Indian Forester, 143 (10) : 1055-1063, 2017
http://www.indianforester.co.in

ISSN No. 0019-4816 (Print)
ISSN No. 2321-094X (Online)

IMPACT OF BUSHMEAT HUNTING ON WILD ANIMAL'S BIOMASS IN THE WESTERN INDIAN HIMALAYA

HILALUDDIN

FRI, Campus, P.O.: New Forest, Dehra Dun (Uttarakhand)
E-mail: hilaluddin@yahoo.com

ABSTRACT

The impacts of bushmeat hunting on wild animal biomass were examined in the western Indian Himalaya. Ungulates and pheasants metabolic biomass were generally higher in protected sites as compared to hunted sites. In protected site, ungulates had the greatest metabolic biomass whereas pheasants the least. Koklass pheasant (*Pucrasia macrolopha*) and Kaleej pheasant (*Lophura leucomelanos*) have shown statistically significant variations for their biomass in hunted and protected sites, whereas densities of Cheer pheasant (*Catreus wallichi*) and Monal pheasant (*Lophophorus impejanus*) did not show statistically significant differences between the two management units. The biomass of Barking deer (*Muntiacus muntijak*), Himalayan tahr (*Hemitragus jemlahicus*) and Serow (*Capricornis sumatraensis*) were significantly higher in protected site as compared to hunted site, whereas Goral (*Nemorhaedus goral*) biomass between the two types of forest patches statistically remained similar.

Key words: Wildmeat, Himalaya, Animal biomass, Ungulates, Pheasants and Impact.

Introduction

Hunting is the prime suspect in the global extinction of many species (Martin and Steadman 1999) and is posing a major threat to populations of hundreds of species worldwide (Diamond and Case, 1986; Redford, 1992; Reid, 1992; Peres and Terborg, 1995; Alvard *et al*., 1997; Bodmer *et al.,* 1997; Hill *et al*., 1997; Wilkie and Carpenter, 1999; Robinson and Bennett, 2000; Mace and Balmford, 2000; Bakkar *et al.,* 2001; Cullen *et al.,* 2000 and 2001; Madhusudan and Karanth, 2002; Rosser and Mainka, 2002; Brashares *et al.,* 2004; Kaul *et al*., 2004; Hilaluddin *et al*., 2005a, 2005b and 2006). However, hunting is apparently sustainable in some areas, either because vulnerable species have already been extirpated (Cowlishaw *et al.,* 2005) or because hunting pressure remains low (Hill and Padwe, 2000). Most efforts to investigate impact of wild animal extractions and its ecological consequences on native wildlife and forest structure so far have been primarily from Africa and Latin America and such information from other terrestrial regions of the world largely remained fragmentary and almost lacking in Asia. This information is increasingly important for those areas where habitat loss is leading to decreasing populations of species that are increasingly fragmented. One such area is Himalaya, which is of global importance for the conservation of biological diversity (ICBP, 1992; Olson and Dinerstein, 1998).

Much of the Himalaya falls within Indian territory, forms 2.4% of the earth's landmass and supports 16% of the world's human population (Anon*.,* 2000). Nineteen per cent of its landmass is under forest (FSI, 2005). However, few areas remain isolated because of the extremely dense human population. Strict enforcement of Indian Wildlife Protection Act 1972 has apparently succeeded in curbing open trade of wild animals and their parts in the western Indian Himalaya, but hunting of animals both for subsistence use and commercial gains continues here (Kaul *et al.,* 2003 and 2004; Hilaluddin and Naqash, 2006). In the western Indian Himalaya (as in Chamba district), hunting activities are mostly outside national parks and wildlife sanctuaries (Kaul *et al.,* 2003). Unlike eastern Indian Himalaya and northeast India where hunting is more evenly distributed among all groups of mammals and birds (Hilaluddin *et al.,* 2005a, 2005b and 2006), hunting concentrates exclusively on large mammals and galliformes in this landscape of the world (Kaul *et al.,* 2003 and 2004). The method of hunting varies from snaring to use of modern firearms. There exists lack of effective community rules (e.g. restrictions on game extractions in terms of protection to sensitive and globally threatened species, breeding seasons, age-sex classes and bag limits) with regard to hunting. Further, people are increasingly switching over to modern hunting devices (e.g. guns) at the cost of traditional ones. The harvest of wildlife is reaching devastating levels due to burgeoning human population, increasing access to forests and use of increasingly efficient hunting technologies. Three major types of hunting activities are prevalent in the western

Ungulates and pheasants metabolic biomass were generally higher in protected sites as compared to hunted sites.

Indian Himalaya. (1) Organized hunting that targeted large bodied species of those with specific market; (2) regular snaring targeted for galliformes in the vicinity of villages, primarily for providing food for family; and (3) opportunistic hunting trips into the forests, mainly for subsistence requirements.

Previous studies (Kaul *et al.,* 2003 and 2004; Hilaluddin and Naqash, 2006) documented extraction levels and patterns of large mammals and galliformes, methods of hunting, and reasons of hunting from this landscape of the world. It, however, remained unclear whether offtake is adversely affecting wild populations of these animal groups. Therefore, the present study is designed to investigate impact(s) of pheasant and ungulate's offtake on their wild populations in and around Chamba district of the western Indian Himalaya.

Study area

Chamba is located in Himachal Pradesh, falls within Indian's bio-geographic province “2B Western Himalaya” (Rodgers and Panwar, 1988) and forms the part of “Western Himalaya Endemic Bird Area” (ICBP, 1992; Satterfield *et al.,* 1998). Evergreen Temperate Pine Forests dominated by Chir pine (*Pinus roxburgii*), Evergreen Temperate Oak Forests dominated by Ban oak (*Quercus leucotrichophora*) and Mixed Evergreen Temperate Forests with extensive Southwest facing grasslands occur in Chamba (Champion and Seth, 1968). The associates of Ban oak and Chir pine are *Rhododendron arboreum*, *Cedrus deodara, Pinus wallichiana, Taxus baccata* and *Abies pindrow*. The undergrowth is predominated by *Berberis* sp. and *Rubus* sp. with some *Rosa* sp., *Daphanae* sp., *Myrsine* sp. and *Rhabdosia* sp.

These vegetation communities in Chamba district support over 200 bird species, including restricted range Red-browed Finch (*Callacanthis burtoni*) and globally threatened Cheer pheasant (*Catreus wallichii*) and Western tragopan (*Tragopan melanocephalus*) (BI, 2004). Several globally threatened mammals, including Brown bear (*Ursus arctos*), Himalayan black bear (*Selenarctos thibetanus*), Himalayan tahr (*Hemitragus jemliahicus*), Indian leopard (*Panthera pardus*), Musk deer (*Moschus chrysogaster*), Rhesus macaque (*Macaca mulatta*), Serow (*Capricornis sumatraensis*) and Snow leopard (*Uncia uncia*) (IUCN, 2004) are also known from the forest areas of Chamba.

In the western Indian Himalaya (as in Chamba), religious ceremonies usually involve slaughtering of domestic livestock and unlike the northeast India, wild animals are not killed. The staple food is mainly cereal and vegetable-based, and therefore consumption of wildmeat probably provides a supplementary source of animal protein. Livestock (usually goat) are mainly raised for sale so that domestic consumption of livestock is usually limited to religious ceremonies or if the animal gets incapacitated. Thus, to supplement the animal protein intake, wildmeat is consumed, either because it is available cheaper than meat of domestic animals or it can be harvested free from the forests. Hunting is rarely considered a full time profession with most practitioners does hunting in their spare time only. A good number of young people depends on part-time jobs for their livelihood, which can range from road works, porters/guides for trekkers to harvesting medicinal plants and herbs or ringal, etc. Therefore, they have ample spare time at their disposal, which they utilize suitably to their benefit by indulging in hunting.

Methods

Animal census

Five forest fragments were studied in and adjoining Chamba district in between 1410 and 3290 meters MSL during summer 2006. Forest fragments here are defined as continuous block of forests surrounded by agriculture fields and human settlements. The greatest distance between sites was less than 150 km. The quantitative hunting pressures were not recorded during the course of the present study. As an alternative, sites were selected based on already documented animal extraction rates and patterns in the forest fragments of the western Indian Himalaya in literature (e.g. Kaul *et al.,* 2003 and 2004; Hilaluddin and Naqash, 2006) and focal discussions held with the hunters. These measures later were used to define sites as the protected and the hunted. The protected sites were located in Khajjayar-Kalatop and Kugti Wildlife Sanctuaries, whereas the hunted sites were in Chamba and Kishtwar Territorial Forest Divisions (Table 1).

Densities of species were estimated using Belt Transect (pre-defined areas) surveys following sample count strategy (Sutherland, 1996). Transects were identified on topo maps after discussions held with the concerned wildlife officials and local hunters. These transect were re-identified on ground during reconnaissance surveys of the sites for their ground verification. The starting and ending points of each transect was permanently marked on trees with the help of paint for future references. The length of transects were measured using Hip Chain Method (Chaturvedi and Khanna, 1982). These transects passed through all major vegetation communities occurring in the two management units. Transects were established at considerable distances from each other for avoiding

Table 1: Profiles of the sampled transects.

Transect name	Nearest village	Beat name	Forest division name	Transect characteristics		
				Length (km)	Area (km^2)	Altitude (MSL)
Burnar-Grad	Tyari	20 A[H]	Kishtwar TRL	5.0	1.5	2600-3075 m
Top-Grad	-do-	20 B[H]	-do-	4.5	1.35	2890-3290 m
Top-Pangi	Ishtiyari	19 A[H]	Chamba and Kishtwar TRL	5.0	1.5	2600-3232 m
Namba Naal	Suendi	Sarah[H]	Chamba TRL	4.5	1.35	2740-3210 m
Dramni Ki Bhurjee	-do-	-do-	-do-	3.7	1.11	2285-2440 m
Grad Bah	-do-	-do-	-do-	3.0	0.9	2020-2539 m
Haath Pav	Lagga	Kiri[H]	-do-	4.5	1.35	1470-1820 m
Sukha Naala	-do-	-do-	-do-	4.5	2.02	1410-2400 m
Kalatop-Lakarmandi	Kalatop	Kalatop[P]	Chamba WL	3.0	1.8	2590-2860 m
Kalatop-Khajrot Nala	-do-	Lakar Mandi[P]	-do-	6.0	1.8	2040-2860 m
Khajrot Nala-Khajjyar	Khajjyar	Khajrot[P]	-do-	5.0	1.5	1610-2040 m
Sunil Lodge-Kuringarh	Rakh	Krangda[P]	-do-	4.0	1.2	1400-1800 m
Kalatop-RFC9	Lakar Mandi	Talai[P]	-do-	5.0	1.5	1820-2480 m
RFC 11-15	-do-	Kalatop[P]	-do-	4.0	1.2	1400-1860 m
Kangroo DPF	Kugti	Lower Kugti[P]	Bharmor WL	4.0	1.2	2190-2840 m
Karog Dhar	-do-	Upper Kugti[P]	-do-	5.0	1.5	2632-3210m

[P]protected site and [H]hunted site. While TRL denotes Territorial Forest Division, WL means Wildlife Division.

double counts of animals. The minimum distance between the two transects at a site was greater than 1km. Generally, existing streams and prominent trails were utilized for sampling areas for visibility. Although, it may cause bias in estimating natural densities of the species as there may be unconscious bias in route selection if landscapes differ, even subtly. However, vegetation characteristics of the hunted and the protected forests did not show statistically significant variations, at least on the basis of the vegetation heterogeneity measured as part of present study (Table 3). Thus, similarities in vegetation characteristics between the hunted and the protected sites will outweigh over variations in animal densities, if any, as a consequence of landscape heterogeneity across the two management units.

Each transect was walked by a team of trained observers who scanned them for pheasants and ungulates on regular daily basis, at least for three consecutive days. The animal counts in all transect within an area were started simultaneously at the sunrise and ended between 830 and 1000 hours depending upon the length of transect. The same census schedule was followed on consecutive days. Walking slowly (approximately 1-1.5 km/ hour) and stopping briefly at every 50-100 meters interval (Emmons, 1984) with the intention of flushing animals did censuses. This helped in avoidance of missing animal sightings to some extent. The teams of observers involved in animal census exercise were trained in identification of all likely occurring species of galliformes and ungulates in the study area prior to commencement of census. Observers were also trained in individual species identification through local names and walking across transect at fixed distances from each other for animal search. A team of 5 observers searched each transect and each observer scanned 10-15 meters area on his both sides depending upon the terrain and visibility. Thus, observers maintained a fixed distance of 20-30 meters from each other during search. They also maintained as much silence as they could during transect walk so that all animals within transect could stay back until detected by them. Nawaz *et al.* (2000) used Belt Drive Count method for estimating pheasant populations in Pakistan Himalaya in winter and recommended use of this method for estimating populations of pheasants elsewhere in the western Indian Himalaya. They, however, documented limitations of this method in extremely steep and rocky terrain covered with snow and suggested that belt transects should be walked downward rather than upward in the hills for detecting all animals within transect as a result of its full view. Similar, steep and rocky terrain existed in our study area and therefore downward pilot surveys were conducted. However, the study area hardly received snowfall during summer and consequently its peaks were free of snow during the course of present study.

The total number of animals seen, time of their sighting, direction of their movements and their activities were recorded. Recording the time of sighting and direction of animal movement made it possible to reduce total count to account for those individuals evidently seen more than once by two different observers.

Vegetation survey

The composition of trees, shrubs and herbs within each belt transect was also assessed. For the purpose, 5 sample points along each belt transect were selected at 500m regular distances on 15m either side of transect in order to avoid relatively disturbed vegetation in the form of trampling by cattle and human. The circular plots of various sizes were laid at each sample point to quantify abundance of perennial woody species (tree and shrub) populations, whereas abundance of annual herbaceous vegetation (herbs) was enumerated in square plots. At each sample point 10m and 3m radius circular plots were established for estimating populations of trees (greater than 31 cm in basal girth) and shrubs, respectively, whereas 1x1m^2 quadrates for quantifying populations of herbaceous vegetation.

In addition, vegetation structure specifically canopy covers of trees, shrubs and herbs were also measured at each sample point. While Grid Mirror Method was adopted to quantify canopy cover of tree species, Line Intercept Method and Crown Diameter Method (Muller-Dombois and Ellenberg, 1974) were used to estimate crown cover of herbs and shrubs, respectively.

Data analysis

Animal densities per unit area at a given day were calculated as the total number of individual of a species seen on a particular transects on a particular day divided by the total area of that transect. A non-parametric Man-Whitney U test was used to compare densities of each animal species in the protected sites with their corresponding densities in the hunted sites to investigate impact(s) of hunting on their populations.

It was assumed that there might be differences in the vegetation structure and composition between the hunted and the protected sites independent of hunting pressure, although all of the patches were once part of the same continuous forest and are of the same geological origin. Therefore, vegetation structural and compositional heterogeneity in the hunted site were statistically compared with their corresponding values in the protected sites using Man-Whitney U test. While vegetation densities of each plant biomorph *i.e.* plant life-form (tree, shrub and herb) at each sample unit was calculated following Curtis and McItonish (1950), their general diversities (H') were computed in accordance to Shannon-Wiener (1963). Species richness of each plant bio-morph was calculated as total number of species of that life-form occurring in a sample unit (Ludwig and Reynolds, 1988). All statistical tests were performed following Sokal and Rohlf (1995).

Crude biomass was calculated for each animal species using the average body weight (BW) of adult individual multiplied by the estimated densities (D) of individuals as (BW x D kg/ Km^2). Mean body weights of individual species were taken from literature (Ali and Ripley, 1987; Prater, 1971). Metabolic biomass was also calculated for each species using average body weight of the adult individual raised to the 0.75 power and multiplied by the estimated individual densities (BW$^{0.75}$ x D kg/Km^2). The relative contribution of a species to the total animal biomass in hunted and protected sites was calculated in percentage (%).

All statistical tests were performed following Sokal and Rohlf (1995). Means ± quartile and percentile values are presented throughout.

Results

Sample size and survey efforts

A total of 16 belt transects (eight each in protected and hunted sites) (Table 1) were actively scanned for searching pheasants and ungulates as part of present study. The census party traveled a total of 212.1 km (108 km in the protected site and rest in the hunted site) distance in 157.48 hr (77.11 hr in the protected site and rest in the hunted site) in all transects for the active search of animals during three consecutive days animal population estimation exercise. All species were observed on more than 5 occasions (Table 3), ranging from 5 for Serow (*Capricornis sumatraensis*) to 91 for Koklass pheasant (*Pucrasia macrolopha*).

The survey team spent a mean of 9.65 hours/ transect $\pm$0.69 CI in the hunted site and an average of 10.01 hours/transect $\pm$0.88 CI in the protected site for active search of animals.

Vegetation structure and composition

With the exception of statistically significant higher shrub densities in the protected site as compared to the hunted site, rest of the vegetation characteristics between the two management units showed statistically non-significant differences (Table 2). Densities, diversities, richness and cover values of trees, shrubs and herbs were generally higher in the protected site as compared with their corresponding values in the hunted site.

Animal biomass

Crude biomass and metabolic biomass of animals varied between sites (Table 3). The protected site had the greatest crude biomass of ungulates (92.99%), whereas pheasants (7.01%) the least. In protected site, goral had the highest crude biomass, whereas Serow the least. Amongst pheasant, Koklass pheasant dominated the

Table 2: Vegetation characteristics (mean ± median) with statistical variations in the hunted and the protected sites.

Plant bio-morph type	Vegetation structural and compositional variables	Hunted site	Protected site	Statistical values	
				Mann-Whitney U test	P
Tree	Density (# of plants/km^2)	307.4 ± 238.8	332.8 ± 302.54	$U_{78} = 721.5$	0.45
	Diversity (H')	0.6 ± 0.56	0.6 ± 0.56	$U_{78} = 797.5$	0.98
	Richness (No.)	2.3 ± 2.0	2.3 ± 2.5	$U_{78} = 765.5$	0.73
	Tree cover (%)	36.0 ± 30.0	30.1 ± 30.0	$U_{78} = 731.0$	0.5
Shrub	Density (# of plants/km^2)	2226.4 ± 1783.45	4024.5 ± 2547.8	$U_{78} = 570.5$	0.03*
	Diversity (H')	0.5 ± 0.5	0.7 ± 0.68	$U_{78} = 664.5$	0.19
	Richness (No.)	2.2 ± 2.0	2.9 ± 2.9	$U_{78} = 663.0$	0.18
	Shrub cover (%)	29.7 ± 25.0	24.1 ± 20.0	$U_{78} = 730.0$	0.5
Herb	Density (# of plants/km^2)	35540.0 ± 30500.0	39582.0 ± 32200.0	$U_{78} = 751.5$	0.64
	Diversity (H')	1.01 ± 1.01	1.3 ± 1.21	$U_{78} = 603.5$	0.06
	Richness (No.)	4.6 ± 4.0	4.8 ± 4.0	$U_{78} = 672.0$	0.21
	Herb cover (%)	37.65 ± 30.0	31.3 ± 20.0	$U_{78} = 735.0$	0.53

* Denotes significant values.

Table 3: Crude and metabolic biomass of animals in the hunted and protected sites in the western Indian Himalaya. Values are presented as mean median with statistical variations throughout

Species		Mean body wt in kg	Crude biomass				Metabolic biomass					
Vernacular name	Latin name		Hunted site		Protected site		Hunted site		Protected site		Statistical values	
			Kg/km^2	%	Kg/km^2	%	Kg/km^2	%	Kg/km^2	%	Mann-Whitney U test	P
Barking deer (10)	*Muntiacus muntjak*	22.5 (10.33)	73.8	5.69	168.75	5.56	33.88	5.96	77.48	6.07	$U_{14} = 10.1$	0.01*
Cheer pheasant (8)	*Catreus wallichi*	1.25 (1.18)	12.5	0.96	15.0	0.49	11.8	2.07	14.16	1.11	$U_{14} = 29.0$	0.6
Goral (49)	*Nemorhaedus goral*	27.5 (12.01)	715.55	55.2	1272.43	41.93	312.5	54.96	555.7	43.54	$U_{14} = 11.5$	0.03
Himalayan tahr (9)	*Hemitragus jemlahicus*	90.0 (29.22)	342.9	26.5	1149.3	37.87	111.33	19.58	373.14	29.24	$U_{14} = 12.0$	0.03*
Kaleej pheasant (23)	*Lophura leucomelanos*	1.1 (1.07)	17.31	1.34	45.97	1.51	16.84	2.96	44.72	3.50	$U_{14} = 4.0$	0.003*
Koklass pheasant (91)	*Pucrasia macrolopha*	1.14 (1.1)	42.5	3.28	84.79	2.79	41.1	7.23	81.82	6.41	$U_{14} = 8.0$	0.01*
Monal pheasant (28)	*Lophophorus impejanus*	2.4 (1.92)	24.2	1.87	67.0	2.21	19.53	3.43	54.05	4.23	$U_{14} = 18.0$	0.1
Serow (5)	*Capricornis sumatraensis*	90.0 (29.22)	66.6	5.14	231.3	7.62	21.62	3.80	75.1	5.88	$U_{14} = 15.5$	0.05*
Total biomass			1295.36		3034.54		568.6		1276.17			
Total biomass of pheasants			96.51		212.76		89.27		194.75			
Total biomass of ungulates			1198.85		2821.78		479.33		1081.41			

NB: Values in parentheses in column 1 are number of times a species was observed, whereas in column 2 are calculated metabolic body weights in kg. * denotes significant values whereas values in parentheses depicted in column 4 are quartiles. Values

crude biomass both in protected and hunted sites, whereas Cheer pheasant the least. Similarly, the hunted site had greatest crude biomass of ungulates (84.74%), whereas pheasants (12.26%) the least. Like protected site, Goral had highest crude biomass in hunted site amongst ungulates, whereas Barking deer the least.

Ungulates had the greatest metabolic biomass (84.30%) in protected site, whereas pheasants (15.70%) the least. Amongst ungulates, Goral contributed maximum into biomass spectrum both in hunted and protected sites, whereas Serow the least. Koklass pheasant dominated metabolic biomass spectrum both in hunted and protected sites, whereas Cheer pheasant contributed minimum into metabolic biomass spectrums of protected and hunted sites.

Discussion

According to Cullen *et al.* (2000), three types of sources of variations might affect animal abundance among different forest patches. First, vegetation structure and composition, which independent of hunting may cause changes in species abundance between the two sites. Secondly, hunting may affect the behaviour of certain species and make them less easy to sight than non-hunted species. Animals in protected site may be less wary and therefore easier to sight than the same species in hunted site. Thirdly, small sample sizes may give skewed values and distorted actual differences.

The comparisons of vegetation characteristics in hunted and protected sites in the study area showed that hunting pressures within the two forest types were independent of vegetation structure and composition. Moreover, the studied forest fragments were once part of the same continuous forest, and are of the same geological origin. Also, habitat heterogeneity does not lead to gross differences in wildlife densities, specifically for species with large geographical ranges and broad ecological tolerances (Eisenberg, 1989; Emmons, 1990; Redford and Eisenberg, 1992; Cullen *et al.,* 2000). Many of the species examined as part of the present study have large geographical ranges and therefore local differences in habitat condition among the forest patches are likely to override differences in quantitative hunting rates of the wildlife species among the forest fragments.

In the present census exercise, forest fragments in hunted and protected sites were scanned for targeted animal species with their total numbers by a team comprising of wildlife staff, hunters and nomadic graziers. Each member searched for animals within 10 meters area on his each side. It was quite unlikely to miss animals in such a narrow strip during the combing operation and thus animal densities between the two management unit are unlikely to get affected due to changed behaviour (more wary in protected site) of animals, if any.

In the present study, a pre-defined area 21.72 km^2 (11.7 km^2 area in protected site and rest in hunted site) within 16 belt-transects/blocks (large sample size) was actively scanned for searching populations of targeted animal groups. Further, all data was normalized prior to parametric statistical analysis and therefore distortions in actual variations in animal densities due to low sample sizes, if any, in the two forest types are unlikely to occur. However, current census results produced density estimates with SE up to around 50% of mean values for certain species, specifically Cheer pheasant, Himalayan tahr and Serow. The impression of the data for these species may be the result of their tendencies to aggregate in extremely steppe areas with specialized habitat conditions. This will inevitably cause animal encounters to very more widely than their populations were more evenly dispersed. Further, there are probably few significant populations of these species left anywhere in their respective ranges across the Indian Himalaya and therefore they were missing from many transects/blocks in the study area as well. This factor, however, remained uniform across the transects/blocks both in hunted and protected sites and therefore made possible viable comparisons of the species abundance across the two management units.

The current study based on a clear set of hunting criteria and using large sample size clearly shows that hunting have drastic effect on the populations of hunted species (Kaul *et al.*, 2003, 2004) in the western Indian Himalaya. Not surprisingly, the hunting has clearly inverted the relative contribution of species to metabolic biomass, especially when animals are grouped into two broad categories viz. terrestrial (pheasants and ungulates) and arboreal (primates). This implies that arboreal animal species dominate the metabolic biomass or relative energy consumption at hunted sites, whereas terrestrial comprise the bulk of the metabolic biomass at protected site. The forests of western Indian Himalaya have much reduced in the area and are increasingly fragmented as a result of logging (FSI, 2005). Against a back grow scenario of burgeoning human population (0.07 ha per capita) in this landscape of the world (Anon, 2000) and presumably accelerated protein demand thereof, the survival prospects for game species in hunted areas is less certain than those in protected areas. In addition, increasing adoption of modern hunting devices and lack of effective community rules will further exacerbate the situation. This accentuated by logging, and agriculture and road network expansion may result in local extinctions of certain game species from several un-protected forest areas. One such example is recent local extinction of Western tragopan (*Tragopan melanocephalus*) from the forest of Kiri Beat under Lower Chamba Range.

The forest patches subjected to hunting undergo for significant changes in vegetation structure and composition due to poor pollination and seed dispersal of dependent plant species (Cullen *et al.,* 2000). This is because changes in vegetation structure and composition may have adverse affect on structure and composition of dependent animal communities as documented in birds in many forest ecosystems across the world (Thiollay, 1999; Raman and Sukumar, 2002; Skowno and Bond, 2003). This ultimately lead to ecological extinction of species both in marine and terrestrial ecosystems (Conner, 1998; Dayton *et al.,* 1998; Estes *et al.,* 1998; Novaro *et al.,* 2000; Redford and Feinsinger, 2001). However, how hunting is affecting the population dynamics of game species are sorely lacking for Asia in general, and India in particular. Further, no information exists on population age structures and demographics of hunted versus protected sites, impact of hunting on animals of different age classes, impact of hunting on vegetation characteristics and demographics of plant populations in hunted versus protected sites and ecological sustainability of wildmeat extractions from this region of the world. These require immediate investigations.

The impact of hunting on animal populations in the western Indian Himalaya is similar to other studies conducted elsewhere across the world. For example, mammal abundance, body mass and population densities in protected and hunted sites in Makokau, Gabon were negatively co-related with impact on species (Lahm, 1993). Similarly, mean body mass of all targeted species was significantly reduced in response to hunting pressures in Amazonian forest patches (Peres, 1999a, 1999b). The animal abundance and crude and metabolic biomass greatly declined in the hunted sites in Atlantic forest patches (Cullen *et al.,* 2000, 2001). Densities of large mammalian species also showed significant decline in response to hunting in Nagarhole, India (Madhusudan and Karanth, 2002). However, observed pattern of current study is contrary to Peres (2000), who showed that overall game densities in hunted and protected sites did not differ significantly, although he observed a reduction in vertebrate game in over-harvesting areas of Amazonian forests. Biomass of small and medium-bodied species greatly increased as a proportion of the overall community, whereas that of the largest body class was significantly depressed at moderately to heavily hunted sites as a result of selective hunting of large-bodied mammals in these forest fragments.

Acknowledgements

The staff of the Wildlife Wing of Himachal Pradesh actively participated in wild animal census. The author thanks the migratory graziers and the hunters for actively participating in the census exercise. Last but not the least, this study was funded by a collection of small grants, one each from Wildlife Conservation Society through Centre for Wildlife Studies-India and Oriental Bird Club, UK. The author is highly indebted to them.

पश्चिमी भारतीय हिमालयों में जंगली पशु की जैवमात्रा में बुशमीट शिकार का प्रभाव

हिलालुद्दीन

सारांश

पश्चिमी भारतीय हिमालय में जंगली पशु जैवमात्रा में बुशमीट शिकार के प्रभाव की जांच की गई। खुरदार एवं फेजेण्ट चयापचयी जैवमात्रा शिकार किए गए स्थलों की तुलना में संरक्षित स्थलों में सामान्यत: उच्च थी। संरक्षित स्थल में खुरदार की उच्चतम चयापचयी जैवमात्रा थी जबकि फेजेण्ट की न्यूनतम थी। कोकलास फेजेण्ट (*पुक्रेसिया मैक्रोलोफा*) और कालीज फेजेण्ट (*लोफूरा ल्यूकोमीलेनोज*) ने शिकार किए गए एवं संरक्षित स्थलों पर अपनी जैवमात्रा के लिए सांख्यिकीय रूप से महत्वपूर्ण विभिन्ता को दर्शाया जबकि चीड़ फेजेण्ट (*केट्रीयस वालिचि*) और मोनाल फेजेण्ट (*लोफोफोरस इम्पीजेनस*) के घनत्वों ने दोनों प्रंबधित इकाइयों के बीच सांख्यिकीय रूप से महत्वपूर्ण विभिन्नता को नही दर्शाया। काकड़ (*मुन्टिएकस मुन्टिजेक*), हिमालयी टहर (*हीमिट्रेगस जीम्लेहिकस*) और सेराव (*कैप्रिकॉर्निस सुमेट्रेइन्सिस*) की जैवमात्रा शिकार किए गए स्थल की तुलना में संरक्षित स्थल में महत्वपूर्ण रूप से उच्च थी जबकि वन खण्डों की दो किस्मों के बीच गोराल (*नीमोरहेइडस गोराल*) जैवमात्रा सांख्यिकीय रूप से समान रही।

References

Alvard M. Robinson J.G., Redford K.H. and Kaplan H. (1997). The sustainability of subsistence hunting in the Neotropics. *Conservation Biology*, 11: 977-982.

Ali S. and Ripley S.D. (1987). *A Handbook of Birds of India and Pakistan: compact edition*. Oxford University Press, New Delhi, India.

Anon. (2000). *Population and Forests: a report on India*, 2000. United Nations Population Fund, New Delhi, India.

Bakkar M.L., da Fonseca G.A.B., Mittermeier R., Rylands A.B. and Paenemilla K.W. (2001). *Hunting and bushmeat utilization in the African rain forests: prospective towards a blueprint for conservation action*. Advances in Applied Biodiversity Science *2*. Conservation International, Washington, D.C., U.S.A.

BI. (2004). *State of the world's Birds: Indicators of Our Changing World*. Birdlife International, Cambridge, U.K.

Bodmer R.E., Eisenberg J. and Redford K. (1997). Hunting and likelihood extinction of Amazonian mammals. *Conservation Biology,* 11: 460 – 466.

Brashares J.S., Arcese P.S., Moses K., Coppolillo P.B., Sinclair A.R.E. and Balmford A. (2004). Bushmeat hunting, wildlife declines and fish supply in West Africa. *Science*, 306: 1180-1183.

Champion H.G. and Seth S.K. (1968). *The forests types of India*. Government of India Press, New Delhi, India.

Chaturvedi A.N. and Khanna L.S. (1982). *Forest Mansuration*. International Book Distributors, Dehradun, Uttaranchal, India.

Conner R.N. (1998). Wildlife populations: minimally viable or ecologically functional? *Wildlife Society Bulletin*, 16: 80-84.

Cowlishaw G.C., Mendelson S. and Rowcliffe J.M. (2005). Structure and composition in a bushmeat commodity chain Southwestern, Ghana. *Conservation Biology*, 19: 139-149.

Cullen L.Jr., Bodmer R.E. and Padua C.V. (2000). Effects of hunting in habitat fragments of the Atlantic forests. *Biological Conservation*, 95: 49-56.

Cullen L.Jr., Bodmer R.E. and Padua C.V. (2001). Ecological consequences of hunting in Atlantic forest patches, Sao Paulo, Brazil. *Oryx*, 35: 137-144.

Curtis J.J. and McItonish R.P. (1950). The inter relationship of certain analytic synthetic phyto-sociological characters. *Ecology*, 31: 434-455.

Dayton P.K., Tegner M.J., Edwards P.B. and Riser K.L. (1998). Sliding baselines, ghosts, and reduced expectations in kelp forest communities. *Ecological Applications*, 8: 309-322.

de Merode E., Homewood K. and Cowlishaw G. (2004). The value of bushmeat and other wild foods to rural households living in extreme poverty in Democratic Republic of Congo. *Biological Conservation*, 118: 573-581.

Diamond J. and Case T.J. (1986). Overview, introduction, extinctions, exterminations and invasions. In: *community Ecology*, 56-79 (J. Diamond and T. J. Case, Eds.). Harper and Row, New York, U.S.A.

Eisenberg J.F. (1989). *Mammals of Neotropics*. Vol. 1. Chicago University Press, Chicago, U.S.A.

Emmons L.H. (1984). Geographic variations in densities and diversities of non-flying mammals in Amazonia. *Biotropica*, 16: 210-222.

Emmons L.H. (1990). *Neotropical Mammals: A Field Guide.* Chicago University Press, Chicago, U.S.A.

Estes J.A., Duggins D.O. and Rathbun G.B. (1998). The ecology of extinctions in kelp forest communities. *Conservation Biology*, 3: 251-264.

FSI (2005). *The State of Forest 2003*. Uttaranchal: Forest Survey of India, DehraDun, Uttaranchal, India.

Hilaluddin, Kaul R. and Ghose D. (2005a). Conservation implications of wild animal biomass extractions in northeast India. *Animal and Biodiversity Conservation*. 28: 1-7.

Hilaluddin, Kaul R. and Ghose D. (2005b). Galliformes extraction and use by some indigenous ethnic groups in Northeast India. In proceedings of 3rd International Galliformes Symposium, 70-76 (R. A. Fuller and S. Browne, Eds.). World Pheasant Association –International, Fordingbridge, U.K.

Hilaluddin, Kaul R., Pradhan S., Lachungpa U. and Taylor J. (2006). Wildmeat and its management in the eastern Indian Himalaya. Proceedings of the national conference on Environment and Education for Sustainable Lifestyle, 19-36. Patna Women's College Patna, Bihar, India.

Hilaluddin and Naqash R.Y. (2006). Survey and census report of Kishtwar High Altitude National Park. Department of Wildlife Protection, Kishtwar, Jammu and Kashmir, India. Unpublished report.

Hilaluddin (2006). Ecological consequences of bushmeat hunting on wildlife in the western Indian Himalaya. Oriental Bird Club, U.K. and Wildlife Conservation Society India Programme, U.S.A. Unpublished report.

Hill K. and Padwe J. (2000). Sustainability of Ache' hunting in the Mbaracayu Reserve, Paraguay. In: *Hunting for Sustainability in Tropical Forest*, 79-105 (J. R. Robinson and E. L. Bennett Eds.). Columbia University Press, New York, U. S. A.

ICBP (1992). *Putting Biodiversity on the Map: Priority Areas for Global Conservation*. International Council for Bird Preservation, Cambridge, U.K.

IUCN (2004). *2004 Red List of Threatened Species*. IUCN, Gland, Switzerland.

Kaul R., Hilaluddin and Jandrotia J.S. (2003). Extraction of wild meat in the western Indian Himalaya: an assessment. Oriental Bird Club, U.K. Unpublished report.

Kaul R., Hilaluddin, Jandrotia J.S. and McGowan P.J.K. (2004). Hunting of large mammals and pheasants in the western Indian Himalaya. *Oryx*., 9: 426-431.

Kumar R. and Shahabuddin G. (2006). Effects of biomass structure, diversity and composition of forests in Sariska Tiger Reserve. *Environmental Conservation*, 32: 1- 12.

Lahm S.A. (1993). Ecology and economics of human/wildlife interactions in north-eastern Gabon. University of New York, New York, U.S.A. Unpublished Ph. D. thesis.

Ludwig A.J. and Reynolds J.F. (1988). Statistical Ecology: A Premier on Methods and Computing. John Wiley and Sons, New York, U.S.A.

Mace G.M. and Balmford A. (2000). Patterns and processes in contemporary mammalian extinction. In: *Future Priorities for the Conservation of Mammalian Diversity*, 27-52 (A. Enthwhistle and N. Dunstone, Eds.). Cambridge University Press, Cambridge, U.K.

Madhusudan M.D. and Karanth U. (2002). Local hunting and the conservation of large mammals in India. *Ambio.*, 31: 49-54.

Martin P.S. and Steadman D.W. (1999). Pre-historic extinctions on islands and continents. In: *Extinction in Near Time: Causes, Contexts and Consequences*, 17-55 (R. D. E. Mocphee Eds.). Kluwer/Plenum, New York, U.S.A.

Muller-Dombois D. and Ellenberg H. (1974). *Aims and Methods of Vegetation Ecology*. New York: John Wiley and Sons, New York, U.S.A.

Nawaz R., Garson P.J. and Malik M. (2000). Monitoring pheasant populations in montane forests: some lessons learnt from Pakistan Galliformes Project. In: proceedings of the 2nd Galliformes Symposium, 196-203 (M. Woodburn, P. J. K. McGowan, J. Carroll, A. H. Musavi and Z. Zheng-wang Eds.). World Pheasant Association-International, Fordingbridge, U.K.

Novaro A.J., Fumes M.C. and Walker R.S. (2000). Ecological extinctions of native prey of a carnivore assemblage in Argentina Patagonia. *Biological Conservation*, 92: 25-34.

Olson D.M. and Dinerstein E. (1998). The global 2000: a representation approach to conserving the earth's most biologically valuable ecoregions. *Conservation Biology*, 12: 502 – 515.

Peres C.A. (1999a). The structure of non-violent mammal communities in different Amazonian forest types. In: *Mammals of Neotropics: The Central Neotropics*, 564-581(J. F. Eisenberg and K. H. Redford Eds.). Chicago University Press, Chicago, U.S.A.

Peres C.A. (1999b). Effects of subsistence hunting and forest types on Amazonian primate communities. In: *Primate Communities*, 268-273 (J. G. Fleagle, C. Janson and K. E. Reed Eds.). Cambridge University Press, Cambridge, U.K.

Peres C.A. (2000). Effects of subsistence hunting on vertebrate community structure in Amazonian forests. *Conservation Biology,* 14: 240-253.

Peres C.A. and Terborg J. (1995). Amazonian nature reserves: an analysis of defensibility, status of existing conservation units and design criteria for the region. *Conservation Biology,* 9: 34 – 46.

Prater S.H. (1971). *The Book of Indian Animals*. Oxford University Press, New Delhi, India.

Raman T.R.S. and Sukumar S. (2002). Responses of tropical rainforest birds to abandoned plantation edges and logged forests in the Western Ghats, India. *Animal Conservation*, 5: 201-216.

Redford K.H. (1992). The empty forests. *Bioscience,* 42: 412-422.

Redford K.H. and Eisenberg J.F. (1992). *Mammals of Neotropics: the Southern Cone*. Vol. 2. Chicago University Press, Chicago, U.S.A.

Redford K.H. and Feinsinger P. (2001). The half-empty forests: sustainable use and ecology of interactions. In: *Conservation of Exploited Species*, 371-399 (J. D. Reynolds, G. M. Mace, K. H. Redford and J. G. Robinson Eds.). Cambridge University Press, Cambridge, U.K.

Reid W.V. (1992). How many species will there be? In: *Tropical Deforestation and Species Extinction*, 55-73 (T. C. Whitmore and J. A. Sayer Eds.). Chapman and Hall, London, U.K.

Robinson J.G. and Bennett E.L. (2000). *Hunting for sustainability in Tropical Forests*. Columbia University Press, New York, U.S.A.

Rodgers W.A. and Panwar H.S. (1988). Planning a protected area network in India. Wildlife Institute of India, DehraDun, Uttaranchal, India. Volume I and II.

Rosser A.M. and Manica S.A. (2002). Over-exploitation and species extinctions. *Conservation Biology*, 16: 584-586.

Shannon C.E. and Wiener W. (1963). *The Mathematical Theory of Communication*. University Illinois Press, Urbana, U.S.A.

Skowno A.I. and Bond W.J. (2003). Bird community composition in an actively managed Savannah reserve: importance of vegetation structure and composition. *Biodiversity and Conservation*, 12: 2279-2294.

Sokal R.R. and Rohlf F.J. (1995). *Biometry: Principals and Practice of Statistics in Biological Research*. W. H. Freeman and Company, New York, U.S.A.

Sttatersfield A.J, Crosby M.J., Long A.J. and Wege D.C. (1998). *Endemic Bird Areas of the World: Priority Areas for Conservation*. Cambridge University Press, Cambridge, U.K.

Sutherland W.J. (1996). Why census? In: *Ecological Census Techniques: a Handbook*, 1-9 (W. J. Sutherland Eds.). Cambridge University Press, Cambridge, U.K.

Thiollay J. (1999). Responses of an avian community to rain forest degradation. *Biodiversity and Conservation*, 8: 513-534.

Wilkie D.S. and Carpenter J.F. (1999). Bushmeat hunting in the Congo Basin: an assessment of impacts and options for mitigations. *Biodiversity and Conservation*, 8: 929-955.