

## Chapter 5: Modelling leopard abundance: What can we learn?

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### 5.1 Introduction

Due to the fact that the management and monitoring of biodiversity is notoriously difficult and expensive, shortcuts such as ‘indicator’, ‘umbrella’, ‘keystone’, and ‘flagship’ species are commonly used to achieve biodiversity conservation targets (Simberloff 1998). According to Simberloff (1998) the ‘umbrella’ species (a species that has such a wide range, that the conservation of this species will encompass the conservation of many other species) seems to be the best approach. For this purpose, large carnivores, which can be seen as indicators of species richness (Gavashelishvili & Lukarevskiy 2008), are often employed as surrogates for biodiversity conservation. This is because they may regulate the numbers of their prey and consequently alter the structure and function of ecosystems (Estes *et al.* 1998; Terborgh *et al.* 1999). Even in ecosystems where prey abundance is regulated by bottom-up effects, such as rainfall, the interactions of carnivores with their prey play an important role in influencing population dynamics, behaviour, and evolution (Mills 2005). Thus, these intricate relationships between predators, prey, and their environment are undoubtedly linked to the functional component of biodiversity (Mills 2005; Miller *et al.* 2001), and provides the basis for the use of predators as conservation surrogates. Thus, the leopard, which is the largest remaining carnivore in the BMR (Henley 2000), can be employed as a surrogate for biodiversity conservation (Gavashelishvili & Lukarevskiy 2008).

Population size estimates are essential for ecological theory and wildlife management (Norton 1986; Smallwood 1997), and should be central to the development of conservation strategies (Smallwood 1999). There is a serious lack of information regarding the distribution and abundance of leopards in the Baviaanskloof Reserve Cluster (BRC), however the population is thought to be small and extremely insecure (Norton 1986; Stuart & Heinecken 1977). This gap in information urgently needs to be addressed in order to assist in leopard conservation (Gros *et al.* 1996). A lack of information on leopard numbers constrains our ability to understand the factors (e.g. retaliatory killing of leopards) shaping this community. This undermines the effective conservation of these felids. Several methods for the determination of carnivore abundance exist, each with their own advantages and limitations.

The first involves relating carnivore density to the density of signs of carnivores such as tracks (mountain lions, *Felis concolor*, Smallwood & Fitzhugh 1995; lions, *Panthera leo*, leopards, *Panthera pardus*, and wild dogs, *Lycaon pictus*, Stander 1998), and scat. This non-invasive technique does not require direct observations of the focal species. However, the relationship between the density of these signs and the population density is rarely known (Caughley 1977).

The second set of techniques involves transect/grid based ground and aerial animal counts. This method generally underestimates densities because of the fact that these carnivores are often well concealed, camouflaged, mostly nocturnal, or occur at very low densities (Mills 1997). Furthermore, the use of aircrafts in aerial censuses can be very expensive.

The third method involves mark-and-recapture as the basis for population estimates. This can be accomplished via physically marking individuals and releasing them back into the population, using genetic microsatellites (European badgers, *Meles meles*, Miller *et al.* 2005) or by using camera traps where the individuals are distinguished based on unique physical features (tigers, *Panthera tigris*, Karanath & Nichols 1998). Here, the Lincoln-method is used to estimate the number of individuals in the population. This can be a very useful technique if the assumptions (equal catchability, closed population, and random mixing of marked individuals in the population) are met. Nevertheless, the assumptions are rarely met (Mills 1997).

The final set of methods incorporates a modelling process, whereby predator density is determined via the use of certain habitat attributes (tigers, *Panthera tigris*, Ranganathan *et al.* 2008). This method relies heavily on the possibility of identifying the limiting resources for carnivore species, estimating the abundance and distribution thereof, and relating it to the abundance of the focal carnivore (Karanth *et al.* 2004). It is, however, notoriously difficult to develop such relationships in practice (Karanth *et al.* 2004). Several methods for the modelling of predator abundance do exist, including the relation of prey availability to predator density. The reason for this being that predators seem to be limited by prey resources, thus, prey availability is the fundamental determinant of the abundance of predators (Carbone & Gittleman 2002). There have been several studies documenting this relationship between prey biomass and predator biomass or abundance (Karanth & Nichols 1998; Karanth *et al.* 2004; Carbone & Gittleman 2002; Hayward *et al.* 2007; Gros *et al.* 1996; Ranganathan *et al.* 2008). Most of these studies of spatial distribution and abundance commonly employ either large-scale game counts or meta-analysis of several studies in order to make inferences regarding predator densities.

Here, a model is presented that utilises an estimate of potential prey availability, as well as game counts, to determine the potential abundance of leopards in the BRC. Firstly, the relationship between leopards and the available prey was used to estimate the density of leopards that could be supported in the BRC. Secondly, the methods of Boshoff *et al.* (2001), Hayward *et al.* (2007), Carbone & Gittleman (2002), and Norton & Henley (1987) was used to determine the density of leopards in the BRC, and these results were compared to the results of the model presented here. Thirdly, the BRC leopard density estimate was compared to the density estimates of other southern African leopard populations. Finally, the limitations, improvements, and conservation applications of the model is highlighted.

## **5.2 Methods**

This approach of estimating potential leopard abundance comprised of the estimation of the potential prey abundance based on Boshoff *et al.* (2001) and direct counts, and then the estimation of leopard abundance based on Hayward *et al.* (2007). This was undertaken in three steps: (1) determination of the distribution and extent of Mammal Habitat Classes (MHC) within the BRC, (2) determination of potential prey abundance for each MHC, and (3) calculation of potential leopard abundance using a model that predicts potential leopard density as a function of potential prey abundance.

### *5.2.1 Distribution of mammal habitat classes*

In order to determine the herbivore distributions and potential abundance in the BRC, MHCs were used as the spatial units for this analysis. These MHCs, which are the planning units for the Subtropical Thicket Ecosystem Planning (STEP) project, were defined according to various biological and environmental characteristics, including, vegetation type and structure, rainfall, geology and altitude (Cowling & Heijins 2001). They were obtained from the STEP project (Vlok & Euston-Brown 2002). ArcGIS 9.2 (ESRI 2006) was used to determine which MHCs occurred within the BRC. This was done by overlaying the BMR reserve and boundary layer, obtained from the BMR Project Management Unit (A. Skowno, unpubl. data), over the MHC layer, and extracting the boundaries of the various MHCs, as well as their extent.

### 5.2.2 Estimation of prey abundance

#### i. Modelled abundance

The potential herbivore distributions were calculated for each MHC. Firstly, a model developed by Boshoff *et al.* (2001; hereafter referred to as the ‘prey model’) was used. This method consists of a simple spreadsheet model that incorporates adjusted agricultural stocking rates based on Large Stock Units (LSUs; see Meissner *et al.* 1983), forage availability, and metabolic requirements of mammal species (see Boshoff *et al.* 2001 for details of the model). Only the 15 mammal species, with the exception of baboons, with a mass greater than *ca.* 2 kg that naturally occur within the BRC were incorporated in the model (Table 5.1).

Baboon numbers were not calculated by using the Boshoff *et al.* (2001) model, as it only caters for herbivore species. Thus, potential baboon numbers were calculated using the home range size of troops occurring in similar habitats; the data from de Vore & Hall (1965), who worked in the Cape of Good Hope Nature Reserve, was used. The number of individuals per troop was plotted against the home range size of each troop. The linear regression equation obtained ( $y = 0.412x + 0.646$ ; where  $y$  is range size and  $x$  is number of individuals) was then used to determine the potential numbers of baboons occurring in the BRC, assuming that there is no levelling off of home range size with an increase in number of individuals.

#### ii. Game count data

Game count data from a section of the BRC was used as an estimate of herbivore density (Venter *et al.* 2008). These counts were obtained by flying 300 m wide transects that were 900 m apart, in order to identify areas with potentially high animal densities, and covered an area of 55 554 ha (Venter *et al.* 2008). The high-density areas were then re-sampled by flying 300 m wide transects, and covered an area of approximately 27 000 ha. All transects were flown at a height of 30 m and at an average speed of 32 knots (Venter *et al.* 2008). These transects were flown via helicopter in triplicate over a period of 9 days, recording the incidence of all wildlife species, their location and their abundance (Venter *et al.* 2008). The resulting density estimates were then extrapolated to the entire BRC and were used to estimate potential leopard abundance.

**Table 5.1:** The body mass of herbivore species with a body mass greater than 2 kg, together with baboons, occurring in the Baviaanskloof Reserve Cluster.

<b>Herbivore</b>	<b>Species</b>	<b>Mean body mass</b>
Baboon, Chacma*	<i>Papio cyanocephalus ursinus</i>	12
Buffalo, African	<i>Syncerus caffer caffer</i>	432
Bushbuck*	<i>Tragelaphus scriptus</i>	46
Bushpig	<i>Potamochoerus porcus</i>	46
Duiker, Common*	<i>Sylvicapra grimmia</i>	16
Eland	<i>Tragelaphus oryx</i>	345
Grysbok, Cape*	<i>Raphicerus melanotis</i>	7
Hartebeest, Red	<i>Alcelaphus buselaphus</i>	95
Klipspringer*	<i>Oreotragus oreotragus</i>	10
Kudu*	<i>Tragelaphus strepsiceros</i>	135
Reedbuck, Mountain*	<i>Redunca fulvorufula</i>	23
Rhebok, Grey*	<i>Pelea capreolus</i>	7
Rhino, Black	<i>Diceros bicornis</i>	800
Steenbok*	<i>Raphicerus campestris</i>	8
Zebra, Cape mountain	<i>Equus zebra zebra</i>	179

\* indicates species used in the determination of potential leopard abundance

### 5.2.3 Estimation of potential leopard density

The potential prey density data was used to determine potential leopard abundance in the BRC. Calculation of potential leopard density followed the method developed by Hayward *et al.* (2007). The prey density (#.km<sup>-2</sup>) data was converted to biomass (kg.km<sup>-2</sup>) by using ¾ of adult female body mass in order to account for sub-adults and young that is preyed upon (Schaller 1972). These mean female body mass estimates were obtained from Stuart & Stuart (2000) and Skinner & Chimimba (2005) and were converted to ¾ adult female body mass.

Hayward *et al.* (2007) developed regression equations relating predator biomass to the biomass of prey falling within the predators' preferred prey weight range (Pw; 10 – 40 kg) as well as for significantly preferred prey species (Ps; bushbuck, common duiker, and impala). The

regression equations for Ps ( $y = -2.455 + 0.456x$ ; where  $y$  = potential leopard abundance and  $x$  =  $\log_{10}$  of prey biomass) and Pw ( $y = -2.248 + 0.405x$ ) were used in order to determine potential leopard abundance in the BRC.

Hayward *et al.* (2006a) found that leopards significantly prefer bushbuck, common duiker, and impala. However, this analysis only incorporated data from bushveld and savannah ecosystems. For this reason, the Ps estimate was expanded to incorporate all prey items (actual prey; Pa) occurring in the diet of leopards in the BRC. The data from Ott *et al.* (2008), who described the diet of leopards in the BMR, was used to determine the Pa. They found that the dominant prey items occurring in the diet consists of 60% native ungulates, 20% rodents, and 12.5% rock hyrax (Table 5.2). All the small mammals (i.e. rodents and rock hyrax), carnivores, exotic species (nyala), and domestic stock were excluded from the analysis due to the lack of data regarding their potential distribution and numbers in the MHCs. Thus, the prey species that formed part of the diet of leopards in the BRC were used in the determination of potential leopard abundance, these included mountain reedbuck, bushbuck, Cape grysbok, common duiker, steenbok, klipspringer, kudu, and grey rhebok. Baboons were also included in the initial analysis even though they do not form part of the Baviaanskloof leopard's diet, according to Ott *et al.* (2008).

The resultant density and abundance estimate was then compared to the estimates of Boshoff *et al.* (2001), Norton & Henley (1987), and Carbone & Gittleman (2002). These models were not developed specifically for the BRC, they were however applied to the BRC in order to make comparisons with our model. Boshoff *et al.* (2001) determined the potential abundance of leopards in the Cape Floristic Region, which resulted in an estimate of 2 individuals per 200 km<sup>2</sup>. Norton & Henley (1987) used home range analysis to estimate the densities of leopards in the southern Cape, which resulted in an estimate of 6 – 9 individuals per 100 km<sup>2</sup>. These two estimates were simply extrapolated to the size of the BRC. Carbone & Gittleman (2002) developed an equation that relates prey biomass to carnivore density. The prey biomass estimate of the BRC was then used to estimate leopard density in the BRC.

**Table 5.2:** Occurrence of indigenous prey items in the diet of leopards in the BMR (from Ott *et al.* 2008).

Prey species		Frequency of
Common name	Scientific name	occurrence (%)
Rodents	Class Rodentia	20.0
Reedbuck, Mountain	<i>Redunca fulvorufula</i>	17.5
Bushbuck	<i>Tragelaphus scriptus</i>	15.0
Hyrax, Rock	<i>Procavia capensis</i>	12.5
Grysbok, Cape	<i>Raphicerus melanotis</i>	10.0
Duiker, Common	<i>Sylvicapra grimmia</i>	5.0
Steenbok	<i>Raphicerus campestris</i>	5.0
Birds	Class Aves	5.0
Wild Cat, African	<i>Felis silvestris</i>	2.5
Klipspringer	<i>Oreotragus oreotragus</i>	2.5
Kudu	<i>Tragelaphus strepsiceros</i>	2.5
Rhebok, Grey	<i>Pelea capreolus</i>	2.5

## 5.3 Results

### 5.3.1 Mammal habitat classes

Twenty-six MHCs were identified in the study area (Table 5.3) and ranged in size from 8 ha for Central Valley Thicket with Succulent Karoo to 83 501 ha for Baviaanskloof Mountain Fynbos.

**Table 5.3:** The 26 Mammal Habitat Classes (MHC) used as the spatial units for the leopard density model, and their respective sizes in hectares (reflecting the extent of the MHCs in the Baviaanskloof Reserve Cluster).

<b>Mammal Habitat Class (MHC)</b>	<b>Size (ha)</b>
Afromontane Forest / Thicket Forest	9 032
Baviaans Fynbos Thicket	5 348
Baviaans Renoster Thicket	4 872
Baviaanskloof Mountain Fynbos	83 501
Central Valley Thicket with Succulent Karoo	8
Cockscomb Mountain Fynbos	44 992
Eastern Valley Thicket with Arid Fynbos	760
Eastern Valley Thicket with Succulent Karoo	1 003
Gamtoos Valley Thicket	3 795
Groot Spekboomveld	26
Groot Valley Thicket	19 183
Humansdorp Grassy Fynbos	3 322
Knysna Afromontane Forest	4 127
Kouga Mountain Fynbos	57 451
Kromme Fynbos / Renosterveld Mosaic	591
Langkloof Fynbos / Renosterveld Mosaic	680
Oudtshoorn Broken Veld	400
Steytlerville Broken Veld	179
Sundays Thicket	1 835
Sundays Valley Thicket	128
Sundays Valley Thicket with Grassy Karoo	1 069
Thicket / Valley Thicket with Fynbos	1 051
Thicket / Valley Thicket with Grassy Fynbos	3 901
Tsitsikamma Mountain Fynbos	21 028
Uniondale Inland Renosterveld	5 221
Valley / Arid Thicket with Renosterveld & Succulent Karoo	76
<b>Total</b>	<b>273 579</b>

### 5.3.2 Potential prey abundance

The Cape grysbok had the highest potential biomass with 86.9 kg per km<sup>2</sup> from the prey model outputs, compared to the grey rhebuck, which has the lowest potential biomass, with 2.6 kg per km<sup>2</sup> (Table 5.4). The potential contribution of each species to support potential leopard abundance thus varies in accordance with their biomass estimates. When comparing the potential biomass estimates, the total game count biomass estimate for the BRC (155.7 kg) is several orders of magnitude smaller than the total prey model estimate for the BRC (9.5 x 10<sup>5</sup> kg). It ranges from 0 to 2.00 x 10<sup>-2</sup> kg per km<sup>2</sup> for the game count, compared to 2.62 – 86.85 kg per km<sup>2</sup> for the prey model estimate (Table 5.4). This substantial difference persists for all the species involved. The most extreme differences occur between the game count and prey model estimates for grey rhebuck, steenbok and Cape mountain zebra. These three species were not recorded in the game count and consequently have a estimated biomass of zero for the game count data.

### 5.3.3 Potential leopard density and abundance

The prey species (Ps) and prey weight (Pw) models, following Hayward *et al* (2007) resulted in potential leopard densities of 0.0003 – 0.215 individuals per km<sup>2</sup> and 0.0005 – 0.0312 individuals per km<sup>2</sup> respectively (Table 5.5). The Ps method was then adapted to incorporate all the prey items (actual prey; Pa) occurring in the diet of leopards in the BRC, which resulted in an estimate that ranges from 0.0007 - 0.0381 individuals per km<sup>2</sup> (Table 5.5). The substantial difference in the potential prey biomass estimates of the prey model and the game count, together with the differences in the leopard models results in a huge range of potential leopard density estimates for the BRC (0.0003 – 0.0381 #/km<sup>2</sup>; Table 5.5).

**Table 5.4:** Comparison of potential wildlife densities and corresponding biomass determined by a prey-abundance model and estimated by a game count in the Baviaanskloof Reserve Cluster.

Species	Estimated Density		Estimated Biomass	
	(#/km <sup>2</sup> )		(kg/km <sup>2</sup> )	
	Model†	Count‡	Model†	Count‡
Baboon, Chacma <sup>a</sup>	2.43	1.09 x 10 <sup>-4</sup>	29.12	1.31 x 10 <sup>-3</sup>
Buffalo, African	0.02	4.62 x 10 <sup>-5</sup>	7.99	1.99 x 10 <sup>-2</sup>
Bushbuck <sup>b c</sup>	0.75	6.35 x 10 <sup>-5</sup>	34.28	2.92 x 10 <sup>-3</sup>
Bushpig	0.34	1.48 x 10 <sup>-6</sup>	15.75	6.81 x 10 <sup>-5</sup>
Duiker, Common <sup>a b c</sup>	1.21	9.01 x 10 <sup>-6</sup>	19.29	1.44 x 10 <sup>-4</sup>
Eland	0.10	1.56 x 10 <sup>-5</sup>	33.16	4.36 x 10 <sup>-3</sup>
Grysbok, Cape <sup>c</sup>	12.41	4.20 x 10 <sup>-6</sup>	86.85	2.92 x 10 <sup>-5</sup>
Hartebeest, Red	0.09	6.53 x 10 <sup>-5</sup>	8.36	6.20 x 10 <sup>-3</sup>
Klipspringer <sup>a c</sup>	1.49	2.11 x 10 <sup>-5</sup>	14.85	2.11 x 10 <sup>-4</sup>
Kudu <sup>c</sup>	0.07	1.48 x 10 <sup>-4</sup>	9.75	2.00 x 10 <sup>-2</sup>
Reedbuck, Mountain <sup>a c</sup>	0.22	2.77 x 10 <sup>-5</sup>	5.00	6.36 x 10 <sup>-4</sup>
Rhebuck, Grey <sup>c</sup>	0.35	0.00	2.62	0.00
Rhino, Black	0.05	n/a	43.05	n/a
Steenbok <sup>c</sup>	1.76	0.00	14.06	0.00
Zebra, Cape mountain	0.13	0.00	23.13	0.00
<b>Total biomass</b>			347.26	5.69 x 10 <sup>-2</sup>

† Estimates obtained from the adapted herbivore model of Boshoff *et al.* (2002).

‡ Estimates obtained from game counts (Venter *et al.* 2008).

a Refers to prey used in the Pw calculation of potential leopard density

b Refers to prey used in the Ps calculation of potential leopard density

c Refers to prey used in the Pa calculation of potential leopard density

n/a Omitted due to security reasons.

**Table 5.5:** Potential leopard density ( $\#/km^2$ ) and population size (in parenthesis) estimated by the three prey-based models, and the estimated biomass used to determine this.

Prey-based model	Biomass ( $kg/km^2$ )	Herbivore model	Game count
Prey species (Ps)	53.57	0.0215 (59)	0.0003 (1)
Prey weight (Pw)	68.26	0.0312 (86)	0.0005 (1)
Actual prey (Pa)	186.70	0.0381 (104)	0.0007 (2)

#### 5.3.4 Comparison of potential leopard densities

Our estimate of 59 - 104 individuals (Table 5.5) was then compared to those of Boshoff *et al.* (2001), Norton *et al.* (1984), and Carbone & Gittleman (2002), with their estimates being 27, 124, and 164 - 246 individuals respectively (Table 5.6).

**Table 5.6:** Comparison of the potential leopard abundance (total number of individuals) and potential leopard density (number of individuals per  $km^2$ ) in the BRC, using several carnivore density estimates.

Reference	Potential leopard density ( $\#/km^2$ )	Potential leopard abundance (total #)
This study*	0.022 - 0.038	59 - 104
Boshoff <i>et al.</i> 2001†	0.010	27
Carbone & Gittleman 2002*	0.046	124
Norton & Henley 1987‡	0.060 - 0.090	164 - 246

\* Models using prey biomass to estimate leopard abundance

† Estimate based on broad habitat requirements

‡ Estimate based on home-range analysis

The published potential leopard densities of several southern African areas are shown in Table 5.7. These estimates were determined by examining populations that occur in a wide variety of habitats and by employing various techniques. The estimates range from a minimum of 0.002 individuals per  $km^2$  in Mountain Fynbos to a maximum of 0.303 individuals per  $km^2$  in woodlands. There is a large range of estimates as well as a large overlap between estimates. When comparing average densities the estimate of 0.030 individuals per  $km^2$  in this study is comparable to Bailey's estimate of 0.035 individuals per  $km^2$ . The average density estimates of

Martins & Martins (2006), Stander (1998), Bothma & le Riche (1984), and Mills (1984) are lower than the present study, and those of Norton & Henley (1987), and Stuart & Stuart (1991) are higher.

## **5.4 Discussion**

### *5.4.1 Potential leopard density and abundance*

The differences in the abundance and biomass estimates between the prey model and the game count (Table 5.4) resulted in a large range of potential leopard densities (Table 5.5). This discrepancy is due to the general inaccuracies of the game count. This because game counts are based on raw count data that has an unknown relationship with true animal density, the spatial units are not conducive to extrapolation to the entire reserve (Karanath *et al* 2004), and are generally biased towards animals with a larger body size (Mills 1997), as can be seen in Table 5.4. The leopard's diet in the BMR also consists mostly of smaller mammals (Table 5.2), which are likely to be under represented in the game count estimate (Norton-Griffiths 1978). When incorporating the game count data, it results in an estimate of one or two individuals. This estimate is incorrect because at least eight leopards have already been collared by a local NGO for the purpose of radio-telemetry (Rogers 2008) in the BMR. This highlights the importance of a limiting factor such as prey density, and the accurate measurements thereof for the prediction of leopard densities. The importance of the relationship between prey biomass and predator biomass has also been documented by several other studies (Carbone & Gittleman 2002). For this reason, the game count estimates of prey biomass were not included in the determination of potential leopard density and abundance. However, if the game count data was more accurate or supplemented with ground based techniques such as survey transects, it would have been conducive to more accurate predictions of potential leopard abundance. By excluding the game count estimates of prey abundance, we arrive at a potential leopard density of 0.022 – 0.038 individuals per km<sup>2</sup> (59 – 104 individuals in the BRC). It must however, be stressed that our model does not precisely predict the actual population size of leopards in the BRC. However, it does give a broad estimation of the potential leopard densities and population sizes, and identifies areas where research needs to be focussed.

**Table 5.7:** Comparison of a selection of leopard density estimates (average densities in parenthesis) for different habitat types in southern Africa

Study area	Habitat type	Estimated leopard density (#/km <sup>2</sup> )	Methods	Source
Baviaanskloof MR, SA	Mountain	0.022 – 0.038	Prey-based model	This study
	Fynbos	(0.030)*		
Cederberg WA, SA	Mountain	0.002 – 0.004	Radio-tracking	Martins & Martins (2006)
	Fynbos	(0.003)		
Cederberg WA, SA	Mountain	0.060 – 0.090	Radio-tracking	Norton & Henley (1987)
	Fynbos	(0.075)		
Cederberg WA, SA	Mountain	0.040 – 0.060	Spoor counts	Stuart & Stuart (1991)
	Fynbos	(0.050)		
Kaudom GR, Namibia	Woodland	(0.013)	Radio-tracking	Stander (1998)
Kruger NP, SA	Woodland	0.006 – 0.303 (0.035)	Radio-tracking	Bailey (1993)
Kalahari Gemsbok NP, SA	Semi-desert & Tree savanna	(0.006)	Spoor counts	Bothma & le Riche (1984)
Kalahari Gemsbok NP, SA	Semi-desert & Tree savanna	(0.011)	Radio-tracking	Mills (1984)

\* Indicates potential leopard density

MR = Mega-Reserve; SA = South Africa; WA = Wilderness Area; GR = Game Reserve; NP = National Park

#### 5.4.2 Comparison of potential leopard densities

A comparison of this result to other techniques applied to the BRC showed that there is a wide range of estimates and techniques used to estimate potential leopard abundance (Table 5.6). The Carbone & Gittleman (2002) model incorporated many carnivores across various habitat types, and is a general model (Carbone & Gittleman 2002), which is not specifically geared towards determining leopard abundance. This model has been used to estimate carnivore densities successfully (Carbone & Gittleman 2002). However, this estimate is in all probability much higher than the actual densities of leopards in the BRC. The Boshoff *et al.* (2001) estimate, which forms the lower limit of the range of estimates, was based on a literature review that incorporates information on leopard densities, social structures, breeding units, territory sizes and home ranges (Boshoff *et al.* 2001). This information was then applied to the CFR, which was treated as a single homogenous unit. In other words, it does not take into account the variation in potential leopard abundance according to habitat type and prey abundance. The Norton & Henley (1987) estimate, which forms the upper limit of the range, was determined by the home-range analysis of leopards in the mountainous areas of the southern Cape (Norton & Henley 1987). However, the accuracy of this estimate is questionable because of its small sample size ( $n=3$ ), and the short duration of their study (Martins & Martins 2006). It may thus, not be applicable to the BRC. Therefore, this model provides a more accurate prediction of potential leopard densities for two reasons. Firstly, it is based on the method of Hayward *et al.* (2007), which accurately predicts leopard densities (Hayward *et al.* 2007). Secondly, the Hayward model is improved by incorporating the potential prey of leopards in the BRC.

Comparing the various leopard densities in southern African regions also produces interesting results (Table 5.7). The estimates for the mountainous Fynbos areas of the CFR range from 0.002 to 0.060 individuals per  $\text{km}^2$  compared to 0.006 – 0.011 individuals per  $\text{km}^2$  for semi-deserts / tree savannas, and 0.006 – 0.303 individuals per  $\text{km}^2$  for woodlands. There is overlap between these various estimates to a certain extent, with semi-deserts having the lowest average density (0.009/ $\text{km}^2$ ), followed by mountain fynbos (0.027/ $\text{km}^2$ ) and finally wooded savannas (0.107/ $\text{km}^2$ ). These differences in leopard densities between various habitat types are due to the productivity of the habitat, which in turn influences prey abundance and distribution (Mizutani 1998). Prey density and distribution emerges as the most important limiting factor (Bailey 1993). This can account for the higher densities of leopards in Fynbos, which is considered to support low densities of large herbivores (Venter *et al.* 2008), compared to dryer areas such as semi-deserts. Leopards in the BRC prey mostly on small

mammals such as rodents, and small antelope (Ott *et al.* 2008), which are non-nomadic in fynbos. In dry areas, the nomadic nature of prey items can result in seasonal prey scarcity, which necessitates large home ranges and consequently much lower densities (Bothma & le Riche 1984). This is the case for the leopard population in the Kalahari Gemsbok National Park, with densities as low as 0.006 individuals per km<sup>2</sup> (Bothma & le Riche 1984). Inter specific social dominance (Durant 1998) and competition (Bailey 1993) plays an important role in structuring carnivore communities. The fact that leopards are the dominant predators in the reserves of the CFR (Henley 2000) might explain the higher densities in the BRC than in dryer areas, where competition with other predators and inter specific social dominance may result in lower leopard densities.

These differences between habitat types are expected. However, one would expect similar results comparing population estimation techniques within comparable habitats. This is however, not the case. Comparing the leopard densities estimates of the Cederberg Wilderness Area highlights the incongruent nature of these various techniques (Table 5.7). The situation seems even more confounded by the difference in estimated leopard densities whilst using the same technique. Take for example the estimates of Martins & Martins (2006), and Norton & Henley (1987) who both determined the potential leopard abundance via radio tracking and home range analysis. The respective estimates of 0.002 – 0.004 individuals per km<sup>2</sup> and 0.019 – 0.031 individuals per km<sup>2</sup> differ by an order of magnitude.

These widely different estimates reflect the various approaches and methods used to determine leopard densities as well as the habitat type in question, and highlight the fact that there is a serious lack of knowledge regarding the estimation of potential leopard abundance. The application of these techniques to leopard conservation can produce varying results, which will ultimately negatively affect leopard conservation. There is a need to identify techniques that will accurately predict leopard densities in the CFR. Our model provides a platform for such a technique.

#### 5.4.3 Model limitations and improvements

The model does not exactly predict the density of leopard in the BRC. It does provide a range of potential leopard densities and identifies areas where the lack of knowledge prevents the accurate prediction of leopard densities. These are: (1) behavioural processes such as territoriality, intra-specific competition, habitat selection, and dominance, which can be obtained from home range analysis, and (2) incorporating accurate information on prey densities, including rodents and other small mammals, as this forms a major portion of the leopard diet (Ott *et al* 2007). Finally, the ideal would be to test the improved model predictions on a population of leopards with a known population size (Hayward *et al.* 2007).

An example of such improvements on the model does exist. Here, we incorporated habitat selection for illustrative purposes only. Our estimate of 59 - 104 individuals for the BRC includes all 26 MHCs occurring within the BRC. However, there is some preliminary data from the BMR (Rogers 2008), that suggests that leopards move mostly in gorges and rivers, only emerging when they move between two gorges and rivers, or for the occasional hunt. It therefore, seems pertinent that the model can be further refined to incorporate various habitat selection scenarios. For this reason, the overall potential leopard density was applied to each of the MHCs in the BRC and only the MHCs occurring in the gorges and rivers were incorporated into the model. This results in an estimate of 10 - 17 individuals for the BRC. This estimate is in all probably a more accurate reflection of the actual leopard population size in the BRC as opposed to the potential leopard densities that can be sustained by the estimated prey base, presented here. If this is the case, this population is extremely small. The BRC therefore needs to be expanded via the purchase of more conservation land or contractual agreements with neighbouring landowners in order to provide additional habitat for leopards (Henley 2000).

In conclusion, the actual number of leopards in the BRC is probably much lower than the upper limit of 104 individuals. This can be attributed to the discontinuous nature of the cluster of reserves intersected by areas of non-conservation landuses. These hostile environments effectively create island populations. This can ultimately result in a potent edge effect, whereby conflict with landowners may lead to the local extinction of these leopards (Woodroffe & Ginsberg 1998).