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Hunting management: the need to adjust predictive models to field observations

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ABSTRACT

Wild meat is a major protein supply for numerous traditional communities worldwide, but impacts ecological processes and consequently challenges the relevance and suitability of adequate monitoring of the sustainability of harvests. In this study we discuss the classic models of theoretical "maximum sustainable offtake" and propose new considerations on sustainable harvest thresholds. The study focuses on French Guiana, northern Amazonia, on four sites harvested by three communities (Amerindian, Creole, and Hmong), mainly for subsistence purposes. We explored how factors related to the number of hunters, the harvested areas, and the surface area hunted, and measured how fauna abundance generates uncertainties on models and increases the errors on sustainable thresholds. Biased or incomplete ethnologic surveys, as well as local and temporal variations in game species density could lead to considerable underestimation of harvests. We proposed a set of corrections that, once applied to the input variables of the offtake model, could limit the risk of erroneous assessment of sustainability thresholds.

Keywords: Bushmeat, Ethnozoology, Ethnoecology.

INTRODUCTION

Wild meat has for millennia been a major protein supply for numerous traditional communities worldwide (Bailey et al. 1989; Bennett 2002). However, local communities harvesting wild meat may have a substantial impact on wildlife populations (de Thoisy et al. 2009; Fa et al. 2002; Redford 1992; Sodhi et al. 2004; Wright 2003), ecological processes, and forest dynamics (Balée 1984, 2006; Holbrook and Loiselle 2009) and even the climate (Brodie and Gibbs 2009). The sustainability of harvesting animals has consequently attracted the attention of conservationists, managers, and politicians over the decades.

In Amazonia, investigations on the consequences of harvesting wild animals have provided "sustainable harvest" models based mainly on the density and productivity of target species (Redford 1992; Redford and Padoch 1992; Redford and Robison1987; Robinson and Redford 1991; Robinson 2000; Robinson and Benett 2004; Silvius et al. 2004), but often the seasonal variability of factors such as the number of hunters, effective surface area of the area hunted, and local and temporal fluctuations of game species density are not fully considered.

More recently, studies have proposed adjustments of the early models (Robinson 2000; Robinson and Bennett 2004), with better quantification of the human factors related to the purposes of harvests, i.e., the maximum

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number of people depending on meat and per capita consumption of wild meat. Others emphasized game subsistence, semi-subsistence, and commerce in tropical regions (Bennet et al. 2002; de Thoisy et al. 2005; Kunz and Blum 2009; Levi et al. 2011; Ling and Milner-Gulland 2006; Lopes et al. 2000; Mena et al. 2000; Nasi and van Vliet 2011; Pangau-Adam et al. 2012; Swamy and Pinedo-Vasquez 2014, Van Vliet et al. 2010; Zapata-Rios 2009). Some, though rare, considered the consequences of hunting practices on wildlife populations using cross-analysis of ethnoecological and fauna data (Mbotiji J 2002; Nasi et al. 2008; Robinson and Bennett 2002).

The present study follows this theoretical orientation. This paper aims to discuss these models (Robison and Redford 1991; Robison and Bennett 2004) and propose new considerations on sustainable harvest thresholds. Robinson and Redford's model was criticized from the point of view of the foundations of the biological data on which it based (density, reproduction rates) and it was considered too simple to report the actual situation. The biological data of the hunted species remain fragmented, and they need to be refined, as do the data on the anthropological factors (number of hunters, hunted surface areas). Yet these anthropological factors very strongly influence the calculation of the potentially exploitable quantities of game. Our purpose was therefore to reduce the uncertainties in the ecological and anthropological factors and thus to contribute to an adjustment of the results obtained by increasing the influence of these factors.

Considering the importance of the human factors taken into account in sustainable harvest models, meticulous ethnoecological fieldwork is required to quantify these factors. Furthermore, these inquiries are a precondition for the implementation of linear transects of hunted and nonhunted sites. The collaboration of human and natural sciences is therefore an essential condition to the implementation of a coherent protocol in a fauna survey for self-sufficiency and/or commercial purposes.

Regarding fauna surveys, some key parameters strongly influence the density of target species calculated with the linear transect method: the sampling effort (i.e., the cumulated distance of the survey); the method to calculate game density from field records; and the density and detectability of game species that may vary according seasons, and biogeographic patterns. Only a high number of surveys on a wide variety of sites can provide an overview of local and seasonal species abundance, which, if not considered in a sustainability model, could result in highly biased stock assessment. The variables considered in the sustainable harvest model developed by Robinson & Redford (1991) assess both current game stocks and their likelihood of survival, according to harvest areas and the number of hunters. However, these variables are difficult to assess for short-term periods. The variation of the number of hunters between dry and wet seasons, and insufficient assessment of harvested areas may induce significant bias in quantifying the game harvest, and consequently in the predicted viability of the remaining stock. Last, field-measured game densities vary between sites and within sites, depending on the season and year (de Thoisy et al. 2010), which complicates the identification of baseline values for inclusion in sustainability models.

Based on ethnoecological data gathered from three traditional communities in French Guiana (Renoux 2002), including seasonal hunting practices, yields of hunting trips, and harvested biomass, we aimed to explore how hunting patterns may influence the maximum sustainability off-take model. It should be noted that the results obtained from the modified model proposed herein indicate only trends. On one hand, these must be refined by monitoring (ecological and ethnological) and, on the other hand in the final analysis, they should be interpreted according to an ecological and ethnological reflection. Hunting practices reduce the fauna and also depend on social, economic, technical, and often religious considerations. This obvious fact must always be considered when developing a sustainable game management model.

MATERIAL AND METHODS

Study area

This study was conducted in French Guiana, a French administrative unit covering 84,000 km^2 located in the northern part of South America, on the Guiana shield. The Guiana shield is one of the largest pristine Neotropical rainforest blocks and a floristically distinctive province compared to the Amazonian basin (Lindeman and Mori 1989). Eighty percent of French Guiana is covered by moist upland forests occurring on well-drained lateritic and oligotrophic soils at altitudes between 0 and 600 m. The alluvial coastal plain is covered by marsh forests, savannahs, transition forests, and herbaceous swamps and is rather narrow on this part of the Guiana shield (de Granville 1988).

Compared to other Neotropical countries, the forest conservation status of this area (Eastern Venezuela, French Guiana, and the Brazilian states of Amapa and Para) is still quite favorable. French Guiana has an extensive network of protected areas including five nature reserves, located in areas on the northern half of the country, and a national park in the south, for a total protected area of 23,000 km^2 (>25% of the country).

From a cultural point of view, French Guiana has a mosaic of human communities of diverse origins, accessing natural resources for different uses. Bushi-Nengue and Amerindians are the only communities using plant and animal forest resources for traditional and subsistence uses. Bushi Nengue (Aluku, Djuka, Saramaka, Paramaka, Boni) are located mainly along the Maroni River, on the Suriname border. Among Amerindians, Wayãpi and Emerillons (Tupi-Guarani group), and Wayana (Karib group) are located on the Oyapock River (Brazilian boundary) and on the upper Maroni. Galibi (Karib group), Arawak, and Palikur (Arawak group) are located in coastal areas. Together with these traditional communities, Hmongs (who migrated from Laos), Brazilians, Creoles, and Metropolitans may also use natural resources for business, subsistence, and recreation.

Ethnoecological surveys

Ethnoecological surveys were conducted on four sites (Fig. 1), for 8 months (4 months during the dry season, 4 during the wet season) for a total of 990 days. Qualitative and quantitative data were collected with both direct interviews and questionnaires provided to hunters of the communities surveyed. The information requested was personal data (age, name), trip duration, transport methods, weapons, harvest sites, and age and gender of harvested species. The hunting sites were specified on the basis of the toponymy known to hunters. We considered the toponymy as a very good indicator of geographical referencing. First, it is the simplest way to inform the community of the place where hunting is planned or the other hunters of the presence or absence of game in sectors in a particular area. Second, use of toponyms also asserts a historic presence on a site by defining its content and border. They therefore need to be as precise as possible. On this basis, the point of departure and arrival were generally named by hunters. In uncertain cases, small streams, large trees, or rocks near a known toponym were geography referencing points.

We were thus able to acquire in situ data connected to the seasonal hunting practices from the technical, economic, sociological, and religious points of view, particularly in all the cultural communities (Amerindians, Creole, Hmong) that participated in the study. Besides these contextual interviews, many of the others were conducted over nearly 10 years with hunters of miscellaneous cultural communities (Ouhoud-Renoux 1998; Renoux 2002).

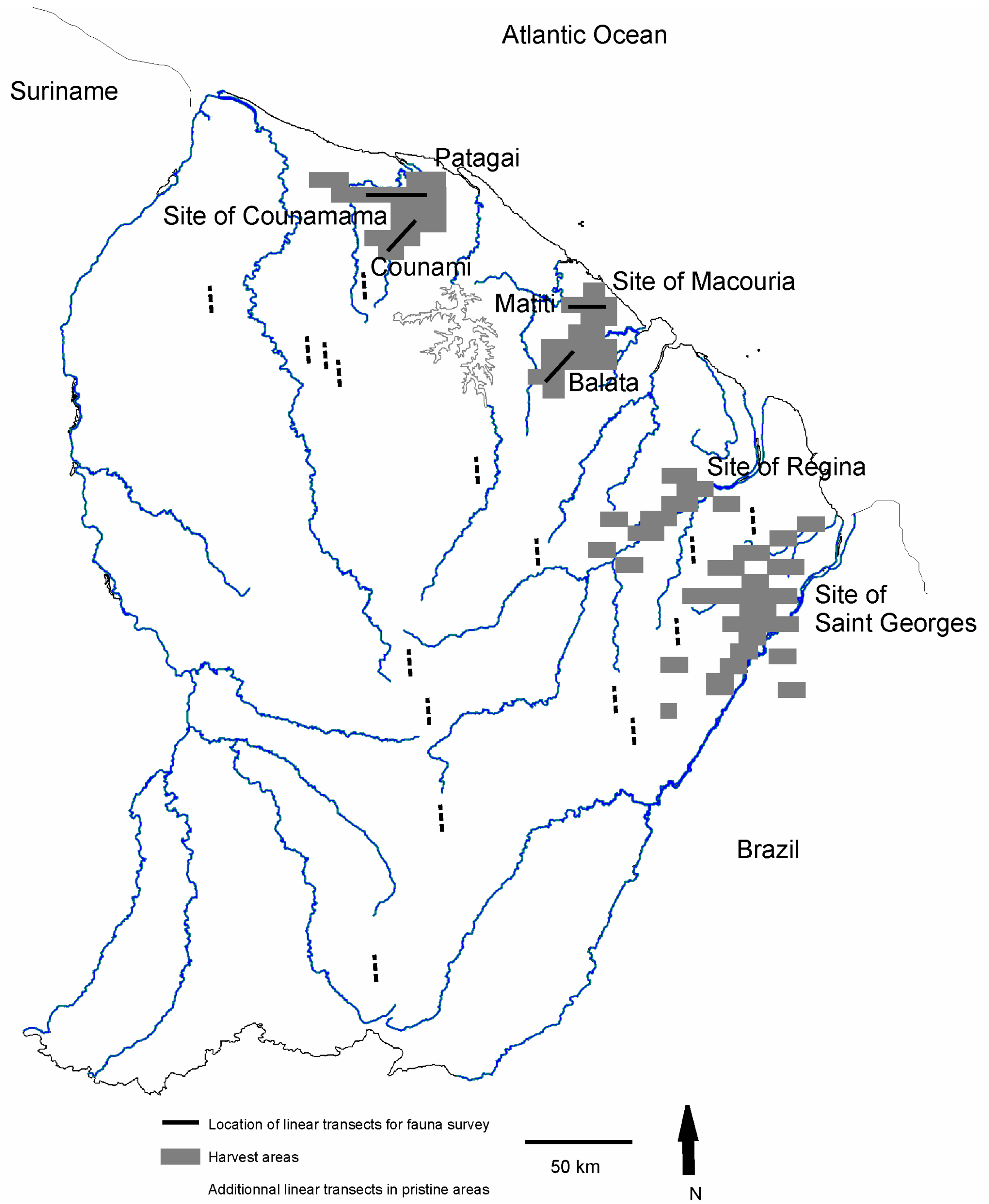

Figure 1. Study sites for monitoring wild game harvests and fauna, French Guiana

Fauna surveys

Linear transects were used in faunal surveys, because they provide relevant estimations of diversity and abundance for neotropical forest species (de Thoisy et al. 2008). The surveys were implemented on two of the above-mentioned sites: one site (Macouria) was hunted by Palikur tribes, although the second site (Counamama) was hunted mainly by Hmongs. Data gathered during ethnoecological surveys evidenced two main harvest areas for each site (Macouria: Balata and Matiti; Counamama: Counami and Patagaï). We conducted a fauna survey in each of the four wild meat harvest areas. The survey was implemented on a 4- to 5-km linear trail, walking slowly (1–1.5

km/h) with daily repetitions, for a total of 100 km, which allowed a relevant estimation of richness and abundance of several species (de Thoisy et al. 2008). We analyzed line transect data using Distance v. 6.0. (Thomas et al. 2010). Of the methods classically used to obtain density from the linear transect, the Leopold method uses the mean sighting distance to assess the effective strip width, the Green method uses the maximum sighting distance, the Kelker method, and Fourrier series-based methods. We also modeled detection functions using half-normal key functions with Hermite polynomial adjustment to provide a more elaborate assessment of density (Thomas et al. 2010). To better assess the seasonal and/or annual variability of measured abundance, together with surveys implemented in the harvested areas of interest, linear transects were also conducted repeatedly during dry and wet seasons at two sites, including one hunted and one not hunted, and repeatedly every year for 5 years. Regional variability of measured abundance was assessed using a set of surveys in pristine areas, either relying on previously published data (de Thoisy et al. 2009) or on new sites (Fig.1).

Both ethnoecological and fauna data were included in a GIS, locating the extracted biomass and implementing the linear transects on the hunting areas. We focused the cross-analysis on two sites, one harvested by Amerindians (Palikur tribe, Macouria site, including the Balata and Matiti hunting sites) and one harvested by Hmong (Counamama site, with the two Counami and Patagai harvest sites). On both sites, hunting pressure has been present for more than 40 years. A grid was drawn for the north of the country with cells measuring 5 km × 5 km, and we considered that every recorded harvest occurred in an area covering 25 km^2. We hypothesized that we could estimate the area of a site that had actually been hunted from two types of data: the mean duration of a hunting outing and a hunter's average speed of travel (0.8 kph). From these data, it was relevant to show that from a starting point (tracks, trails, and rivers), a round trip route that added up to nearly 5 km, is a maximum depth of 2.5 km from any point. Using this distance, we generated buffers (2.5 km) around forest tracks and rivers on the Counamama and Macouria sites.

RESULTS

Game species density: a heterogeneous variable

Focusing on primates, a group with both greater conservation issues and that can be easily detected in the field, precluding sampling bias, we observed high variation in the calculated densities, depending on the methods used to transform abundance to density (Table 1).

Table 1. Densities of three monkey species in the Counami harvest area (expressed in individuals / km^2), calculated with four concurrent methods.

	Green method	Leopold method	Fourier series	Distance ©
Alouatta macconnelli	5.0	12.4	7.2	7.3
Saguinus midas	6.9	19.2	12.3	12.4
Sapajus apella	5.6	16.5	8.2	8.2

Second, abundance levels of the four primates that are regularly targeted by hunters showed great geographic, seasonal, and temporal variations (Figs. 2, 3a, 3b).

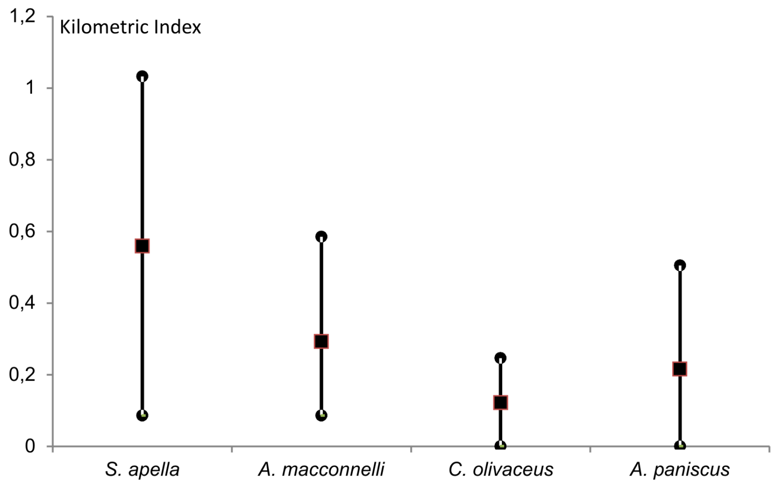

Figure 2. Wild meat harvest areas on three sites, showing the overestimation of catchment size using the quadrat vs. the buffer method.

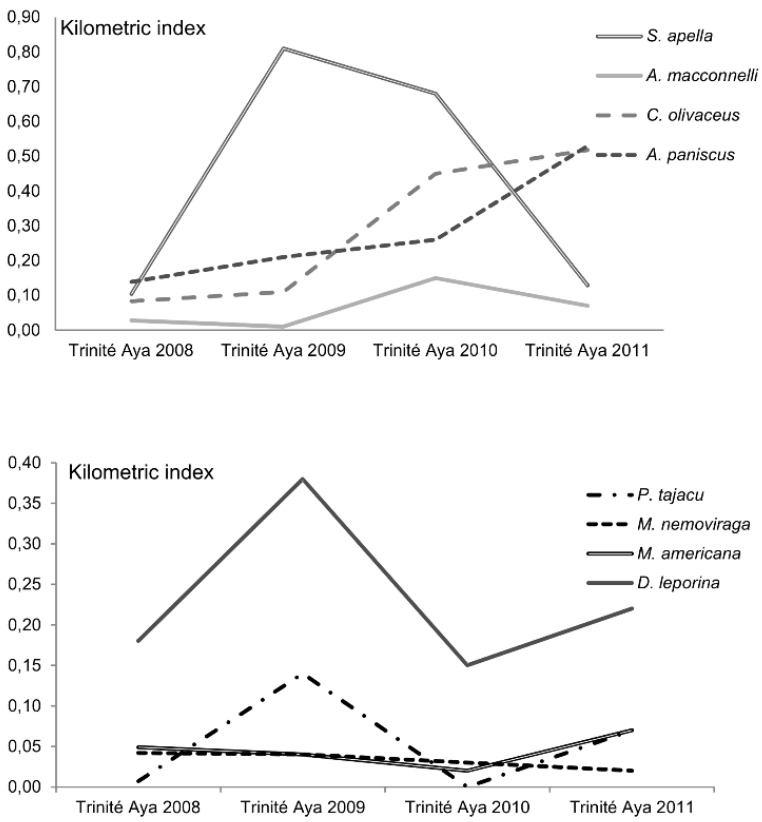

Figure 3. Yearly changes in abundance (expressed as the number of sightings / km) of four primate species (*Sapajus apella*, *Alouatta macconnelli*, *Cebus olivaceus* and *Ateles paniscus* (Above) and three ungulates and one rodent on a nonhunted site (*Pecari tajacu*, *Mazama nemoviraga*, *Mazama americana* and *Dasyprocta leporine*)

Variables related to hunting area

The data allowed identifying the size and the catchment areas (Table 2), but also showed that the mean hunting trip lasted 5–6 h, for a maximum hunting trip 10–15 km long.

Table 2. Harvested areas, distribution with other communities, and number of hunters, on each site.

	Macouria	Counamama
Surface area hunted (quadrat method / buffer method)	278 / 250 km^2	405 / 375 km^2
Territorial Competition	√	√
Number of hunters	41 (ws: 57)	28 (ws: 35)
Duration (months)	8	8

ws: wet season

Our interviews suggested that the mean harvest trip was around 50% of the maximum hunting trip with a maximum linear distance from boat or car access less than 2.5 km. Using this distance, we generated buffers around forest tracks and rivers on the Counamama and Macouria sites (Table 2, Fig.4). Comparison of grid cells and buffers showed that the harvest areas were 5–13% smaller when using the buffer method than when using the grid with 5×5-km cells, and that 90% of harvests remained within the 2.5-km buffers. Consequently, we suggest that, when assessing the size of the catchment area with a grid cell method, which is the easiest to implement, a factor of +10% should be applied to the calculated size.

Variables related to hunters

The number of hunters also strongly influenced the sustainable harvest of game, and for this reason it is important to pay very particular attention to its estimation. From our communities, we show that (Table 3), during the 33 months of surveys, the hunter population varied depending on the season (≈ ± 25%), in relation to other economic activities (agriculture, temporary work). Also, because of rains that preclude efficient hunting, hunting trips were less efficient, much shorter but more numerous during the rainy season (Table 3).

Table 3. Temporal variation of hunting practices and harvests on the two sites.

	Dry season	Wet season
Hunters (n)	69	92
Hunting trips (n)	135	242
Hunting trip duration (mean)	6	5.3

The seasonal difference in the number of hunters is important in this study. We noted the same thing in previous studies and can therefore confirm here that the number of hunters is consequently difficult to assess with only short and/or seasonally biased interviews. Furthermore, whatever the duration of the interview campaigns, it is almost impossible to obtain an exact number of hunters using an area, since most sites are used by different and often competing communities. We consequently increased the number of hunters by a minimum of 25% when surveys were conducted during the dry season only and in areas harvested competitively by several communities. Naturally, the value of this correction can vary according to local and geographical socioeconomic conditions. It must be emphasized that this correction establishes only the basis of a calculation, which contributes to obtaining a safety margin to define a trend.

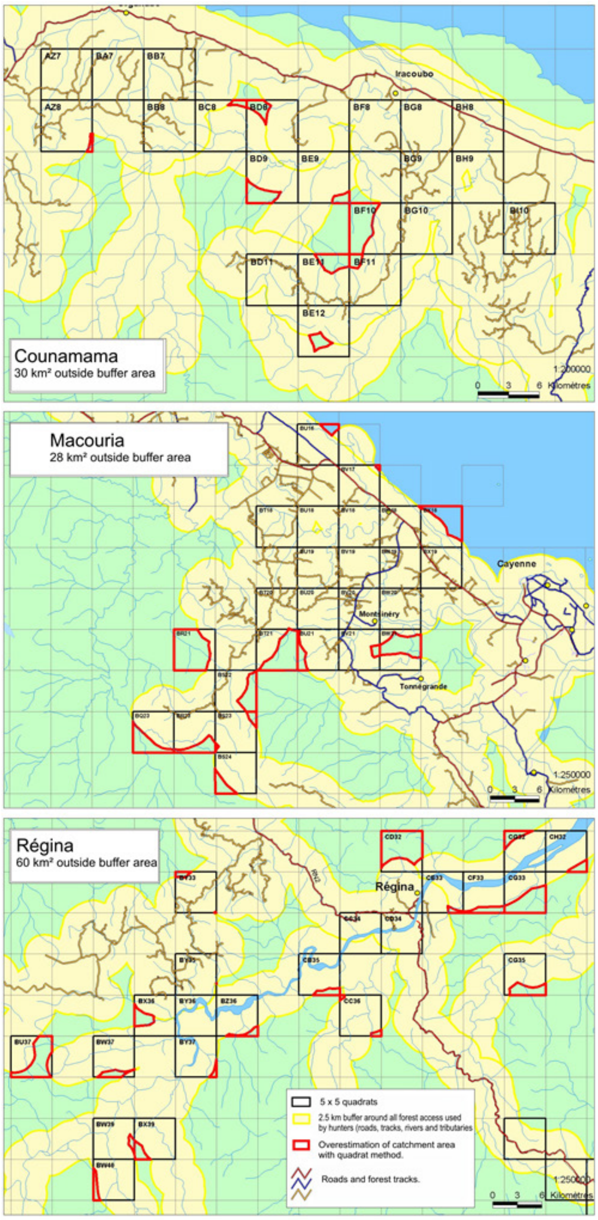

Figure 4. Wild meat harvest areas on three sites, showing the overestimation of catchment size using the quadrat vs. the buffer method.

Sustainable harvests and calculated thresholds

In the first example, we tested the sustainability of harvests on the study sites. The thresholds calculated with the density values provided by Robinson and Redford, without any correction, are shown in Table 4. No species was subject to overharvesting, suggesting that current hunting pressure does not threaten the long-term survival of animal populations. Table 5 shows the maximum thresholds with the same extracted biomass but applying (i) recorded densities and (ii) 50% of the maximum threshold in order to consider the uncertainty related to the method used to calculate density and the associated sampling effort, (iii) –10% harvest areas, and (iv) +25% hunters

Table 4. Sustainability of harvests in the four hunting areas vs sustainability of harvests in the four hunting areas applying several correcting factors, including field recorded densities, 50% of the maximum threshold, a 10% decrease of harvest areas, a 25% increase of the number of hunters.

		Density		Pop. size in the area (a)		Ratio that could be harvested** (b)		Nb of animals that could be harvested (c)		Nb of animals harvested (d)	Harvest / max potential harvest % (e)	
		Est.	Cor.	Est.	Cor.	Est.	Cor.	Est.	Cor.		Est.	Cor.
	Sapajus apella	9.82	10.70	2455	2408	3	3	73.65	72.2	31	42.1	42.9
Macouria,	*Dasyprocta leporina*	18.7	6.25	4675	1406	80	80	3740	1125	52	1.4	4.6
Balata	*Mazama americana*	5.67	0.70	1418	158	20	20	283.6	31.5	4	1.4	12.7
(250 km^2)	*Mazama nemoviraga*	8.12	2.00	2030	450	25	25	507.5	112.5	4	0.8	3.6
	Tayassu pecari	5.24	25.00	1310	5625	26	26	340.6	1462.5	66	19.4	4.5
	Pecari tajacu	8.05	3.30	2013	743	50	50	1006.5	371.3	8	0.8	2.2
	Sapajus apella	9.82	4.60	1228	518	3	3	36.84	15.5	5	13.6	***32.3***
	Alouatta macconnelli	19.32	3.00	2415	338	3	3	72.45	10.1	16	22.1	***158.4***
Macouria,	*Dasyprocta leporina*	18.7	5.60	2338	630	40	40	935.2	252	21	2.2	8.3
Matiti	*Mazama americana*	5.67	0.70	709	79	20	20	141.8	15.8	1	0.7	6.3
(125 km^2)	*Mazama nemoviraga*	8.12	2.70	1015	304	25	25	253.75	75.9	1	0.4	1.3
	Tayassu pecari	5.24	4.30	655	484	26	26	170.3	125.8	39	22.9	**31.0**
	Pecari tajacu	8.05	1.70	1006	191	50	50	503	95.6	3	0.6	3.1
	Alouatta macconnelli	19.32	3.60	200	324	3	3	6	9.7	3	50.0	30.9
Counamama,	*Dasyprocta leporina*	18.7	6.25	300	563	40	40	120	225	4	3.3	1.8
Patagaï	*Mazama americana*	5.67	0.70	100	63	20	20	20	12.6	1	5.0	7.9
(100 km^2)	*Mazama nemoviraga*	8.12	0.70	100	63	25	25	25	15.8	1	4.0	6.3
	Tayassu pecari	5.24	1.50	900	135	26	26	234	35.1	11	4.7	***31.3***
	Pecari tajacu	8.05	5.30	2900	477	50	50	1450	238.5	36	2.5	15.1
	Alouatta macconnelli	19.32	12.40	2898	1674	3	3	86.94	50.2	3	3.5	6.0
Counamama,	*Mazama americana*	5.67	0.90	851	122	20	20	170.2	24.3	3	1.8	12.3
Counami	*Mazama nemoviraga*	8.12	3.10	1218	419	25	25	304.5	104.6	4	1.3	3.8
(150 km^2)	*Tayassu pecari*	5.24	_	786	_	26	_	204.36	_	_	_!	_
	Pecari tajacu	8.05	3.80	1208	513	50	50	604	256.5	4	0.7	1.6

Est : Estimated, Cor. : Corrected, *(Robinson & Redford, 1991), **(Robinson, 2000), (a): density x size of the catchment area (buffer method), (b): % of specimen that can be sustainably harvested (Robinson & Redford 1991), (c): number of specimens that could be harvested in the catchment area, (d): number of harvested specimens, (e): (c) x (a)

DISCUSSION

This paper has addressed the importance of a multidisciplinary approach, both to promote an ecosystemic vision of conservation programs and to highlight the relevance of an ethnoecological view when considering game sustainability models.

In French Guiana, as in other tropical areas, hunting practiced by native communities allowed maintaining microscale economic activity based on the harvest and sale of natural resources (Ouhoud-Renoux 1998; Renoux 2002). These traditional meat harvest activities may occur in areas, or close to areas, undergoing conservation measures. Consequently, the impacts of wild meat harvests on local and regional biodiversity have to be adequately measured. Also, practical and legal conditions have to guarantee the opportunity for these communities to continue their traditional way of life in areas that they have occupied for centuries.

Suitable assessments of populations and harvests encounter a number of methodological problems. From a theoretical point of view, we suggested using sustainable harvest thresholds (Robinson, 2000), derived from observed densities, and we used successive error ranges in order to ensure that observed meat harvests remained below the theoretical threshold of sustainability. Considering the insufficiency of current knowledge on biotic and abiotic drivers of diversity and species abundance, use of mean values for density may distort the assessment of the threshold of sustainable harvests. Consequently, we suggest that densities measured *in situ* are the only relevant basis for calculating thresholds, and we suggest a confidence interval.

For modeling some authors suggest using indirect evidence of hunting activities as a proxy for the number of hunters such as the number of gun shots (Peres 2000), the number of carriages (Cullen & al. 2001), the number of dog and hunter tracks, or the number of spent cartridges (Wright & al. 2000). We consider that these indications cannot precisely measure the number of hunters in a given area. We favored two simple, quantitative and easily adjustable variables: the number of hunters and the size of the catchment area. The range of variability of these two variables has a direct and quantifiable impact on modeling. Our data show the importance of conducting mid-term fieldwork to include seasonal variations of game harvests related to both hunting practices and biological patterns of game species allowing hunters to optimize yields of hunting trips. Hunting practices follow a seasonal pattern due to other tasks, such as agricultural needs (slash and burn activities) and opportunities for more lucrative activities (cassava flour, traditional crafts, the wood industry, construction work, etc.). The weighting factor we propose to apply on the variable related to hunters makes it possible to consider these seasonal variations. Furthermore, some methodological concerns should be mentioned when evaluating socioeconomic constraints and the needs of indigenous communities. More specifically, demography is not usually considered in conservation initiatives, although substantial population expansion results in substantial qualitative and quantitative modifications of natural resource harvesting, often associated with the overexploitation of areas that communities have been given for subsistence uses (Renoux 2002; de Thoisy et al. 2005).

Considering our results, which indicate rates below what is theoretically harvested, it seems necessary to adopt urgent conservation measures for some game species, notably *Alouatta macconnelli* at Macouria matiti (158.4%) and Counamama Patagaï (30.9%), *Sapajus apell*a at Macouria Balata (42.9%) and Macouria Matiti (32.3%). The hunting pressure on primates is largely explained by an optimization of captures during the period when the botanical species comprising their diet bear fruit. This is less true in Counamama Counami where, for cultural reasons, Hmong consume many fewer primates than the other communities (Palikur and Creole). However, these explanations alone do not allow researchers to determine these trends as being above or close to the sustainable harvest.

The model proposed herein works correctly when attempting to estimate harvest quantities of game in a given area and according to a number estimated by hunters. Importantly, the correction rates we proposed for the definition of the threshold of sustainable harvests need to be adjusted to every case study, not only to ensure the conservation status of a species, but also to guarantee local communities access to protein resources. In a context of conflictual land use, unmanaged threats on both species and habitats (Constantino 2016), both excessively permissive and overly conservative approaches of traditional hunting management should be avoided. Local communities should not suffer from inadequate and unfair coercive measures, only because other pressures are assumed to be more complicated to mitigate.

The rates of wild meat harvesting that the corrected model has calculated show trends with a sufficient safety margin to be able to adjust policies and management of fauna preservation. However, it should be remembered that these rates do not explain the causes of the trends that they highlight. If managers intend to adopt relevant management of game over the long term, it seems desirable to closely consider the causes of these trends, which may depend on very diverse factors.

With an eye to the future, conservation measures should be supported by an accurate understanding of the causes of over-harvesting and unsustainable use of nature resources so as not to be forced to make permanent, unproductive and sometimes coercive adjustments (Makagon et al. 2014; Shoereman-Ouimet and Kopnina 2015). It is on this point that conservation policies generally show shortcomings because counterintuitive conservation measures can finally cause more problems than they solve.

Offtake rates are indicators of trends. However, these indicators do not assume the underlying cause of these trends (Constantiono, 2016), but they must help ask pertinent questions in order to provide pertinent responses. Can change social that impose, first, the technical changes, be the cause? Can collision between traditional social cohesion in hunter communities facing news adjusts and rules imposed by modern exogenous constraints be the cause? Does anomic behavior in the way of hunting stand out as of modern expected well-shared standards? Finally, can biotic and abiotic factors act in unison with these social causes on the observed trends? This list of questions is not exhaustive but opens to rather wide exploration of recurring problems in an attempt to launch a constructive analysis. Relevant answers require time, which is increasingly lacking in basic research. As for the present study, the corpus of ethnographical data that we have accumulated over the long term in French Guiana has allowed us to highlight an important factor in assessing the rates of meat harvesting: the notion of "probity " in the cynegetic act is appropriate to most of communities with which we worked (Ouhoud-Renoux, 1998; Renoux, 2002). First, this notion is philosophical and/or religious, which in no way limits the power that must be exercised to limit individuals' activities. In addition, the notion of the global economy exerts a powerful influence over socioeconomic behaviors.

CONCLUSIONS

A key goal of this article was to propose an improved and relevant calculation method to explore three objectives: (i) conserving species that participate in forest dynamics and regeneration; (ii) maintaining sustainable exploitable stocks of hunting game that constitute a food resource, either for self-consumption or for commerce; (iii) maintaining traditional communities in their self-sufficiency hunting practices, essential to their economy and their social cohesion as well as enabling rural communities to exploit a marketable resource. These three objectives may appear antithetical, but they are not. To be maintained over the medium and long term, implementation of restrictive management plans and coercive policies require staff in large numbers and substantial financing, which is actually rarely granted and results in little return on investment. For most of Amazonian native communities, game is either a food resource necessary for subsistence or a market resource for local economies to keep their communities out of poverty. Before setting up plans to manage game, this should be taken into account. Should the opposite occur, two problems are likely to arise: either these communities turn to an inconsistent mode of subsistence (purchase of imported food requiring incomes that they rarely possess) or they invest ever-increasing human and technical resources in hunting practices to participate in a speculative market (scarce supply and strong demand). These two orientations are very often the sociological markers that announce a social disorganization of communities, a loss of the values of their cosmology and thus a breakdown of their social cohesion. At stake is the long-term maintenance of game stocks that can satisfy real needs while being ecologically and economically sustainable.

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