CHAPTER 3
ABUNDANCE AND OCCUPANCY ESTIMATION OF JACKAL

3.1. Introduction:

A population estimate is an approximation of population size \( (N) \) derived from a sample of the population \( (C) \), accounting for the fact that not all the animals in the study area are counted but assuming that the sample represents the entire population to draw inferences. The estimate of the population size is thus computed by estimating the proportion of animals of the entire population that have been sampled i.e. \( N = \frac{C}{p} \) \( (N\) - The estimate of the population size; \( p\) - Detection probability or capture probability) (Royle and Nichols 2003). Abundance (the number of individuals in a population) is one measure that can be used to characterize the state of a population for a single species, with changes in abundance reflecting changes in the population’s status. However, in order to make an accurate conclusion about changes in abundance, it is important that the probability of observing an individual is incorporated into the inferential process. The estimation of animal abundance is an important measure in both the theoretical and applied biological sciences.

Ecological research depends on the knowledge of animal abundance and how abundance is changing over space and time (Krebs, 2001). For many species, however, an accurate measure of abundance is difficult to obtain because some animals present in the study area may go undetected (Thompson, 1992). Instead of accounting for this, some monitoring programs use the observed counts as a proxy for true population size. This comes with the implicit assumption that either detection probability is perfect (if interest is in abundance), or the detection probability is constant on average across time (if interest is in trend in abundance). It has been shown for many species that analyzing raw counts without accounting for detection probability leads to bias in abundance and trend estimates (Wenger and Freeman, 2008).

In many monitoring programs one of the most difficult steps to overcome is obtaining data on elusive species. It is more challenging to estimate abundance of
some species that are nocturnal, cryptic and not individually identifiable from natural markings, by using conventional non-invasive techniques, such as distance sampling or photographic capture-recapture methods.

The reason behind using occupancy rather than abundance is that at an appropriate scale the two state variables should be positively correlated (i.e., occupancy may increase with increasing abundance), although the two state variables address distinctly different aspects of the population dynamics. However, for certain species the discrepancies between the two state variables may be minimal if the size of sampling unit is chosen appropriately. Few species are likely to be so conspicuous that they will always be detected at a sampling unit (site) when present. Dependent upon the survey methods being used, there may be a reasonable chance that the species goes undetected and is declared to be “falsely absent”. By not correcting for the fact that the species may go undetected, a naïve count of the number of sites where the species is detected will underestimate the true level of occupancy. Inferences about changes in occupancy based upon an observed difference between two (or more) naïve counts to be made carefully, as the difference may be the result of a change in ability to detect the species rather than a change in occupancy. The arguments against using a naïve count for occupancy are very similar to those given for not using a simple count as an index of abundance (e.g., Yoccoz et al., 2001, MacKenzie & Kendall, 2002; Williams et al., 2002; Schmidt et al, 2003).

In biodiversity monitoring, it has become increasingly important to use presence-only data due to the high cost of collecting count data with a strict protocol. However, recent advances in ‘occupancy modeling’ of animal presence data derived from photographic captures might provide solutions to the problems of monitoring non-identifiable species. Site occupancy (presence/absence) modeling is recognized as an effective technique for monitoring populations of secretive species on a landscape scale, historically considered a daunting or even impossible task. Site occupancy modeling may be the only feasible metric for monitoring population status of some species, such as those with recapture probabilities too low to use mark-recapture models effectively (Dorcas and Willson 2009). For categorical, presence–absence data to be useful, however, estimates of species-specific detection probabilities must be incorporated into surveys. A specific occupancy approach – the
Royle and Nichols (2003), allows for reliable estimation of abundance at best, and of an index of abundance or occupancy rate at the least, without the need for individual identification of animals. A variety of methods are available for estimating animal abundance (Lancia et al., 1994), but all involve the issue of estimating detection probabilities for specific kinds of count statistics (Buckland et al., 1993; Seber, 1982; Williams et al., 2002). Depending on the species being studied, the techniques available for gathering appropriate data, and incorporating the limitations of time, money and effort, only one or just a few of these methods may be suitable.

The density of lesser carnivores is perceived to be relatively low, thus making mark-recapture studies highly impractical for conspecifics. It is not always possible to obtain photographs of individuals within the species from camera traps. Mark-recapture methods require repeated efforts to capture or observe animals (Otis et al., 1978, Pollock et al., 1990) and even observation-based methods such as distance sampling (Buckland et al., 2001) or multiple observers (Cook and Jacobson 1979, Nichols et al., 2000) are viewed as being time and effort consuming.

Despite the logistic constraints, these methods have been widely applied. A potential approach to estimating abundance involves shifting the focus from numbers of animals to numbers of sample units occupied by animals (Royle and Nichols 2003). Methods employing this general approach are based on presence-absence data from the sampling units. Royle and Nichols (2003) have developed a model based on this focus to estimate abundance from repeated presence-absence data or point counts. They link the probability of detecting presence and the abundance at a sampling unit by using repeated detection-non-detection data gathered from occupancy surveys, suggesting a maximum likelihood approach at estimating the parameters (that includes abundance). In spite of the relative ease with which presence-absence data may be gathered, achieving large samples for analysis as suggested by Royle and Nichols (2003) for even practical estimates of the parameters might be difficult. Therefore this study has relevance to advancing both the scientific understanding and conservation of non identifiable species in the study area and elsewhere in the world.

Management and conservation of species relies heavily on understanding the variation in population abundance or density of a target species. Small carnivore species tend to be difficult to study due to their elusive habits, and crepuscular or
nocturnal activities. The population density of carnivores is governed by several factors, including prey availability (Carbone and Gittleman 2002, Ramesh 2010), habitat structure (Pereira 2009) and hunting by humans (Duckworth et al., 2005). Camera trapping is becoming increasingly common in documenting small carnivores (Gardner et al., 2010, Gerber et al., 2010, Pereira et al., 2011). Field studies have recently provided empirical support for the use of a maximum likelihood spatially-explicit capture-recapture model (Kalle et al., 2011). Despite the availability of newer models, it is still common for studies to use traditional ad hoc density estimation techniques (Negroes et al., 2010). Small carnivores often have low detection rates, even with intense sampling efforts, which can either inhibit the application of closed capture-recapture analyses or simply provide imprecise estimates (White et al., 1982, Maffei et al., 2004).

However, modern analytical approaches that shift focus from counting animals to counting patches or sites potentially occupied by them can permit estimating populations of non-uniqely identifiable species. They must also deal with the fact that not all animals present even within a sampled unit are detected during the survey, and therefore that the probability of detecting an animal in the sampled area is often less than one (Williams et al., 2002, Karanth et al., 2004). Camera traps can indicate relative abundance of a species with the assumption that photo detection rates are related to animal abundance (Morruzzi et al., 2002). Although trapping rate surveys (e.g. Carbone et al., 2001) have been used as indicators of animal abundance, detection probability is generally not estimated while doing so. As a result, its use is controversial both on theoretical and practical grounds (Jennelle et al., 2002). If species presence data from camera trap photos could be used to model and derive animal abundances or at least to derive reliable indices of abundance, (which can effectively deal with imperfect detections) it will be a major advance in animal monitoring science.

In this chapter, I estimated the abundance of jackal using presence/absence data from repeated samples using camera traps. The present study also attempted to compare jackal occupancy and detection probability between seasons in Sariska.
3.2 Materials and methods

3.2.1 Sampling design for camera traps

**Camera traps:** A preliminary survey was carried out in the intensive study area of 160 km² in the National Park by surveying available trails. Indirect signs such as spoor, scats and track signs of jackal were identified and marked using a handheld Global Positioning System (Garmin® 72). Camera traps were placed in 2 x 2 km² grid on the basis of any evidence (spoor, scats and track sign) on the trails where the probability of theft of cameras was low (figure 3.1). Total 42 units of digital cameras that worked on passive infrared motion and heat sensors were deployed, out of which 20 units of cameras were of Spy-point FLA1 and 22 units were of Moultrie cam. The camera traps were having resolution of 4 megapixel, detection angle of 95° where, one sensor covering five zone detection and having date, time and identity (as specified by the user) printed in the picture obtained. The camera delay was kept at a minimum by default of 15 seconds and multi shot options at level three (three consecutive picture in every detection). A total of 160 locations were selected for the placement of camera traps in the study area. At each location, pair of passive infrared digital camera traps was placed opposite each other along the roads and trails. Since it was logistically difficult to conduct sampling at all these camera trap locations simultaneously, the trap points were divided into four blocks, having 21 camera trap stations at each block depending upon the feasibility of camera trap locations in the study area. Total 84 trapping locations covered minimum convex polygon area of 118.7 km² and an effective trapping area (ETA) of 223.8 km². The mean inter trap distance was 750 m (ranging from 700 m to 1.2 km) and each camera traps ran for 32 consecutive occasions with the total sampling period amounting to 128 days (1344 trap nights per season). Capture matrix was prepared from each sampling occasion combined with captures from one day drawn from each block (Otis et al., 1978; Harihar et al., 2010; Gupta et al., 2010). The camera trapping was done in two seasons, winter (mid of October to February) and summer (March to July mid) depending on the logistics.

Cameras were kept approximately 25 cm above the ground and set to be active for 24 h/day. Bob cat lure was used at each location to attract animals. I
checked sampling stations on an average of every 5 days to ensure continued operation of cameras and replaced batteries and SD memory cards when required.

Figure 3.1. locations of camera traps and effective trapping area in the study area.

3.2.2. Relative abundance index

Photographic rate is the relative index of the animal’s spatial use and a crude abundance estimate (Carbone et al., 2001). Photographs provided information on date and time of the picture taken. Sometimes, individuals were photographed from only one of the two cameras operating at a single camera station. The event of capturing an individual, whether it was photographed by two camera traps or one, was considered to be an independent record of that individual. On some occasions, individuals were captured more than once at a camera station during a period (<1 min); thus, to avoid
pseudo-replications, I considered the first capture of the jackal as an independent record. Photos with more than one individual in the frame were counted as single detection for the species. I calculated relative abundance index (RAI) for each camera trap location by dividing the total number of independent records from the total trap nights x 100 (Carbone et al., 2001, O’Brien et al., 2003). Hence the RAI for each species is presented as the number of photo captures per 100 trap nights (Kalle, 2013). To compare RAI of a species between seasons, I used Student’s t-test in program SPSS 16 (SPSS 2007).

3.2.3. Occupancy analysis

Since the identification of individual jackal is difficult, the estimation of abundance using capture-recapture models used in camera-trap surveys of large cats was not possible. One possible solution is to use occupancy as a surrogate for abundance (MacKenzie and Nichols 2004). MacKenzie et al., (2002) developed a model to estimate site occupancy and detection probability based on repeated presence–absence data from multiple sites. Royle and Nichols (2003) extended this model to allow for abundance-induced heterogeneity. The Royle and Nichols (2003) model assumes that populations are closed and individuals are distributed in space according to a Poisson process. If these assumptions are violated, the estimated parameters should not be interpreted as abundance but rather as a random effect (MacKenzie et al., 2006). However, occupancy estimates will still be less biased than models that do not include heterogeneity. For analysis, I defined the minimum time between two independent events as 5 min. This means that if the individual was photographed more than once by the same camera in the course of 5 min, this was only counted as one event. Capture histories were developed for each location by treating one day as 1 trapping occasion (e.g., days 1 = first trapping occasion, days 2 = second trapping occasion, etc.). For each occasion the target species could get a 1 or a 0, where 1 indicates that the animal was captured at the sampling station during that trapping occasion and 0 if it was not captured.

I used Royle and Nichols (2003) heterogeneity model and repeated count method (Royle 2004). The Royle and Nichols (2003) model assumes that variation in animal abundance from one site to the next is probably the most important source of heterogeneity in detection probability among sites. They suggest that by basing heterogeneity in detection probability on varying site-specific animal abundance, it is
possible to exploit this very relationship to estimate the abundance parameter, and consequently derive the probability of occupancy. They use the occupancy based approach and assume that the detection probability of a given species at a particular site is directly dependent on the abundance of that species in that site for a given animal’s specific detection probability. Consequently, the heterogeneity in detection probabilities across sites is caused by the heterogeneity in abundance across those sites. And, by modeling the variation in abundances according to some probability distribution model (e.g., Poisson, when the number of animals inhabiting one site is random and independent of the number of animals at other sites), they build a model based on maximum likelihood to arrive at estimates of abundance in these sites. Modeling was conducted based on a priori ecological information and home range estimates of these species from available literature. There could be a possibility that these species would even travel beyond 1 km² thus modeling based on the aforementioned assumption would give unbiased abundance estimates.

The Royle and Nichols (2003) model assumes that populations are closed and that individuals are distributed in spaces according to a Poisson process, and the probability of detecting an animal at a site is a function of how many animals are actually at that site. The spatial distribution of animals is simply how many animals occur at each camera site within the study area. Each of the camera sites will contain some number of animals (some sites may contain 0 animals). That number, the site abundance, is a function of the mechanism governing the distribution. The spatial distribution of animals can meet Poisson assumptions when the number of animals inhabiting one camera site is random and independent of the number of animals at other sites. Eventually there are chances of multiple individuals occupying a camera trap site. Both models were analyzed in program Presence 4.1 (http://www.mbrpwrc.usgs.gov/software/presence.html).

To investigate the relative abundance for golden jackal I looked at differences in detection probability, occupancy rates and average abundance between seasons in each year. Hence estimated average abundance (λ) is interpreted as individuals per km². Mean abundance estimates were compared between two years and seasons using Student’s t-test in program SPSS 16 (SPSS 2007). I performed a Pearson correlation
test between mean abundance ($\lambda$) as obtained from occupancy approach and relative abundance index (captures/100 trap nights).

3.3 Results

3.3.1 Capture success and sample adequacy

Totally, 6720 trap-nights yielded 202 photographs of jackal, of which 11.88 % (n = 17) was obtained in summer 2011, 40.56 % (n = 58) in winter 2011-12, 14.69 % (n = 21) in summer 2012 and 32.87 % (n = 47) in winter 2012-13 (table.3.1). Jackal photo captures rate stabilized at minimum of 29 occasions (29 days) in 223 km$^2$ effective trapping area in the study area (figure.3.2).

![Figure 3.2](image)

Figure 3.2. Number of total photographs with increasing number of sampling occasions to evaluate sampling adequacy during the study period in Sariska Tiger Reserve, Rajasthan.

3.3.2 Relative abundance index

Average capture frequencies of jackal across the years ranged from 1.26 captures/100 trap-nights to 4.31 captures/100 trap-nights. Jackal RAI differed significantly across two years (t test; t =6.23, df = 2, P = 0.00) but, did not differ seasonally (t test, t = 9.51, df = 1, P = 0.06).

Table 3.1. Overall photo captures and relative abundance index (captures/100 trap nights) of golden jackal in Sariska Tiger Reserve (2011 to 2013).
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<table>
<thead>
<tr>
<th>Year</th>
<th>Season</th>
<th>Photo captures</th>
<th>RAI</th>
</tr>
</thead>
<tbody>
<tr>
<td>2011</td>
<td>Summer</td>
<td>17</td>
<td>1.26</td>
</tr>
<tr>
<td>2011-12</td>
<td>Winter</td>
<td>58</td>
<td>4.31</td>
</tr>
<tr>
<td>2012</td>
<td>Summer</td>
<td>21</td>
<td>1.56</td>
</tr>
<tr>
<td>2012-13</td>
<td>Winter</td>
<td>47</td>
<td>3.49</td>
</tr>
<tr>
<td></td>
<td>Average</td>
<td>35.75</td>
<td>2.65</td>
</tr>
</tbody>
</table>

Table 3.2. Detection probability ($r$), average abundance/km$^2$ ($\lambda$), site occupancy ($\Psi$) and total abundance ($N$) with associated standard errors ($\pm$ SE) for golden jackal based on camera trap data using the Royle-Nichols Heterogeneity model.

<table>
<thead>
<tr>
<th>Year</th>
<th>Season</th>
<th>$r_{\text{mean}} \pm \text{SE}$</th>
<th>$\lambda_{\text{mean}} \pm \text{SE}$</th>
<th>Naïve occupancy</th>
<th>$\Psi_{\text{mean}} \pm \text{SE}$</th>
<th>$N_{\text{mean}} \pm \text{SE}$</th>
<th>95% CI$\text{mean}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>2011</td>
<td>Summer</td>
<td>0.04±0.01</td>
<td>0.22±0.13</td>
<td>0.17</td>
<td>0.22±0.12</td>
<td>14 ± 6</td>
<td>7.5-34.2</td>
</tr>
<tr>
<td>2011-12</td>
<td>Winter</td>
<td>0.06±0.01</td>
<td>0.37±0.09</td>
<td>0.25</td>
<td>0.31±0.06</td>
<td>29 ±7</td>
<td>18.2-47.2</td>
</tr>
<tr>
<td>2012</td>
<td>Summer</td>
<td>0.04±0.01</td>
<td>0.28±0.10</td>
<td>0.19</td>
<td>0.24±0.07</td>
<td>16 ± 5</td>
<td>7.9-33.4</td>
</tr>
<tr>
<td>2012-13</td>
<td>Winter</td>
<td>0.07±0.01</td>
<td>0.38±0.10</td>
<td>0.26</td>
<td>0.31±0.06</td>
<td>27 ± 6</td>
<td>16.9-49.3</td>
</tr>
</tbody>
</table>

3.3.3. Detection probability

Estimates of detection probability of jackal varied from 0.039 to 0.069, with standard errors from 0.012 to 0.014 (table 3.2). Detection probability varied across the years ($t$ test, $t = 7.26$, $df = 3$, $P = 0.01$) and did not differ significantly in summer ($t$ test, $t = 1.48$, $df = 1$, $P = 0.07$) and winter ($t$ test, $t = 10.5$, $df = 1$, $P = 0.06$) across two years.

3.3.4. Occupancy and average abundance

Detection probability ($r$) of jackal was estimated high in winter (0.07 ± 0.01) as compared to summer (0.04 ± 0.01) and the probability of site occurrence ($\Psi$) of
jackal was found high in winter (0.31 ± 0.06) than summer (0.23 ± 0.19). Seasonal estimates of occupancy varied from 0.22 to 0.31, with standard errors from 0.07 to 0.12 (table 3.2) and the average abundance ranged from 0.22 to 0.38 with standard errors from 0.09 to 0.13 in two years (tables 3.2). Modeled site occupancies were observed greater than naïve occupancy. Jackal site occupancy differed significantly across years (t test, t = 11.51, df = 3, P = 0.001) and in summer (t test, t = 23.00, df = 1, P = 0.03) although no significant difference was observed in winter across two years (t test could not be done for the same values of the occupancy estimates). Abundance (N) obtained from occupancy approach and relative abundance index (captures/100 trap nights) were not significantly correlated (Pearson correlation test, p = 0.93, n = 4, P = 0.07) with each other. And there was a significant positive relationship between mean abundance (λ) and RAI (Pearson correlation; R² = 0.87, n = 8, P = 0.004, figure 3.3).

![Figure 3.3](image)

**Figure 3.3.** Relationship between mean abundance (λ) and relative abundance index (captures/100 trap nights) for jackal in Sariska Tiger Reserve, Rajasthan.

### 3.4 Discussion

I provided estimates of detection probability, site occupancy and relative abundance for jackal in Sariska using occupancy based Royle and Nichols (2003) model which provides a useful tool for the estimation of abundance of non-uniquely identifiable and cryptic species, since it is relatively inexpensive to obtain presence-absence data from sites. There was considerable inter season variability in species detectability. However, occupancy estimates for frequently occurring species such as jackal, were statistically robust and useful as a population index over time, although results should be interpreted cautiously. Confidence intervals were high indicating
small sample size of captures in terms of total effort. Home range of jackal is unknown for the area and hence it could exceed the subunit area sampled such that occupancy estimates represent an area that is used by the species (MacKenzie et al., 2006). Occupancy and abundance estimates are comparable to the other studies (Gupta 2011). Assumption of Royle and Nichols model (2003) showed that animal detected on one site will not be detected in another site, and instead making the assumption that the abundance at each site at any given point remains constant, irrespective of immigration or emigration to or from the site, the estimate of the animal specific detection probability is still very low (Gopalaswamy, 2006). Although heterogeneity could be accounted for indirectly, in traditional methods, by inclusion of covariates in a model for \( p \) (capture probability), it is not always possible to observe covariates that are correlated with abundance. The benefit of occupancy approach (Royle and Nichols 2003) is that it is based on the direct linkage between \( p \) and local abundance \( N \), which is a consequence of binomial sampling. The approach uses the data from occupancy surveys to draw inferences not only about proportional occupancy of sample units, but also about abundance in some situations.

It is believed that occupancy surveys also will be useful in meta-population studies involving multiple visits to many different sites among which animal movement is hypothesized (Royle and Nichols 2003). The method provides a means of properly estimating occupancy probabilities and associated functional relationships when there is heterogeneity of detection probabilities associated with variation in abundance over patches or sampling units. In addition to reasonable estimation of occupancy rates, the information on the distribution of abundance over sampling units may be very useful in meta-population studies as well (Royle and Nichols 2003). Meta population models range from very simple to fairly complex. Under the simplest models (e.g., Levins 1969, 1970), all patches are assumed to be equal with respect to the parameters governing patch occupancy dynamics, local colonization, and extinction probabilities. Perhaps the most detailed meta-population models involve the use of structured multisite matrix models that model the abundance of each age or stage class at each location as functions of class and location specific vital rates and rates of movement (Rogers 1966, Le Bras 1971, Schoen 1988, Lebreton 1996, Lebreton et al. 2000). Models of intermediate complexity and detail include those of Gyllenberg and Hanski (1992) and Gyllenberg et al. (1997), in which the abundance or population size of each patch is a state variable of interest. The method
provides means of estimating distribution and yield opportunities to test model predictions.

The results from the study by Gupta (2011) in the same study area showed that increase in number of sites had a little effect on variability of parameter estimates. This is because increase in home range sizes of species have added more spatial replicates to each site/location for analysis, thus there was reduction in number of available sites for analysis which caused little variability. Jackal seemed to be a generalist with high occupancy in Sariska where, variation in detection rate was observed across the four blocks. High abundance of jackal in open scrub habitat was documented by Mukherjee (1998) and Gupta (2011). Jackal spatial distribution in the dry season could have been clumped towards water sources resulting in its low site occupancy in summer. Estimates of carnivores from available studies were mostly photographic indices that are only an index of abundance. Much work on lesser carnivores have used night walks along established trails to estimate encounter rates or densities, however in many areas, these may be time consuming and labour-intensive and preclude wider spatial coverage (Kalle, 2013). In this study, occupancy rate under both the models and abundances were expected to be close to those estimated. However, fair estimates with relatively high standard errors can be expected when the numbers of detection sites are clumped or close to each other as revealed in the present study. Jackals were found to be abundant in number based on direct sightings but showed low capture rates in camera traps. This may be attributed to not using of trails/roads frequently by this species in the study area. Animal specific detection probability \((r)\) for all the season showed variability (0.03 to 0.06) and abundance range from 14.46 to 29.31. When the value of abundance was very high (>30), the site specific detection probability was less sensitive to change the abundance (Gopalaswamy, 2006). Hence it is suggested to use Royle and Nichols (2003) model with respect to occupancy as compared to the Mac Kenzie \textit{et al.}, (2002) model which implicitly assumes that sites have a constant or nearly constant abundance.

Nevertheless this study will serve as scale database against which future estimates from the same area/habitats/landscape can be compared and refined. This study showed that the models can indeed be used to estimate abundance for jackal for
which conventional methods cannot be used. To understand jackal ecology and status, future monitoring should consider comparative methods to assess abundance with field methodological modifications such as howling (Jaeger et. al., 1996).

The use of a simple Relative Abundance Index (RAI) based on camera-trap encounter rates for ecological studies is controversial particularly when a large number of variables (e.g. body size, average group-size, behavior) are likely to affect trapping rates and detection probability and thus confound the relationship with actual abundance (Carbone et al., 2001, Jennelle et al., 2002, Treves et al., 2010). However, there was an increasing evidence for a linear relationship between RAI and abundance estimated through more rigorous methodologies (Rovero and Marshall 2009). I used scent stations using bobcat lures to address the issues with the variable body size and behavior in photo trapping success. Camera trapping rate proved to be a useful index of abundance for jackal in this study. However further comparative studies over different habitats, forest types at the landscape scale may aid in standardizing this technique. It is intuitive that camera trapping rate should be related to abundance. As density increases, the chance of encounters between individuals and cameras would be expected to increase. The likelihood of the observed relationship between camera trapping rates and density (from line transect) was applicable in forest ungulates (Rovero and Marshall 2009). This method has the potential for temporal comparison of populations and may facilitate to standardize and reduce costs of monitoring programmes. However different camera trap models would have different detection probabilities that could affect trapping rates which should be considered. However, the performance of the camera traps used in present study were satisfactory having average of 1032.2 ± 473.6 pictures/camera trap/day obtained in each block seasonally. High standard error was accounted for variability in detection rate at different location and climatic conditions such as rain and wind resulting in waste captures.

Jackals are observed to be more patchily distributed than large species in the study area. Sariska is a relatively large contiguous semi-arid forest system now having eight villages inside the Intensive Study Area (ISA), inducing high human and livestock disturbance. The border of Sariska is surrounded by crop fields and small towns which are strongholds of crop land, open scrub and human generated waste that
have great potential in having high rodent densities and other food source in terms of garbage. Although the study area is under tremendous pressure from anthropogenic activities, jackal seem to be tolerant to some degree of habitat alteration however, through long-term monitoring it is necessary to understand their spatial responses towards the degree of human disturbance and weather. Protected Area (PA) provides them cover. The results of this study showed that the occupancy-based Royle and Nichols (2003) can provide useful tool for the estimation of abundance of non-uniquely identifiable and cryptic species. Parameter estimates with high standard errors have high uncertain estimates, and therefore reliable estimates of absolute abundance may not be obtained. However, it is still possible to arrive at an index of abundance that takes detection probability into account. Such an index is comparable across temporal scales for instance, and can provide important insights into population trends.

Jackals were not widely dispersed across sampling locations, but clustered in a few locations in the ISA. Sites may not be independent, especially if sites are adjacent (such as in a grid) because animals leaving one site will enter another site by necessity (Dail and Madsen 2010). One way to model this dependence may be to include the abundance at surrounding sites, weighted by distance. An alternative is to use Bayesian hierarchical modeling and the spatial dependence models developed by Royle et al. (2007). However, incorporating these models has proved difficult, so this remains an area of active research. Second, the number of animals surviving and staying at a site will not be a binomial random variable if clusters of animals survive or emigrate together. Although, Royle (2008) gives a framework for modeling the effect of cluster size in wildlife surveys, patch occupancy provide extremely useful tools for the detection of trends in wildlife population abundance and species presence/absence (Mackenzie et al., 2006). The main advantages they offer are explicit treatment of detection probabilities, error assessments, and estimation of confidence intervals (Mackenzie et al., 2006). The inter-annual variation as well as seasonal variation in abundance of jackal could be related to changes in resource abundance, shifts in habitat use or other non measured biotic/abiotic factors such as competition with dominant predators, suggesting that future studies on population monitoring must be continued over time taking these unmeasured variables into account. I recommend that in order to efficiently document abundance camera-
trapping will need to be augmented by other methods such as spotlight surveys and howling stations to detect jackal presence and territorial behavior. This study confirms that Sariska Tiger Reserve harbors fair abundance of jackal population, an opportunistic predator surviving in presence of large carnivore like tiger, leopard and hyena.