# Effects of Poaching on Game Species Communities in the Chiquibul Forest

## Study No. 2

# Effects of Poaching on Game Species Communities in the Chiquibul Forest
## Study No. 2

From: Friends for Conservation and Development (FCD)
Prepared by: Boris Arevalo
Funded by: Protected Areas Conservation Trust

# Effects of Poaching on Game Species Communities in the Chiquibul Forest

## Study No. 2

## ABSTRACT

*Unsustainable hunting of terrestrial vertebrates, is a widespread phenomenon in tropical forests; even small-scale subsistence hunting can result in marked population declines in large-bodied birds and mammals. Our 2016 results indicate that overall game species mean abundance was not significantly different from that of 2013, but mean abundance of large bodied game species was significantly greater in 2016 than in 2013, while there was no significant difference between the two periods in mean abundance of medium and small game species. Based on the hunting pressure stratification, no significant difference was recorded. This may be an indication that game species distribute evenly throughout the Chiquibul Forest, thus making it very difficult to isolate the true impact of illegal hunting on target species. Evidence of hunting observed along trails and illegal camps, makes us infer that illegal hunting is occurring throughout the Chiquibul Forest.*

# INTRODUCTION

Unsustainable hunting of terrestrial vertebrates, is a widespread phenomenon in tropical forests (Peres 2000). This most widespread form of non-timber forest product resource extraction (Peres, 2001) can be detrimental to target populations (Mittermeier, 1987). Mittermeier (1987) even suggests that hunting can be more detrimental than habitat destruction, as it may lead from local to regional population extinctions. Thiollay (1986) and Peres (1990), indicate that even small-scale subsistence hunting can result in marked population declines in large-bodied birds and mammals; altering trophic levels, eventually affecting forest dynamics, given that many large terrestrial vertebrates play an important role in seed dispersal, predation (Peres & van Roosmalen, 1996) and herbivory (Dirzo & Miranda, 1991). These factors impact tree species distribution and forest structure modification.

Unlike other anthropogenic disturbance such as deforestation, fragmentation and degradation; over-hunted areas are impossible to be detected and mapped using conventional remote-sensing technology, thus its quantification remains a challenge since the presence of large trees and forest canopy does not guarantee the presence of native fauna (Redford, 1992). It is only possible to speculate on an observable decrease in densities (or relative abundance) of large-bodied terrestrial vertebrates (Peres 2001). The effects of many anthropogenic disturbances operate synergistically, for example in the Chiquibul Forest hunting is an opportunistic activity conducted by individuals involved in illegal logging and Xaté extraction. These activities alone may contribute significantly to a reduction in large-bodied terrestrial vertebrate densities due to habitat disturbance.

The objectives of this study were to i) determine the diversity of terrestrial games species, ii) calculate the relative abundance by species for each game species recorded, and iii) compare the diversity and abundance of game species with the 2012 and 2013 surveys.

## METHODOLOGY

### Study site

This study was carried out in the Chiquibul Forest (Figure 1), located within the Cayo District, covering an area of 176,999 ha (437,376 acres). It comprises of three protected areas being the Chiquibul National Park (106,838 ha), Chiquibul Forest Reserve (59,822 ha) and the Caracol Archeological Reserve (10,339 ha), with central UTM coordinates 1,878,200 – 1,871,800 North and 265,600 – 322,600 East. Meerman and Sabido (2001) identified 17 different ecosystems within the area, all being variants of Tropical Broadleaf Forests, except for a pine forest and a small non-mechanized agricultural category. The region has a subtropical climate with a marked dry season between February to June and a rainy season coinciding with the hurricane season which starts from July to November (Salas & Meerman 2008).
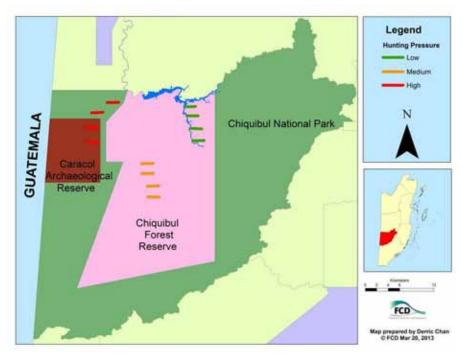
***Figure 1:*** *Distribution of line-transects surveyed during the study.*

**Data collection**

Employing the same methodology and research effort as in 2012 and 2013 surveys, a total of 384 km (128 km per stratification) of standard line transects were surveyed. This methodology provides for a rapid, efficient and reliable assessment of species richness and abundance (Peres 1999), which are important factors in determining conservation priorities (Silveira *et al*. 2003). Transects were established following a stratified-systematic sampling design, based on accessibility. Stratification was based on a hunting pressure gradient - assuming that closer to the Belize-Guatemala International Border the greater the hunting pressure and a reduction in intensity as one moves deeper into Belizean territory. Stratification was based on three categories being high [numerical value of 3] in the Caracol Area, as this is closer to the Belize-Guatemala border); medium [numerical value of 2] in the Millonario to San Pastor Area; and low [numerical value of 1] in the Macal and Raspaculo River areas furthest away from the Belize - Guatemala border) hunting pressure. Four line-transects measuring two kilometers (distance measured with a GPS unit) each oriented East-West were established in each strata, each located 2 km apart. Transects were opened using machete and compass and marked (using flagging tape and paint) at every 50 m indicating distance. All transects were opened with a width of no more than 1 m, clearing only necessary vegetation and allowed a "rest period" (left alone by observer) of 15 days after being opened and at least 5 days after each survey. This allowed the disruption created by the trail preparation and post survey to normalize allowing wildlife to redistribute themselves in space along the transect area in the total absence of observer disturbance (Peres, 1999). Transects were quietly surveyed at an average speed of 1 km/hour by an observer and a recorder, initiating at 0600 to 1100 hours; avoiding the survey of transects during heavy rainy days. Observers briefly stopped at very 100m in order to scan the forest for potential sightings and listened to sound cues. Data collected from each animal

sighting included: species, abundance, transect bearing, animal bearing, animal sighting distance and mode of detection.

**Data analysis**

Data was calculated per transect but analyzed by stratification. ANOVA tests were employed to compare species richness and abundance among the different strata; this was achieved using the statistical package InfoStats. Following Peres (2000) species were grouped into three body size classes: 1) small species (< 1 kg); 2) medium species (1 – 5 kg); 3) large species (> 5 kg). Game species were also grouped based on their conservation priority following the IUCN classification system. These categories were also employed in the ANOVA tests.

**RESULTS**

A total of 1,022 individuals representing 24 species (14 mammal and 10 bird species) were recorded during the 2013 surveys, while in 2016 a total of 1,184 individuals representing 25 game species (16 mammal and 9 bird species) were recorded. All but five (5) games species (4 mammal and 1 bird species) recorded in 2013 were also registered in 2016. Species recorded during the 2013 surveys only, were *Leopardus pardalis pardalis* (ocelot) and *Meleagris ocellata* (ocellated turkey); while *Dasypus novemcinctus mexicanus* (nine-banded armadillo), *Dicotyles pecari ringens* (white-lipped peccary) and *Agouti paca nelson* (gibnut) were recorded during the 2016 surveys only. *Sciurus deppei vivax* (Deppe's squirrel) and *Nasua narica* (coati) were most abundant in 2013 while *N. narica* (coati) and *Pecari tajacu* (collared peccary) were most common mammals during the 2016 surveys (Figure 2). Most abundant game bird species during the 2013 surveys were *Penelope purpurascens* (crested guan) and *Ramphastos sulfuratus* (keel-billed toucan); while *R. sulfuratus* and *Ortalis vetula* (plain chachalaca) were most common in 2016 (Figure 3). Overall game species mean abundance was not significantly different between 2013 and 2016 (f = 1.01; p = 0.3249).
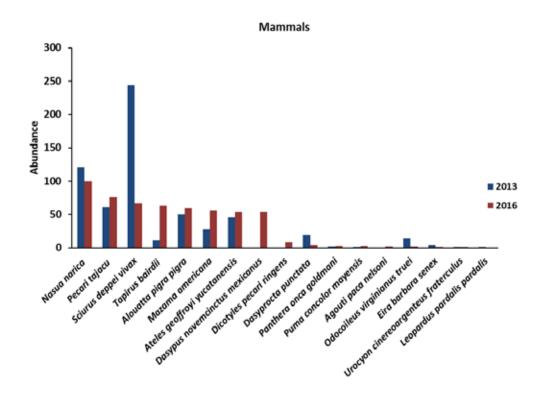
**Figure 2:** *Abundance of mammal game species recorded during the 2013 and 2016 game species surveys in the Chiquibul Forest.*
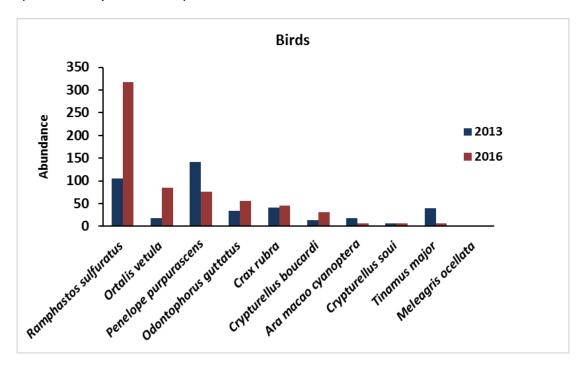


**Figure 3**: *Abundance of bird game species recorded during the 2013 and 2016 game species surveys in the Chiquibul Forest.*

There was no significant difference on the mean abundance of all game species combined at the different hunting pressures strata from 2013 to 2016 (High Hunting Pressure: F = 0.34, p = 0.5824; Medium Hunting Pressure: F = 3.67, p = 0.1039; Low Hunting Pressure: F = 0.76, p = 0.4162).  Results indicate that there was a decrease in global abundance of mammals during the 2016 surveys in the areas having a high and low hunting pressure (Figure 4). The greatest difference was that of a 33% decrease in abundance at the low hunting pressure areas but statistically the mean abundance was not significantly different (High Hunting Pressure: F = 0.13, p = 0.7277; Medium Hunting Pressure: F = 0.66, p = 0.4475; Low Hunting Pressure: F = 2.56, p = 0.1610).  For game bird species an increase in abundance was observed from the 2013 to 2016 surveys in areas with high hunting and medium hunting pressure (Figure 5). Areas with a medium hunting pressure recorded an increase of 99% from 2013 to 2016 and the mean abundance was significantly different (F = 5.21; p = 0.05).
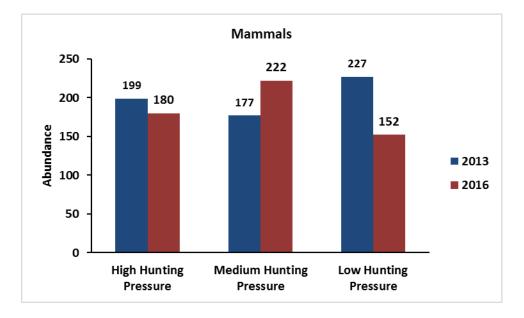


**Figure 4**: *Mammal abundance recorded by hunting pressure strata during the 2013 and 2016 game species surveys in the Chiquibul Forest.*
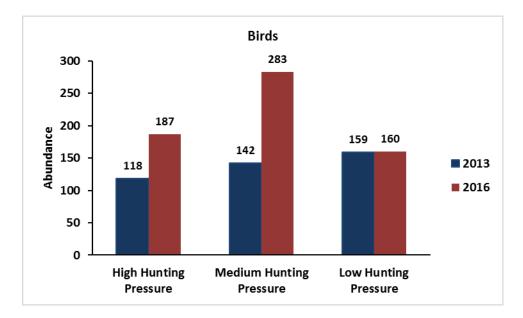
**Figure 5**: *Bird abundance recorded by hunting pressure strata during the 2013 and 2016 game species surveys in the Chiquibul Forest.*

The mean abundance of large bodied game species was significantly greater (F = 5.02, p = 0.0354) in 2016 (mean = 26.42 individuals) than in 2013 (mean = 17.58 individuals) while there was no significant difference in the mean abundance of medium (F = 1.00, p = 0.3273) and small (F= 2.72, p = 0.1133) game species from 2013 to 2016. Large bodied game species were less abundant than small bodied individuals at all hunting pressure zones (Figure 6). An increase in the overall observed abundance of large and small bodied game species was recorded at the high and medium hunting pressure zones from 2013 to 2016, while for the rest of the categories especially at the low hunting pressure zones a decrease in game species abundance was recorded from 2013 to 2016 (Figure 6).
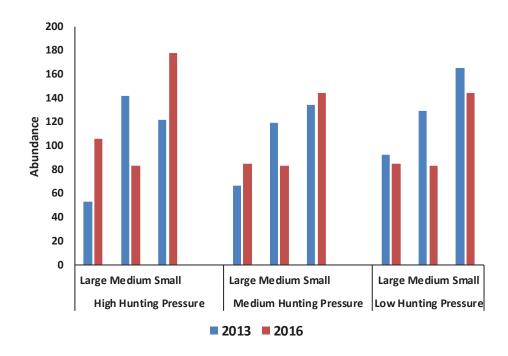
***Figure 6***: *Abundance of game species base on body weight recorded by hunting pressure strata during the 2013 and 2016 surveys in the Chiquibul Forest.*

There was no significant difference in the mean abundance of species of conservation value from 2013 to 2016 as well as the abundance of Least Concern game species (F = 0.69, p = 0.4153) but an increase in observed global abundance of game species which are vulnerable, near threatened and endangered (high conservation concern) was reported in 2016 compared to 2013, except for abundance recorded in the low hunting pressure zone which showed a decrease (Figure 7).
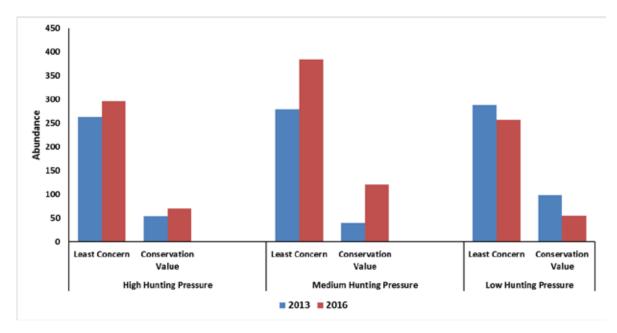
*Figure 7:* *Abundance of game species base on conservation value recorded by hunting pressure strata during the 2013 and 2016 surveys in the Chiquibul Forest*

## DISCUSSION AND CONCLUSION

Our findings indicate that there was no significant difference in the mean abundance of game species recorded in 2013 and that of 2016, but some species such as the Baird's Tapir and White-lipped Peccary were recorded more frequent during the 2016 surveys. In fact, the White-lipped Peccary was not recorded at all in 2013. The first recording of the species since 2006 in the Chiquibul Forest was made until 2014. A more interesting finding was the significant difference on mean abundance of large bodied game species which was greater in 2016 than in 2013. This is also explained by the greater frequency of Baird's Tapir and White-lipped Peccary observations made during 2016. Interestingly the White-lipped Peccary was recorded during two occasions on transects located in the high hunting pressure zones but also an incidental sighting of a drove of an estimated 80 individuals was sighted between the Raspaculo and Macal Rivers (low hunting pressure zone). The White-lipped Peccary is a very nomadic species, moving with shifting patterns of food availability; an important factor that may explain the recording of the species throughout the forest, as the species has also been recorded on the Maya Mountains Main Divide and around the Monkey Tail and Chiquibul River. Even though, mean abundance was greater in 2016 than in 2013, large bodied game was less abundant than small and medium size body species as was the case in 2013. This similarity of results reported by Peres (1990) and Mena *et al*. (2000) is a phenomenon commonly observed in natural ecosystems. This may go in line with the conclusions presented by Peres & Nascimento (2006) and Jerozolimski & Peres (2003) that hunters prefer large bodied game species over smaller ones, which is explained better by the Optimal Foraging Theory but will target smaller bodied species as large game is depleted. Some evidence of this was recorded in 2013 but not during the 2016 surveys. A reason why less hunting evidence was recorded in this most recent study may be attributed that at the present moment there has been observed a dramatic decrease in illegal xate extraction in the area plus the complete cessation of illegal logging. These activities are highly associated with illegal hunting. If such a scenario is maintained in the future, this will have a positive outcome in the overall abundance of game species and on their distribution patterns in the Chiquibul Forest.

Based on the hunting pressure stratification, no significant difference was recorded. This may be an indication that game species distribute evenly throughout the Chiquibul Forest, thus making it very difficult to isolate the true impact of illegal hunting on target species. It is only possible to infer that illegal hunting is happening throughout the Chiquibul Forest as evidence of this is frequently observed along trails and illegal camp sites. Even though results do not support the hypothesis that game species are less abundant closer to the Guatemala-Belize Western Border, it is believed that areas closer to farm clearings have less densities of game species; however; these areas are impossible to survey due to security reasons. Areas that were categorized as having high hunting pressure are at least 6 kilometers away from the border and the impact of illegal hunting on game species densities and abundance may be diffused at this point or game species from further forested areas migrate to these areas due to food

availability and if heavily hunted in these areas will have a huge impact in their abundance in more conserved areas.  This may be seen as areas closer to the border acting as 'sinks' while areas with low hunting pressure as the 'source'.  It becomes important to continue with a long term monitoring of game species abundance to determine if abundance is changing.

Many game species play a critical role in tropical forests serving as seed dispersers and or herbivores. Once these are absent it affects forest structure and composition (Stoner et al. 2007), through limited seed dispersal (Muller-Landau 2007), lowering germination success as seeds do not experience gut passage (Traveset & Verdu 2002), lowering seed predation (Corlett 2007) and finally, hunting alters the plant species composition and spatial distribution of the seedling and sapling layers (Nunez-Iturri & Howe 2007, Wright *et al*. 2007 & 2003).  Other authors such as Andresen & Laurance (2007) conclude that hunting indirectly affects both diversity and abundance of dung beetles due to a decrease in dung availability.  Dung beetles also play an important role as seed dispersers as they bury seed with the dung protecting seeds from granivores (Vulinec 2000).

Gregarious species such as White-nosed Coati, Ocellated Turkey, Spotted Wood Quail, Collared and White-lipped Peccary are more at risk of experiencing abundance declines than more solitary species like the Red-brocket deer, Crested Guan and Agouti, due to illegal hunting as hunters will take more than one individual if a drove is encountered.  Also the recordings of such sightings (of gregarious species) add variability to the results that may help to conceal any difference in game species abundance at hunting pressure strata and over time.

## RECOMMENDATIONS

Based on the results of this study the following is recommended:

- ✓ Continue with law enforcement activities to deter poachers and the illegal extraction of natural resources from the Chiquibul Forest.

- ✓ Be more vigilant to record and address illegal wildlife trafficking before it becomes prolific.

- ✓ Continue to monitor game species abundance in the Chiquibul Forest.

- ✓ Explore the viability of mounting GPS collars on White-lipped Peccary to study their local migration patterns in the Chiquibul Forest.

## REFERENCES

Andersen, E; Laurance, S. 2007. Possible indirect effects of mammal hunting and dung beetle assemblages in Panama. Biotropica 39:141-146.

Corlett, R. 2007. The impacts of hunting on the mammalian fauna of tropical Asian forests. Biotropica 39: 292-303.

Cornec, J. H. 2003. Geology Map of Belize

Dirzo, R; Mirada, A. 1991.Altered patterns of herbivory and diversity in the forest understory: a case study of the possible consequences of contemporary defaunation. In P.W Price, P.W. Lewinsohn, G. W. Fernandes and W.W. Benson. Eds. Plant-animal interactions: evolutionary ecology in tropical and temperate regions. New York. 272-287 p.

Jerozolimski, A; Peres, CA. 2003. Bringing home the biggest bacon: a cross-site analysis of the structure of hunter-kill profiles in Neotropical forest. Biological Conservation. 111:415-425.

Mena, VP; Stallings, JR; Regalado, BJ; Cueva, LR. 2000. The sustainability of current hunting practices by the Huaorani. *In*: Robinson, JG; Bennett, EL. (eds), Hunting for Subsistence in Tropical Forests. Columbia University Press, New York, pp. 57–78

Mittermeier, RA. 1987.  Effects of hunting on rain forest primates. Primate conservation in the Tropical Forests 109-146.

Muller-Landau, HC. 2007. Predicting the long-term effects of hunting on plant species composition and diversity in Tropical Forests. Biotropica 39(3): 372-384.

Nunez-Iturri, G; Howe, HF. 2007. Bushmeat and the fate of trees with seeds dispersed by large primates in a lowland rainforest in western Amazonia. Biotropica 39: 348-354.

Peres, CA. 1990. Effects of hunting on Western Amazonian primate communities. Biological Conservation. 54:47-59.

Peres, C; van Roosmalen, MGM. 1996.  Avian dispersal of mimetic sedes in *Ormosta lignivalvis* (Leguminosae: Papilionaceae): deceit or mutualism? Oikos 75: 249-258

Peres CA. 1999. General Guidelines for standardizing line-transect surveys of tropical forest primates. Neotropical Primates 7(1): 11-16

Peres, C. 2000. Effects of subsistence hunting on vertebrate community sutructure in the Amazonian Forest.  Conservation Biology 14(1): 240-253.

Peres, CA. 2001. Synergistic effects of subsistence hunting and habitat fragmentation on Amazonian Forest Vertebrates. Conservation Biology 15(6): 1490-1505.

Peres, CA; Nascimento, HS. 2006. Impact of game hunting by the Kayapo of south-eastern Amazonia: implications for wildlife conservation in tropical forest indigenous reserves. Biodiversity and Conservation. 15: 2627-2653.

Redford, KH. 1992. The empty forest: Many large animals are already ecologically extinct in vast areas of neotropical forest where the vegetation still appears intact. BioScience 42(6): 412-422

Salas, O; Meerman, JC. 2008. Chiquibul National Park Management Plan 2008-2013. pp. 191.

Silveira, L; Jacomo, ATA; Diniz-Filho, JAF. 2003. Camara trap, line transect census and track sureys: a comparative evaluation. Biological conservation 114: 351-355

Stoner, KS; Vulinec, K; Wright, SJ; Peres, CA. 2007. Hunting and plant community dynamics in Tropical Forests and future directions. Biotropica 39(3): 392-2007.

Thiollay, JM. 1986. Structure compare du peuplement avien dans trios sites de foret primaire en Guyane. Rebue d'Ecologie. 41: 59-105.

Traveset, A; Verdu, M. 2002. A meta-analysis of the effect of gut treatment on seed germination. *In* Levey, DJ; Silva, WR, Galetti, M. (eds). Seed dispersal and frugivory: Ecologu, evolution and conservation. CABI International, Wallingford, UK. pp. 339-350

Vulinec, K. 2000. Dung beetles (Coleoptera: Scarabaidae), monkeys, and conservation in Amazonia. Florida Entomology 83: 229-241.

Wright, ACS; Romney, DH; Arbuckle, RH; Vial, VE. 1959. Land in British Honduras. Colonial Res. Publication No. 24.

Wright, SJ. 2003. The myriad consequences of hunting of vertebrates and plants in tropical forests. Perspectives of Plant Ecology Evolution and Systematics. 6: 73-86.

Wright, SJ, Hernandez, A; Condit, R. 2007. The bush meat harvest alters seedling banks by favoring lianas, large seeds and seed dispersed by bats, birds, and wind. Biotropica 39: 363-371.

Wyatt, JL; Silman, MR. 2004. Distance-dependent in two Amazonian palms: Effects of spatial and temporal variation in seed predator communities. Oecologia 140: 26-35.