mangabey	*Cercocebus torquatus*	LR/nt[2]
	Red-eared monkey	*Cercopithecus erythrotis*	VU A1cd+2cd[2]
	Black colobus	*Colobus satanas*	VU A1cd+2cd[2]
	Cross river gorilla	*Gorilla gorilla*	CR A4cd[1]
	Drill	*Mandrillus leucophaeus*	EN A1acd+2cd[2]
	Chimpanzee	*Pan troglodytes vellerosus*	EN A4cd[2]
	Preuss's red colobus	*Procolobus pennantii preussi*	EN B1+2ac[1]
Proboscids	African elephant	*Loxodonta africana*	VU A2a[1]
Rodents	Brush-tailed porcupine	*Atherurus africanus*	LC[1]
	Giant pouched rat	*Cricetomys emini*	LC[1]
	Savannah cane-rat	*Thryonomys swinderianus*	LC[1]
Manatees	West African manatee	*Trichechus senegalensis*	VU A3cd; C1[1]
Ungulates	Peter's duiker	*Cephalophus callipygus*	LR/nt[2]
	Bay duiker	*Cephalophus dorsalis*	LR/nt[2]

Group	Common name	Scientific name	Red List category and criteria*
Ungulates	Blue duiker	*Cephalophus monticola*	LR/lc²
	Black-fronted duiker	*Cephalophus nigrifrons*	-
	White-bellied duiker	*Cephalophus leucogaster*	LR/nt²
	Yellow backed duiker	*Cephalophus sylvicultor*	-
	Jentink's duiker	*Cephalophus jentinki*	VU A1c, C1²
	zebra duiker	*Cephalophus zebra*	VU A1c, C1²
	Maxwell's duiker	*Cephalpphus maxwelli*	-
	Black duiker	*Cephalophus niger*	LR/nt²
	Ogilby's duiker	*Cephalophus ogilbyi*	LR/nt²
	Giant forest hog	*Hylochoerus meinertzhageni*	LR/lc²
	Water chevrotain	*Hyemoschus aquaticus*	DD²
	Red river hog	*Potamochoerus porcus*	LR/lc²
	African buffalo	*Syncerus caffer*	LR/cd²
Reptiles			
Crocodylids	Crocodiles	*Crocodylus niloticus*	LR/lc²
Testudines	Forest hinged tortoise	*Kinixys erosa*	DD²

CR = Critically endangered, EN = Endangered, VU = Vulnerable, LR = Low risk; DD = Data deficient
Source: *IUCN Red Lista Databook (IUCN, 2007; www.iucnredlist.org)

3.1 Ungulates

Duikers are the most frequently hunted group of animals across the region, accounting in some localities up to 95% of biomass. As such, they are an obvious candidate group for wildlife management regimes in community managed hunting grounds (Waltert et al., 2006).

Newing (2001) summarized our current knowledge of duiker ecology throughout the forest regions of Central and West Africa. The Central African duiker community consists of six species.

- Blue duiker (*Cephalophus monticola*)
- Black-fronted duiker (*C. nigrifrons*)
- White-bellied duiker (*C.leucogaster*)
- Peter's duiker (*C. callipygus*)
- Bay duiker (*C. dorsalis*)
- Yellow-backed duiker (*C. sylvicultor*)

The Water chevrotain (*Hyemoschus aquaticus*) is an additional small ungulate species (but is not closely related to duikers). The West African duiker community constists of two species endemic regions extending as far west as Coted'Ivoire (Jentink's duiker *C. jentinki* and the zebra duiker *C. zebra*), three species thought to be restricted to West Africa (Maxwell's duiker *C. maxwelli*, black duiker *C. niger* and Ogilby`s duiker *C. ogilbyi*) and only two species which also occur in Central Africa (bay duiker *C. dorsalis* and yellow-backed duiker *C. sylvicultor*).

Several recent studies of *Cephalophus spp*. have taken the critical next step of measuring not only how much bushmeat is harvested but also estimating the catchment area utilised by hunters. This permitted to calculate the range of harvest rates that exist across the region. Table 3 summarises harvest rates in several sites across the Congo Basin for duikers (Wilkie et al., 1999).

Table 3: Duiker harvest rates across the Congo basin

Site	Range km²	Blue duikers[a] kg/km²yr	Red duikers[b] kg/km²yr	All[c] kg/km²yr
Cameroon–village zone[1]	37	16	62	81
Cameroon–forest zone[1]	270	4	68	74
Cameroon–Dja[6]	600	8	100	114
Cameroon–Lobéké[7]	3,113	18	56	74
Cameroon–Korup[8]	-	-	-	217
Congo–Diba[2]	55	14	141	162
Congo–Oleme[2]	81	15	39	56
CAR–Dzanga-Sangha[3a]	1	22	93	115
CAR–Dzanga-Sangha[3b]	110	67	32	99
DRC–Ituri[4]	12,899	-	-	75
Equatorial Guinea–Bioko[5]	-	2	30	32
Gabon–northeast[9]	-	-	-	75-1390

Source: [1]*(Dethier, 1995);* [2]*(Gally and Jeanmart, 1996);* [3]*(Noss, 1995) aSnares and guns, bnets;* [4]*(Wilkie et al. 1998b);* [5]*(Fa et al. 1995) Catchment area was not reported. Primates provided the highest % of hunter captures;* [6]*(Ngnegueu and Fotso, 1996) Extrapolated from 11 of 30 hunters monitored over 5 of 12 months.; 7(WCS, 1996);* [8]*(Infield, 1988);* [9]*(Feer, 1993)*

[a]*Blue duikers = Cephalophus monticola* [b]*Red duikers = Cephalophus callipygus, C. dorsalis, C. leucogaster and C. nigrifrons*

[c]*Includes C. sylvicultur*

Regardless of which method was used to assess duiker productivity, comparison of average harvest rates (97 kg/km²/year) with average production rates (170 kg/km²/year) suggests that duikers are being overharvested across much of the Congo Basin-assuming that, Robinson and Redford suggest (1994), relatively short-lived animals should not be harvested at a rate that exceeds 40% of annual production (i.e. 68 kg/km²/year) (Wilkie et al. 1999).

3.2 Primates

Primates are the second most hunted group accounting up to 45% of total bushmeat biomass (Wilkie et al., 1999). Given the extended life histories of primates, low reproduction rates and large age of first reproduction, they are especially vulnerable to hunting. Waltert et al. (2002) suggested that sharp population declines and possibly local extinctions of primate species have occurred in the periphery of Korup National Park, Southwest Cameroon. Large-bodied, terrestrial species, such as the endangered drill (*Mandrillus leucophaeus*) are especially vulnerable. Decreases of primate densities were lower over the study period in logged areas, when compared to unlogged areas, confounding our ability to separate the impact of hunting from that of logging activities on primate numbers. However. Considering reports from other logging sites (i.e. Struhsaker, 1997), where primate densities of certain species increased in selectively logged sites, the population declines around Korup National Park are likely attributable to high hunting pressure.

3.3 Rodents

Rodents rank a close third in the list of most common bushmeat species in the Congo basin, accounting for up to 37% of total bushmeat biomass (Wilkie et al., 1999). The savannah cane-rat (*Thryonomys swinderianus*) and the brush-tailed porcupine (*Atherurus africanus*) are two of the most commonr bushmeat species in Nigeria (Jori et al., 1998). The brush-tailed porcupine is a hystricomorph rodent, which frequents the forests of West and Central Africa and is favourite source of meat for rural and urban populations

3.4 Others

Although there are certainly some group of animals which are preferred by hunters, in general hunting in Central and West Africa tends to be catholic in the diversity of species occasionally taken (anything large enough to fetch more than the cost of a cartridge in the market is taken, or large enough to trigger snares). For instance, people of the Takamanda Forest Reserve in Southwest Cameroon collect several reptile species for food, often as by-catch in fishing nets or on fishing lines, or while farming or when encountered along forest trails. Although such groups, like reptiles, may not be the preferred quarry, nevertheless the harvest rates could be higher than the growth rate of the local populations. For instance, tortoises (i.e. *Kinixys erosa*) in southwest Cameroon could be experiencing harvest rates as high as 0.7 *Kinixys* per km^2 (Lawson, 2001).

4. How to estimate sustainable bushmeat offtake?

4.1 Estimation of population density

In order to assess wildlife population densities and their change over time, different challenging monitoring techniques have been used by conservation biologists in dense rainforests. Monitoring wildlife density changes over time are crucial for identifying important areas for conservation of vulnerable species, and for evaluating the effectiveness of wildlife management systems. The following paragraphs provide a quick historical overview of wildlife monitoring techniques. In order to interpret data collected from monitoring programs, information on environmental variables is also needed.

Historical overview of wildlife monitoring techniques:

Duiker densities have been historically estimated with a variety of direct and indirect counts. For instance, Dubost (1980), Hart (1985) and Koster and Hart (1988) undertook census of duiker populations using nets. Some net-counts relied on the hunters actively trying to encircle duikers within a survey area, and at other times relied only on counting duikers caught at randomly placed nets. The first technique has the disadvantage of not being random. The search area is determined by the path length (the diameter of the net circle, the number of times the nets are set, and an estimated 200 m between sets) and the path width (two times the diameter of the net circle). The total count includes all captured duikers plus those that escaped, and assumes that all animals within the search area were detected (which is realistic if experienced hunters are used).

Feer (1988) estimated duiker densities by measuring the home ranges/territories of radio-collared animals and by conducting night counts along transect lines.

Fitzgibbon et al. (1995) used line transects to determine the relative abundance of prey in the three main habitat types of the Arabuko-Sokoko Forest, Kenya, and to compare abundance between areas with high and low trapping intensity. They focused on a few key groups, namely primates (*Papio* spp. and *Cercopithecus* spp.), duikers (*Cephalophus* spp.), bushpigs (*Potamochoerus porcus*), elephant shrews (*Rhynchocyon* and *Petrodomus* spp.) and squirrels (*Heliosciurius* and *Funisciurus spp.)*. Densities were estimated either directly from sightings or indirectly from nests, dung piles or feeding signs. Transects were positioned randomly in highly hunted and non-hunted areas. The abundance of animals sign was also recorded for each transect, recording duiker dung piles and elephant shrew nests within 3 m on either side and the number of paths that crossed. Furthermore, bushpig activities, such as holes, feeding sides, and dung were recorded within 5 m on either side of the transect. Abundance of squirrels and primates were estimated by observers walking along the path, recording their approximate distance from the path (using 50 m as the cut-off distance) and the group size. Additionally, a range of vegetation characteristics was measured along each transect. The total amount of vegetation cover was noted by assessing the amount of sky obscured when looking vertically upwards through a tube 5 cm in diameter.

Komers et al. (1997) used the pellet matching method to identify the distribution and density of Kirk’s dik-dik (*Madoqua kirki*) in Kenya and Namibia. The occurrence of dung pellets, pellet group sizes, and defecation rates have been used in population estimates of several species (Bennett *et al*., 1940; Emlen *et al*., 1957; Cochran & Stains, 1961; Koster & Hart, 1988). However, these estimates require good knowledge of a species’ defecation rate, pellet decay rates in different seasons, accurate estimation of a dung pile’s age, and the rate of pellet loss due to dung beetles or rain. Defecation rates and pellet counts in Kirk’s dik-dik (Boshe & Lyimo, 1983) proved to be an inaccurate approach for determining changes in population size, especially since different shapes of pellets at various sites confused correct identification of species (Amubode & Boshe, 1990). Komers et al. (1997) expanded on this idea by presenting for the first time the use of pellets from each given dung deposit as a ‘fingerprint’ for specific individuals. Groups of pellets were qualitatively catalogued, by matching the sizes and shapes of pellets in different groups. At the time that the distribution of pellet groups was recorded, no information was available on the population or the identity of each individual. However, the range of individual duiker were eventually inferred from the distribution of distinguishable pellet groups. Fresh dung was collected (a minimum of 30 pellets), air dried for 24 h, and placed in a numbered transparent plastic bag. The position of the collection site was noted on a map. Pellet groups from each newly found dung pile were compared to the existing reference collection, based on differences in shape and size. Additionally, some duikers were captured during moonless nights by temporarily blinding the animals with a strong flashlight. When the capturer approached the individual to within 10 m, a battery powered horn was used to cover the noise of the researcher's approaching steps. When within reach, the individual was grabbed by its hind legs and was marked with unique combinations of colour coded ear tags. Some animals were also fitted with radio collars. Those that were not fitted with collars were observed opportunistically, so that at the end of the study period, the researchers had information on the identity and space use of all individuals in the study area. Comparing the ranges determined by the pellet matching method and by radio telemetry, Komers et al. (1997) found that only one territory border was incorrectly classified by the pellet matching method.

Caro (1999) examined five simple ground-based conservation monitoring methods of deriving densities of large and medium-sized mammals using line transects driven through miombo woodland habitat in Africa. These methods calculated area by dividing the number of individuals seen by

1) an average of each species' sighting distances,
2) a fixed 200 m belt width,
3) the area visible from the centre of the transect,
4) visible area weighted by species' vegetation preferences, and
5) by dividing the number of groups seen by an area visible from the transect.

The different methods produced differing estimates of species' densities and overall biomass, with belt transects giving the lowest estimates. When the ground-based monitoring techniques were compared with aerial surveys, the former proved to be more precise.

Noss (1999) undertook census of rainforest game species using participatory monitoring, communal net hunts. He accompanied net hunts of the BaAka village of

Mossapoula between 1993-1994, registering and measuring captured animals. During every hunt, all encounters with wildlife were recorded - escapees as well as captured animals. From these data, wildlife abundance and density estimates were calculated. BaAka hunts were grouped into three categories according to the distance of the hunt from the settlement.

Waltert et al. (2002) analyzed line transect data using the *Distance* (Distance 4.0; Buckland et al. 2001) sampling method to estimate primate densities in the support zone of Korup National Park, Southwest Cameroon. Line transects were established in logged and unlogged sites and data from diurnal visual and acoustic encounters were used to estimate species densities.

Sunderland et al. (2003) also conducted wildlife surveys using the *Distance* sampling methodology. Additionally to data collected on line transects, which included indirect signs of apes (dung and tracks), ape nests and direct observations (animals seen or heard), random searchers were made for Cross River gorilla (*Gorilla gorilla dielhi*) nests in Takamanda Forest Reserve, Cameroon. These data supplemented information recorded on transects, in an effort to obtain more accurate estimate of gorilla mean groups size and total population in the area.

Noss (2004) suggested that the key to successful wildlife management programs is the effective involvement of local hunters and communities in the monitoring, planning, decision-making and implementation of projects. In order to evaluate the effectiveness of such a monitoring scheme, Noss’s team relied on self-monitoring of wildlife by hunters. Between 1997 and 2000, Izoceno hunters from 22 communities in the Bolivian Chaco voluntarily participated in monitoring their hunting activities, measuring and recording data on captured animals and hunting methods in personal notebooks. Despite the lack of compensation, participation exceeded 60% of all active hunters. However, the data collected were not complete, and there appeared to be a biased towards over-reporting certain hunting methods and prey characteristics. Complementary research is essential to further improve this technique.

Brugière et al. (2005) carried out a census of ungulates in the Haut Niger National Park, Guinea in May 2002 and compared their results to a census from 1997. They analyzed line transect data using the *Distance* technique. This limited the extent that data could be compared between the different study periods. Therefore they limited comparisons to encounter rates of the observed species, as opposed to total population densities.

Refisch et al. (2005) estimated monkey population density in the Taï National Park, Côîte d'Ivoire, using transect previously used by Whitesides et al. (1988). The *Distance* method was used to analyse the line transect data. Two areas differing in extent of poaching were compared.Density estimations were performed for those species that met the minimum statistical requirement of 40-60 observations (Buckland et al. 2001). For those species that did not meet the minimum criteria, they employed sweep samples where four observers walked simultaneously along parallel transects. Transects were 100m apart and had a total length of 1200m.

Waltert et al. (2006) conducted a survey of blue duiker densities (*Cephalophus monticola*) in the Korup region, Southwest Cameroon from 1999 till 2002 using the D*istance* sampling methodology. Both diurnal and nocturnal line transects were conducted. The transects were located in unlogged forest and near a town in a logging concession, where moderate to heavy selective logging had taken place. Permanent 2km line transects were established, each being parallel to and at least 200m apart from each other. Diurnal transects were conducted between 6.30 and 9.00 hours and nocturnal surveys between 19.00 and 21.00 hours with the help of a torch light. Perpendicular distances were measured to the nearest meter from the line to the position of each detected object of interest (Buckland et al., 2001).

Feer et al. (2007) assessed the changes in diversity of duiker within the Ipasse Man and Biosphere Reserve, north-east Gabon by comparing his results, collected in 2005-2006, with information gathered two decades earlier (Feer, 1988; Dubost 1980). They conducted censuses using four different methods, ensuring a pretty reliable assessment of duikers' diversity. Censuses were made along line transects using night-time and daytime visual counts. They also used call points, a method often practised by hunters to attract duikers: hunters imitate the call of a distressed duiker, causing duikers in the vicinity to either run or cautiously approach the caller. Call points were placed every 400 m along four transects. A total of 112 call points were sampled at different seasons by a team of one experienced hunter making the call and one researcher watching for approaching animals. Lastly, duiker presence was assessed with fresh dung counts along seven, 12m wide strip transects. Collected specimens were stored in tubes with silica gel for DNA extraction and sequencing, using a protocol specifically designed for duiker identification using mitochondrial

DNA sequences (N. van Vliet, R. Nasi, P. Taberlet, C Miquel, S. Zundel, unpublished data). The presence/absence data for each species was analysed. In the 1970-1980s, *C, monticola, C. callpygus,* and *C. dorsalis* were identified. The blue duiker was the most abundant species (76-80%), followed by the bay duiker (8-22%) and Peters' duiker (3-12%). In 2005-2006 only blue and Peters'duiker were encountered, with the blue duiker being the most abundant duiker species representing 80-93 % while Peters' duiker represented 7-19 % of the duiker individuals depending on the survey method used. They found no evidence of bay duiker presence in 2005-2006. Considering that similar methods detected the presence of bay duikers in a near by, un-hunted area, the absence of bay duikers in the hunted sites was attributed to hunting.

Noss (1998) conducted a daytime line transect survey using trails including four parallel and straight paths 500 m apart, each roughly 2m wide and 2km long, and two 1.5 km perpendicular paths connecting the ends of four parallel trails. Night surveys were not conducted in this study because of the potential dangers (night-time gun hunters, abundant elephants and gorillas). For each animal seen or heard, the observers recorded species, number, and perpendicular animal-to-path distance. Afterwards the population density was calculated according to Burnham et al. (1980); Whitesides et al. (1988); and Buckland et al. (1993):.

Sunderland et al. (2007) carried out large mammal surveys in order to identify sites for priority conservation in Southwest Cameroon. They used reconnaissance walks (recce) to collect data on animal signs, including dung, nests, feeding signs and human activity signs (i.e. hunting trails, huts, village footpaths, gunshots, cartridges and snares). All recce walks ran perpendicular to existing footpaths and were placed at least 5 km apart from each other. The cutting of recces and recording of habitat types and animal signs followed the procedures of White & Edwards (2000).

Suggestions for the biological methodology related to the Community Based Management Model:

For a wildlife management model it is particularly useful to establish long-term data sets to be able to compare the current abundance and distribution of populations with the past. It is also essential that the methods used permit comparisons with other studies and remain the same over the tears. If methodological modifications are to be made, it is important that the effect of these changes on the data are accessed. For instance, for a period of time both the new and the old techniques can be concurrently used, in order to compare how data from each approach compare.

Data collection should extend over different seasons (wet and dry), ensuring that intra-annual variations are detected. Similarly, in order to be able to estimate the effect of specific human activities, data should be collected at various sites with different land-use schemes (i.e. logged and unlogged, hunted and protected etc.). It is also important that transects (if this method is used) are randomly placed, ensuring that all habitat types are represented.

Night counts along line transects seems to be an appropriate technique for duiker surveys, as according species seem to be less prone to being undercounted with this approach. Both diurnal and nocturnal species can be detected. For instance, duiker estimates from nocturnal surveys were up to four times higher than estimates from diurnal surveys (Waltert et al., 2006). Payne (1992) also reached a similar conclusion about the limitation of diurnal blue duiker surveys (c. 50-57% lower population densities estimates compared to nocturnal surveys).

Due to the poor visibility in the tropical rain forests, the combination of direct and indirect surveying methods may be beneficial. For instance, along with transect surveys, where visual encounters are recorded, information on dung piles, nests, feeding signs etc. can also be collected. At the end, the data of both approaches can be compared. These indirect techniques, despite caveats that they may have, still remain as one of the best option for estimating the abundance of cryptic and elusive species. Therefore, there is value in including them in community based monitoring programs of wildlife.

Call points, a method often practised by hunters to attract duikers, also has the potential of being included in the set of methods used during a long term monitoring program.

Effective methods are also needed to measure wildlife harvest rates. Such records will permit to interpret the wildlife density data in light of hunting pressure levels over time. Participatory monitoring techniques should compliment conventional analysis such as socio-economic questionnaires and surveys of bush-markets. In addition, researchers could accompany hunters in their hunting trips. However, hunters do not randomly use their forest nor is this method likely to be useful for estimating hunting pressure on rarely encountered/hunted species (even though the pressure may still be high). The method of "self-monitoring" by hunters may generate data of doubtful usefulness, but could be an appropriate way for generating awareness among the local populace of wildlife management issues.

Aerial photographs could be useful for monitoring large scale land-use changes over time, using a GIS for analysis.

Lastly, the importance of fisheries as a wildlife resource should not be ignored, as changes on the status of local fisheries could be reflected on changes in hunting pressure. For instance, Mdailhli et al. (2003) used information on fisheries on the southern border of Takamanda Forest Reserve in drafting a participatory plan for the protected area.

4.2 What could be the maximum sustainable harvest rate of bushmeat species?

Given population densities of game species, maximum sustainable harvest rates can be estimated using calculation models, such as the Robinson and Redford model (1991). This topic is examined in more detail in another discussion paper of this series.

5. Extent of hunting zones

Most studies of bushmeat hunting have given little or no attention to the definition of territories or catchments areas from which animals are "harvested" or "cropped". Such information is very important to an understanding of hunted dynamics as well as for assessing the impact of hunting on prey populations. To define hunting zone sizes, human activity signs (snares, hunting trails) should be recorded and input in a GIS. This way, the extent and level of hunting pressure in a hunting zone can be monitored over time and used to interpret the wildlife density data. Hunting zones can also be classified according to their overall distance from human settlements – the ones being adjacent to farms considered as core hunting zones, ones further away as intermediate hunting zones, and those that require extensive travel as expedition hunting zones. It may well be that different type of animals and at different times of the year are hunted in each of these hunting zones, depending on farming activities etc.

6. Agroferestry and captive breeding of wild species

Agroforestry is a land-use type that combines some forest cover with the production of food or cash crops. Livestock can also be combined in such systems. Instead of domesticated species, the use of wild species (wild mini-livestock) has been considered over the years, often with mixed results. However, there is still unexplored potential for some vertebrate or invertebrate species. For instance, Giant African snails (*Achatina* spp) are a highly prized food in West and Central Africa and Asia, and are produced commercially at some locations. Among vertebrates, large African rodents have attracted considerable attention (i.e. the cane rat *Thryonomys swinderianus*, the giant pouched rat *Cricetomys emini*, and the brush-tailed porcupine *Atherurus africanus*). Unlike solitary duikers, who give birth to a single off-spring, rodents have higher growth rate and shorter life-histories, which could make them more suitable for viable small-scale harvesting schemes. The bursh-tailed porcupine specifically (*A. africanus*) appears to be easily adapted to captivity and seems to be little affected by stress. Moreover, it is a polyembryonic species with a rapid growth rate similar to that of other African rodents (Anizoba, 1982; Adjanohoun, 1992). Although its current productivity in captivity is limited to a single young per birth and two to three births per year per female, this species could be a good candidate for minilivestock programmes in African forest areas if its current reproductive potential in captivity could be improved.

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