SPATIAL PATTERN IN THE OCCUPANCY AND ABUNDANCE OF RED JUNGLEFOWL *GALLUS GALLUS* IN WESTERN SHIVALIK LANDSCAPE

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DOCTOR OF PHILOSOPHY IN
WILDLIFE SCIENCES
BY

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Wildlife Institute of India

CERTIFICATE

This is to certify that the thesis of Mr Merwyn Fernandes entitled "**Spatial patterns in the occupancy and abundance of Red Junglefowl *Gallus gallus* in western Shivalik Landscape**" is an original piece of work submitted to the Saurashtra University, Rajkot (Gujarat), for the award of the **Doctor of Philosophy in Wildlife Science.**

Mr Merwyn Fernandes has put in more than six terms of research work embodied in this thesis under my guidance and supervision. The work presented in this thesis has not been submitted for any other degree of any other University or Institution

Place: Dehradun

Date: 24th March 2014

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SUMMARY

Red Junglefowl *Gallus gallus* is a pheasant that represents some of the most charismatic but threatened bird species of all avian groups. This is principally due to hybridisation, over-exploitation – either through hunting for food or for their plumage and habitat fragmentation and degradation. Conservation measures are all the more urgent since it may be considered as a ‘barometers’ or indicators of the status (or ‘health’) of certain forest habitats. Unfortunately, detailed quantitative data regarding many aspects of its ecology, in particular, factors that influence patterns of abundance, habitat occupancy and its present distribution are sorely lacking. In order to supplement the existing knowledge on the species ecological requirements, population estimates and distribution was carried out in the western Shiwalik Landscape of India. The goal of this study was to identify landscape-level patterns of distribution, abundance and habitat occupancy of *G. gallus* across different spatial and temporal scales in a forest landscape, to help guide strategies for Galliformes conservation management. It would be interesting to know whether habitat occupancy by *G. gallus* differs across the breeding and non-breeding season and what are the factors that significantly influence the species abundance.

The present study was undertaken in an area (29˚54’ to 30˚24N and 77˚32’ to 79˚12’ E) that is situated within the Shiwalik Hills of Northwestern India, in the districts of Dehradun and Haridwar in Uttarakhand State. This linear landscape which is *ca.*100 km long is not more than 25 km wide and is oriented along the northwestern-southeastern direction. The total area measures *ca* 802 km^2 of which 584 km^2 falls under the protected area network – namely the western section of Rajaji National Park (henceforth Rajaji NP), 119 km^2 in Dehradun Forest Division (hereafter Dehradun FD) and 99 km^2 within the Timli Range of the Soil Conservation Department.

G. gallus is listed as a ‘Least Concern’ species, making an assessment of their status requires reliable information on population size and distributions. In total, 105 sites were sampled during the non-breeding and breeding season. The detection probability for the species was higher during the breeding season than compared to the non-breeding

with shrub height and protection status being the most important factors. For the non-breeding season, terrain was the most important determinant of for the species occupancy, while for the breeding season undergrowth cover and disturbance explained occupancy for the species. While, comparing the occupancy rates between the non-breeding and breeding season there were no changes in 47% of the grids, positive change in 16% and a negative change in 37% of the grids.

Understanding the relative importance of different factors in producing the abundance pattern is essential for protecting the habitat in order to maintain the populations at the landscape level. Following a total effort of ~208.4 km of walk across the two seasons, a total of 260 individuals during the breeding season and 251 individuals during the non-breeding season was encountered. The results indicate that there were no significant difference in the encounter rate, group size and individual density. The median group size was 2 and no such differences across the season for *G.gallus*. The estimated half-strip width for the non-breeding season was lesser than compared to the breeding season. In order to assess the factors influencing abundance at multiple scales an intensive study area was selected. From a total effort of 129 km, there were a total of 64 segment detections out of 115, with 283 individuals being detected. Issues of multi-collinearity were address by eliminating correlating variables. At the local scale there were three variables selected. The best model included an interactive model with canopy cover: shrub height and canopy cover: tree height. At the landscape scale, the best model was an interactive variable of NDVI: terrain ruggedness, NDVI and terrain ruggedness.

The Red Junglefowl has an extent of occurrence of about 5,100,000 km^2. One of the subspecies *G g murghi* has its distribution within India. In order to address the issues of status and distribution we resorted to using presence-only models. These models overcome the cost and time constraints when dealing with a large ranging species. Species site locations were all collated by using primary field data, network of field biologist, literature records, museum specimens and archived databases. A total of 511 geo-rectified data points were used (357 records training and the remaining for testing), along with predictable variables such as bioclimatic factors, digital elevation model and forest cover. These variables were used to run maximum entropy models using the product function, the test data has an AUC score of 0.950 to 0.978, the jackknife test for variable importance was precipitation in the dry quarter was the most important variable having a contribution of 36.2% followed by annual mean temperature having a contribution of 16.6%. The total predicted probability suitable area in India is approx 189,492 km^2. There are three distinct landscapes within India namely north (12%), central (52%) and northeast (36%).The central landscape is isolated and is not connect to the north or northeast landscape. The north and northeast landscape is connected to each other through the forest patches in Nepal and Bhutan. The PA network in India, accounts for nearly 13% of the area (47,648.98 km^2), where the NPs represent 4.32% (15,335.84 km^2) and the WLSs 8.52% (30,257.13 km^2).Within the high suitability area nearly 19% of the total area lies within the PA network of India, while the rest of the area is outside the purview of the PA network. The species is still reported from 205 districts out of the 270 districts in range 21 states.

The structural component of understorey was one of the most important factor that determined occupancy and detection probabilities for the species. The structural components are known to provide cover to *G.gallus,* and other gallinaceous species that nest on the forest floor. Strategies or plans are necessary which may look at regeneration as a possible alternative but keeping in mind that apart from the structural component, resource availability is also strongly correlated to the understorey. The timing of understorey management should take into consideration the breeding season for this species. There is a need to increase efforts to understand whether the species is prevalent in forested tracts outside the PA network. While, states that have restricted distribution *viz;* Sikkim, Andhra Pradesh, Maharashtra and Punjab and fragmented distribution *viz;* Haryana, Bihar and Uttar Pradesh there needs to be increased efforts to map the distribution of the species, especially at the range boundaries. The main threat to *G. gallus* is hybridisation and hence there is an urgent need to assess ranging patterns, survivorship and other basic demographic parameters, population status, genetic variability and purity of *G.gallus* in areas adjoining PAs where most of these forests have multiple uses and domestic/feral fowls are in close proximity in order to keep the common species common.

CHAPTER 1

INTRODUCTION

1. Background

The present dominance and resilience of the human race has allowed it to colonise most of earth's ice-free land surface (Vitousek et al. 1997). With varying levels of presence, another dimension and stages of interactions adds greater complexity and therefore agents of change (Ellis 2011). It is this mobility, human demography (Cincotta 2000), structured engineering and their synergistic effects (Brook et al., 2008) that have brought about habitat loss (Geist and Lambin 2002), fragmentation (Foley et al. 2005) and species introduction which have been reflected as the principal drivers (Rhymer and Simberloff 1996) along with overkill as the "evil quartet" (Diamond 1989) of species extinction. These may have caused many more species to reach their critical threshold levels (Simberloff 1986). Proactive response to ameliorate impacts have been proposed keeping in mind scenarios (Brooks et al. 2006). But the fundamental question that prevailed is from where do we start from whom? Even if biodiversity is reduced conceptually to the species level, quantifying and managing it would normally be extremely time consuming and expensive. One of the most effective way in which such a scenario has been framed is by knowing how many species are available within a particular area - a certain form of "irreplaceability" (Pressey 1999, Ferrier et al. 2000). One of the descriptor of any given taxonomically defined assemblage is the species-abundance distribution (SAD), when this relationship is graphed we get a hollow-curve. It is this arithmetic relationship (a vector of abundance of all species in the community) that is regarded as one of ecology's universal laws (Gleason 1929). On the horizontal axis there is a huge accumulation of species near the abscissa, the less abundant namely the "rare species" while at the further end of the axis are species that are abundant also called as 'common species'. The relevance of this relationship is being effectively used to slower down disboscation and prioritising sites for preserving biological diversity (Soule 1986) *in sensu* protected area network. However, the underlining mechanism of this very concept irreplaceability, has evolved from early 1970's (eg. Goldsmith 1975) where it is now central to the theory of spatial conservation planning (Margules and Pressey 2000) in order to maintain a minimum 10 percent target set at the Convention of Biological Diversity (CBD) that is critical for the functioning of earth's life support system (www.cbd.int).

How species have coped with these changes that lead to its extirpation is of concern and forms the fundamentals of conservation biology. But, such understandings have been lacking and ignored at the scale of the species range. Depending on whether these processes are localised or widespread there are two different hypotheses for range extinctions, the demographic hypothesis and contagion hypothesis. The contagion hypothesis makes the implicit assumptions that the extinction factor must extend across the entire range of the species and is uniformly distributed because any portion that is not impacted by the extinction factor would expect the species to persist. In addition, the intensity of the factor must be uniformly distributed, or else peripheral populations might persist where the intensity of the factor is reduced. The other important aspects of the factor should be synchronous and sudden onset. If the onset is staggered then, those populations that are last impacted regardless of their location may persist longer regardless of their historic range size. The widespread colonisation and domination of the human race has formulated this contagion through a suite of disturbances (Channell and Lomolino 2000) creating landscapes with a "variegated" effect (McIntyre and Barrett 1992).

Effective conservation of a species in such variegated habitats requires a broader landscape or regional perspective (for eg. Radford et al. 2005, Pearce et al. 2008). This involves examining habitat patches as part of a larger landscape that includes patches of various sizes (eg. Andren, 1994), patch isolation (eg. Stouffer and Bierregaard 1995), microhabitat features (eg. Sisk et al. 1997), and the role of the surrounding 'matrix' (eg. Fahrig and Merriam 1994, Fischer et al. 2005). Many recent studies have pointed out the importance of the landscape matrix in animal population changes (eg. Prugh et al. 2008), reserve planning (Lord and Norton, 1990), and management practices (Franklin and Lindenmayer, 2009). The importance of the matrix is often demonstrated by the strong correlation between the abundances of species in the matrix habitat and their presence in focal habitat patches (eg. Estades and Temple 1999, Gascon et al. 1999). This indicates that the 'nature' of the matrix habitat can determine how well habitat patches will maintain viable species populations in the long term (eg. Ricketts 2001, Tabarelli and Gascon 2005).

1.1. Landscape – a matrix of habitats

Across variegated landscapes, the inability of habitat specialists to utilise the surrounding matrix may be an important factor in population declines and localised extinctions of threatened species, particularly for populations of forest-dependent species (eg. Bierregaard and Stouffer 1997, Antongiovanni and Metzger 2005). In many cases, forest habitat is often modified, but not destroyed (McIntyre and Hobbs 1999), leaving a gradient of habitat features e.g. scattered trees and smaller forest patches. Some forest-dependent species can perceive such landscapes as gradients of suitable habitat, rather than a simple 'binary landscape' of habitat and non-habitat (Manning et al. 2004). Thus it is imperative to examine habitat patch configuration within the landscape since various ecological, geographical and climatic factors can collectively influence patterns of habitat patch occupancy of species at multiple spatial and temporal scales (Levin 1992, Gaston and Blackburn 1995, Ferrier et al. 2002, Rushton et al. 2004).

Even if biodiversity is reduced conceptually to the species level, quantifying and managing it would normally be extremely time consuming and expensive. Hence, preferentially endangered and threatened species are the focus since when confronted with anthropogenic perturbation they are generally among the first to go locally extinct (Woodroffe and Ginsberg 1998). Relatively, common species perform better with most pressures and hence can be (Noss 1996, Lambeck 1997, Coppolillo et al. 2004) and have been used in a working landscape (Githiru et al. 2007). Although, commonness being a rare phenomenon has received very little attention through the years there is increased focus in understanding the role and function that they play in the community (Gaston 2009). In part this has been driven simply by a recognition of key gaps in understanding, but it has also followed from deepening concerns over declines of many common species and the consequences these declines might have (Gaston 2011). In order to understand reasons for such declines is to understand factors that govern the distribution of the species.

1.2 Birds

Birds being prominent representatives have been effectively used as a community (Raman, 2004), or at the species level (Jeganathan et al. 2004) in understanding habitat relationships and conservation at a landscape level (Pearce et al.

2008, Jayapal et al. 2009) through properly marked distribution (Eken et al. 2004). Recent studies have shown that the multi-scale approach is needed, since habitat suitability is related to factors that operate on different spatial scales. Thus, there has been considerable focus on documenting species distribution patterns and identifying the underlying processes that govern them. A consequence has been a realisation that the way a species is distributed is inherent to the ecological and geographical factors operating at multiple spatial and temporal scales (Gottschalk et al. 2005). Knowledge about these factors is fundamental in understanding patterns and processes influencing the distribution of the species (Lawton 1996, Gaston and Blackburn 1999, Ferrier 2002, Ferrier et al. 2002) supplementary to identifying the internal distribution of abundance within range boundaries. Indeed, large variations in abundance can result in gaps within a species range that can create internal borders (Brown et al. 1996). The characterisation of species ranges is, however, complicated by their spatial and temporal dynamics (Fortin et al. 2005). Climatic and historical factors might explain distribution patterns at a regional scale while biotic interactions play an important role at the local scale (Ricklefs 1987). Any species sensitivity to changes at a particular scale is also a function of its specialisation at that particular scale (Howell et al. 2000). Hence the quantification of spatial pattern of the outer species border limit, as well as within-species range boundaries, may help to resolve outstanding questions in spatial ecology (Pulliam, 1988).

1.2a. Galliformes -birds of a paradox

Galliformes is one of the most threatened avian orders, with nearly 25% of the 300 species considered to be at risk of extinction compared with the overall 12% of all birds in the world (McGowan and Garson 2002). This group of birds is represented by partridges, francolins, quails, snowcocks and megapodes. The word ‘Pheasants’ are referred to those members of subfamily Phasianinae, typified by greater sexual dichromatism, dimorphism and largely possess chicken-like morphological and behavioural traits (Delacour 1977). These are large, ground dwelling birds with brightly coloured plumage that represent the family Phasianidae of the order Galliformes. Sixty-seven species (including subspecies) belonging to 20 genera are reported in the world (Madge and McGowan 2002), except the Congo peafowl *Afropavo congensis*, rest all are asian in origin (Kimball et al. 1997, Stidham 2008).

Currently, 34 species of family Pheasanidae are listed in the IUCN as threatened birds (McGowan and Garson 1995, Fuller and Garson 2000).

There are 17 species of pheasants reported within India and only one genus *Gallus* is represented throughout except for the Islands of Lakshadweep and Andaman and Nicobar (Johnsgard 1986). The name '*Gallus*' is due to the fleshy erect comb on the crown of the head on the male bird. These birds are medium sized tropical birds that are highly dimorphic where the members of this family are called Junglefowl. The genus *Gallus* is represented by four species namely Green Junglefowl *Gallus varius*, Grey Junglefowl, Ceylon Junglefowl *Gallus lafayettii* and Red Junglefowl *Gallus gallus* confined to the Indomalayan sub-region of which the latter three occur within the Indian subcontinent, whereas the Green Junglefowl is restricted to Java and the neighbouring island eastwards (Fig 1.1a). Out of these four species, three *viz*. green, grey and ceylon are endemic and have restricted distribution, whereas the Red Junglefowl is widely distributed, having distinct clinal intergradation between five recognised subspecies. These subspecies differ chiefly in colour and shape of the neck hackles in male and size and colour of the facial lappet (Johnsgard 1986). The five subspecies that are recognised are Cochin-Chinese Red Junglefowl, Burmese Red Junglefowl, Javan Red Junglefowl, Indian Red Junglefowl and Tonkinese Red Junglefowl (Fig 1.1b). The distribution and characteristic of these five subspecies of Red Junglefowl is given in Table 1.1.

1.2b. The Red Junglefowl

The Red Junglefowl males are more conspicuous than the females, by presence of characteristic red comb, wattle, golden yellow hackles with a prominent black streak, blackish coloured breast, two long metallic green sickle shaped tail feathers with a white patch present at the rump. The female is slender, thin and smaller in size (41-46 cm) than compared to the male (65-78 cm). The female has elongated golden-buff, black centred, feathers across the nape and mantle with rudimentary or no comb. The rest of the upperparts are rufous brown, finely vermiculated with black and has rufous brown under parts streaked with buff. The males and females have reddish to pale white ear lappets colour, blackish brown, slender, smooth and thin legs with a spur present in the males. The first year male is superficially similar to adult male, but much duller and hackles less developed. The chicks are precocial when born, and distinguished from the domestic chicks by presence of blackish-brown eye stripe (Ali and Ripley 1983). They prefer well watered areas around *nullahs* (streams) and in moist mixed forests below 2,500m and have been recorded in mangrove forests and scrub jungles with patches of cultivation (Madge and McGowan 2002).

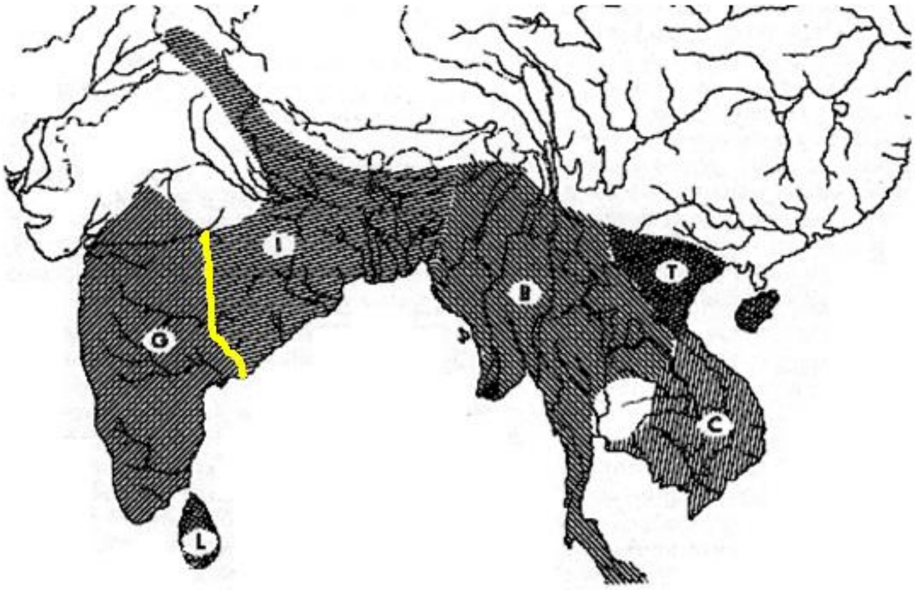
Table 1.1. Distribution and characteristics of the five subspecies of Red Junglefowl

S. No.	Species	Distribution	Characters
1	Cochin-Chinese RJF *G. g .gallus* Linnaeus 1758	Cambodia, C and S Vietnam, C and S Laos and E Thailand	The neck hackles of cock are long and golden orange to bright red and ear lappets large and white
2	Burmese RJF *G. g. spadiceus* Bonnatere 1791	SW Yunnan (China) and northeastern parts of Arunachal Pradesh and in other NE states (India), along the Indo-Myanmar border, Myanmar, Thailand, peninsular Malaysia and N Sumatra	The neck hackles are golden yellow, comb and facial lappet red and all are short compared to *G. g .gallus.*
3	Javan RJF *G. g. bankiva* Temminck 1813	South Sumatra, Java and Bali.	The neck hackles are rounded, golden-yellow and even shorter than *G. g. spadiceus*
4	Indian RJF *G. g. murghi* Robinson and Kloss 1920	India, Nepal, Bhutan and Bangladesh	The neck hackles are yellow, longer with a black shaft streaks, ear lappets pinkish and small, hen is paler than other race.
5	Tonkinese RJF *G. g. jabouillei* Delacour and Kinnear 1928	North Vietnam, SE Yunnan and Hainan	These are darker and redder than *G.g. gallus*. Their hackles are short, less pointed and golden yellow. Facial lappet and comb are short and all red.

Source Johnsgard 1986

Fig 1.1a: Global Distribution of Junglefowl (*Gallus* spp) and its subspecies.

Source: Johnsgard (1986)



G = Grey Junglefowl, L= Ceylon Junglefowl, B= Burmese Red Junglefowl, T=Tonkinese Junglefowl, C= Cochin-Chinese Junglefowl.

Fig1.1b. The five subspecies of Red Junglefowl *Gallus gallus*

Photo: John Corder Source: Tring Museum. U.K.

1. *G.g. jabouillei*, 2. *G.g. bankiva*, 3. *G.g. gallus*, 4. *G.g. spadiceus* and 5. *G.g. murghi*

They are generally found in small groups or parties, with one cock to several hens. They are very shy, secretive birds that scurry into cover on slightest disturbance, and are known to fly considerable distances when flushed. Usually, found during dawn and dusk in the open forest tracts, trails, fire lines and fields abutting the forest where they come to feed (Ali and Ripley 1983). During the day, they are found in the thickets of bushes and at noon often resting in the shade of shrubberies. They roost in trees and bamboo clumps. Nests are built on the forest floor, within thickets of the shrubs. The clutch size varies from 4-7 eggs per clutch and the incubation period is 20-21 days (Ali and Ripley 1983, Madge and McGowan 2002 and Rasmussen and Anderton 2005).

In terms of threats perception to the species, the International Union for Conservation of Nature (IUCN) has listed it as “Least Concern” while the Indian Wildlife (Protection) Act, 1972 has listed it in Schedule IV. Convention of International Trade in Endangered Species of Wild Fauna and Flora (CITES) has not listed the species.

1.3. Previous research on Red Junglefowl

There is considerable archaeological evidence to suggest that *G.gallus* were probably domesticated as early as 5400 BCE (West and Zhou, 1988) with both hypothesis ie., monophyletic (Fumihito et al. 1996) and multiple species (Liu et al. 2006 and Kanginakudru et al. 2008) suggesting that RJF plays a major role as a progenitors for modern day breeds (Hillel et al. 2003, Moiseyeva 1998) while molecular evidence suggest that other *Gallus* species have contributed to some breeds (Nishibori et al. 2005). The close proximity of feral/domestic breed and the effective gene flow to and from the wild counterparts has raised concerns in regards to the purity, possibly loss of the progenitor (Hanotte and Jianlin 2005) and the effect of hybridised individuals in the system especially hybridisation, a result of human intervention is of concern (Randi 2008). It is imperative that we conserve or identify the non-hybridised individuals, however, this would largely be impossible due to hybrid swamps and to tease out whether the 5% or less proportion of hybridisation is an effect of admixture or natural (Brisbin and Peterson 2007). The last two concerns have been highlighted in this species (Brisbin 1995, Peterson and Brisbin 1999, Brisbin et al. 2002 and Brisbin and Peterson 2007). Hence, Kaul et al (2004) evaluated captive species with reference to phenotypic characters while subsequently molecular analysis (Sathyakumar et al 2012) was undertaken for captive and free ranging birds. However, the pure wild *G.gallus* represent a pool of genes for future breeding and research purposes.

Despite the species well-documented historical relationships with humans (Storey et al., 2012), very little quantitative data exist regarding various aspects of its ecology due to their extreme cautious nature (Collias and Collias 1967, Hakansson and Jensen 2005). Most behavioural inferences have been drawn from birds housed in various zoological parks, pheasantries or at research stations outside their native distributional range (Collias et al. 1994 and Collias and Collias 1996). In such ambience, where there is a relaxation of natural selection pressure Schutz et al. (2001) has shown a modification of the natural behaviour towards less energy demanding strategies, such behavioural differences are a cause of concern (Hakansson and Jensen 2005).

In polygamous species, where males typically do not provide paternal care, resources and where their contribution to reproduction is assumed to be limited to the provision of semen has led to the hypothesis that female preference may be explained by genetic benefits (Birkhead and Parker 1997, Parker 2003). Preference for socially dominant males may thus be driven by the pursuit of alleles associated with superior fitness. However, there is some evidence that social dominance can be transmitted from parents to offspring in some species (Craig et al. 1965). Theoretically, males may influence female reproductive success directly even without providing paternal care (Reynolds and Gross 1990, Sheldon 1994). It has been reported that both sexes are to some degree sexually promiscuous probably due to some degree of mating propensity by males and females (McBride et al. 1969, Thornhill 1988, Collias and Collias 1996, Ebenhard 1996, Etches 1996, Pizzari 2001). The role accorded to females (utilisation of male gametes) with respect to sexual selection and thereby the consequences of co-evolution in male traits and the nature of female attention to and assessment of males are of considerable importance (Janetos 1980, Bradbury and Andersson 1987, Andersson and Iwasa 1996). Although male competition and female selection often operate in unison, part of the variance in male reproductive success is often explained by variation in a male’s ability to compete for copulation (Darwin 1868) by variation in fertilisation opportunities (Birkhead and Pizzari 2002, Pizzari and Birkhead 2002) and sperm investment (Pizzari et al. 2003). However, it is well established that females that attend to and choose mates during courtship also consistently prefer to be inseminated by and use the sperms of certain males (Pizzari and Birkhead 2000, Birkhead and Pizzari 2002). This may not always be the case as the female is known to mate with multiple males (Ligon and Zwartjes 1995). As mating strategies change due to the change in the value of copulation over time, selection on males to mate at a higher rate than females often results in male harassment of females and counteracting female responses (Pizzari et al. 2002, Lovlie and Pizzari 2007). One potential explanation of this result is that while the mating propensity of individual males may increase with more males in a population, male interference and inhibition may also increase as more males compete over fewer copulation opportunities. This mechanism may be particularly relevant for social species such as the junglefowl, where socially dominant males are able to interrupt or altogether inhibit the sexual investment of subordinate males (Pizzari 2001).

The species has been extensively used as a model to elucidate the complex behavioural mechanisms of mate choice and evolutionary patterns of evolution using phenotypic characters. In particular, *G. gallus* research has greatly improved our understanding of sexual selection using experiments with captive birds examining: mate choice selection for males (Zuk et al. 1990a, Ligon et al. 1990, Parker and Ligon 2002) or females (Johnson et al. 1993, Zuk et al. 1995b) and whether these choices are repeated (Johnsen and Zuk 1996), the effect of plumage ornamentation (Zuk 1990c, Zuk 1993, Ligon et al. 1998); and the influence of parasite load on mate-selection and endocrinology (Zuk 1990b, Zuk 1990d, Zuk et al. 1995a). Other research has focussed in various aspects of the species vocalisations (Chappell et al. 1995, Arshad et al. 2000b, Wilson and Evans 2008), food and feeding habits (Arshad et al. 2000a), roosting behaviour (Arshad et al. 2001), breeding behaviour (Arshad and Zakaria 1999), plumage variations within hybrids (Morejohn 1953, 1955, 1968 a, b) and differences in behaviour and morphology within captive and free ranging birds (Hakansson and Jensen 2005) due to relaxed natural selection pressure (Schutz et al. 2001).

In India, the earliest known mention of the species is from Tuzuk-e-Jehangiri written between 1605 and 1623 by the Mughal emperor Jehangir (Beveridge 1909) with subsequent contributions to the species knowledge by naturalist and ornithologist

(Hume and Marshall 1879, Beebe 1922, Baker 1920, Bump and Bohl 1961, Ali and Ripley 1983, Madge and McGowan 2002). Collias and Collias (1967) were perhaps the first who undertook studies reporting population and behavioural account of the species in the forest of Dehradun. Kalsi (1992) studied the habitat use of this species in Kalesar Forest Reserve of Haryana and suggested that the species preferred mixed forest with cultivation more then other habitat, a view also endorsed later by Javed and Rahmani (2000) in Dudhwa National Park. The species had higher encounter rae in unlogged forest of western Arunachal Pradesh (Dutta,2000) while preferring densely vegetated areas on a sloping altitudinal gradient is preferred in Deva Vatsala National Park (Subhani et al. 2010). All the above mention studies also report incomparable abundance estimates except Azar et al. (2010) where efficiency between the fixed-width and line transect was evaluated and Harihar and Fernandes (2011) considered the effects of season on abundance estimates. There are other studies which documented the geographic distribution of the species (Fernandes et al., 2009).

1.4 Objectives

The goal of this study is to identify landscape-level patterns of distribution, abundance and habitat occupancy of *G. gallus* across different spatial and temporal scales in a forest landscape, to help guide strategies for Galliformes conservation management. It would be interesting to know whether habitat occupancy by *G. gallus* differs across the breeding and non-breeding season and what are the factors that significantly influence the species abundance. To achieve this goal, the following objectives are laid down:

1. To estimate site occupancy of *Gallus gallus* and identify factors that influences their occupancy.

2. To identify factors those that influence the species abundance across the landscape

3. To quantify the distribution of the species and suggest important sites for conservation.

1.5. Organisation of the thesis

The findings of the field study that spanned for three years is summarised into five chapters *viz*: 1. Introduction, 2. Study Area, 3. Spatial patterns in the occupancy across seasons, 4.Determinants in the pattern of Abundance and 5. Species distribution.

Chapter 1 gives a general introduction to the study and the species the Red Junglefowl *Gallus gallus* based on a complete literature review of the species. Chapter 2 summarises the physical (abiotic) attributes and biological (biotic) attributes including the people residing within the area. This has been done by collating information from literature of the area. Chapter 3 deals with detection probabilities and site occupancy of the species across the non-breeding (2009) and breeding (2010) seasons within a landscape. Chapter 4 gives the abundance estimate for the area and compares factors influencing abundance at the local and landscape level. Chapter 5 delves in modelling a presence-only model for the species geographic distribution. Chapter 6 Synthesis the findings and suggests recommendations. Finally, the literature cited from all chapters has been compiled together alphabetically and chronologically in the reference section.

Plate 1: Male and Female Red Junglefowl

CHAPTER 2

STUDY AREA

2.1. Physical Attributes

2.1a. Location

The present study was undertaken in an area (N 29˚54’ to 30˚24 latitudes and E 77˚32’ to 79˚12’ longitudes) that is situated within the Shiwalik Hills of Northwestern India, in the districts of Dehradun and Haridwar in Uttarakhand State. For most part, the Shiwalik Hills is continuous with the outer Himalayas, except few places such as the Doon Valley, where the hills are truly represented. This area is uniquely poised lying at the juncture of three biotic provinces namely the Himalaya - Western Himalayan (2B), Semi-Arid- Punjab Plains (4A) and Gangetic Plain - Upper Gangetic Plain (7A) (Rodgers et al. 2000). This landscape thus linear in form is bound on either sides by rivers; namely Yamuna (west) and Ganga (east) and sprawling urban cities/towns and industrial townships in the north to southeastern part (Dehradun, Rishikesh and Haridwar) along with numerous villages skirted all along the southern border (Fig 2.1).

2.1b. Area

This linear landscape which is *ca*.100 km long is not more than 25 km wide and is oriented along the northwestern-southeastern direction. The total area measures *ca* 802 km^2 of which 584 km^2 falls under the protected area network – namely the western section of Rajaji National Park (henceforth Rajaji NP), 119 km^2 in Dehradun Forest Division (hereafter Dehradun FD) and 99km^2 within the Timli Range of the Soil Conservation Department. The Rajaji NP can be broadly sub-divided into the north facing section (250 km^2) and south facing section (334 km^2), while the Dehradun FD comprises of the following ranges Asarori (44km^2), Malhan (75 km^2). For the intensive study the western section of Rajaji NP, Asarori and Malhan ranges of Deharadun FD and Timli range of Soil Conservation Department was selected

2.1c. Topography

The terrain is undulating with an elevation ranging from 300 to 1,200 m. The hills that are steep and rugged are along the Main Frontal Thrust have gentler northeastern facing slopes than slopes facing south-western while the plains are gentle. There are numerous ridges that run parallel to each creating many river beds termed as "*Raus*". The minimum elevation at the northern side of the hill range is higher than that of the southern side. The ridge peaks from the central part of the hill range; central part of the landscape has the area of the hill range has the peak elevation.

2.1d. Drainage structure

The drainage basins of two river systems (Yamuna and Ganga) flow through the area that is dominated by moist deciduous forest (following Wikramanayake et al. 2001). There are many *raus* that do not show surface flow throughout the year and hence dry except when they receive torrential waters during the monsoon; along with it large volume of boulders and loose shingles that are carried down. This undermines the banks making them wide, shallow and sometimes may cause shifts in the flow course. The *raus* on the north-eastern face of the hill discharge into the Suswa and Song rivers, while *raus* of the south-west face flow through the plain and eventually join river Yamuna. There are few *raus* that have surface water during the summers are important sources of water for the animal life (Anonymous 2012).

2.1e. Geology and Edaphic properties

This geologically young mountain chain of Himalayan orogeny comprises of Pleistocene molassic sediments of Muree and Shiwalik formation. There are variations in the rock types, where the sediments have been grouped into middle and upper Shiwalik type. While, the mid Shiwaliks have sandstones with interbed clays, the upper Shiwalik has conglomerate with thin interbedded clay and sandstone. The main elements of conglomerates consists of boulders and pebbles of quartzite, gneiss, Chandrapur shales, Blainis boulders and limestone with a mix of quartz, feldspar, mica hornblende, tourmaline etc. The clay is silty and micaceous allowing water to percolate to form streams. The sandstone is fine to coarse consisting of few mineral constituents. There has not been much effect due to compaction on recrystalisation of the cementing material thereby resulting in loose and soft nature of the conglomerates and sandstones and thus cohesion with the soil is loose. Along the river and floodplains the soil texture varies from loamy sand to sandy clay loam (Atkinson 1882).

2.1f. Seasonality

The study area has pronounced wet and dry season. The ‘wet’ season (June to October) with an average annual precipitation of 1500 mm and daily maximum temperatures range from ~25^{0}C to ~35^{0}C and the ‘dry’ season (November to May) with daily maximum temperature range from ~16^{0}C to ~40^{0}C receives less than 25% annual precipitation.

Table 2.1. Monthly average of climatic conditions in western Shiwalik for 2009 and 2010

Year	Month	T	TM	Tm	H	P	RA	TS	FG
2009	January	13.9	23	8.1	71.6	4.82	3	0	4
	February	16	25.3	9.5	62.3	32	4	3	0
	March	20.6	29.5	13.5	51.1	8.14	5	3	0
	April	26	35.3	17.7	38.6	31.25	4	3	0
	May	27.4	36.2	20.8	52.2	59.69	6	10	0
	June	30	37.9	23.2	47.5	117.35	7	7	0
	July	27.3	32.7	23.5	80.7	438.91	21	18	0
	August	26.3	31.7	22.9	86	453.38	23	11	1
	September	25.5	31.5	21.7	83.6	298.68	17	3	0
	October	20.6	30	14.2	70.6	109.22	3	3	0
	November	16.6	26.3	10.6	70.8	1.27	2	1	0
	December	13	22.6	7.3	70	0	1	0	0
2010	January	11.9	21.8	6.3	75.8	3.3	2	0	13
	February	15.7	24.5	9.2	64.3	56.89	4	2	0
	March	23	31.5	15.6	50.7	1.12	1	2	0
	April	28.3	37.3	19.4	35.2	3.56	2	3	0
	May	29.9	37.9	22.2	42	19.57	6	9	0
	June	29.8	37.7	22.3	50.5	133.08	6	8	0
	July	26.6	31.9	23.5	84.7	553.22	25	17	0
	August	25.9	30.8	23.3	89	980.43	27	16	0
	September	24.3	29.8	20.9	87	861.57	22	9	0
	October	22.4	30.4	16.8	68.7	15.24	5	2	0
	November	18.1	26.8	12.5	68.9	24.89	2	1	0
	December	12.7	22.4	7.2	70.9	40.9	3	2	0

Source: www.tutiempo.net/en/

T = Mean temperature (°C); TM = Maximum temperature (°C); Tm = Minimum temperature (°C); H = Mean humidity (%); P = Precipitation (mm); RA = Days of Rain or Drizzle; TS = Days of Thunder; FG = Days of Fog

2.2. Biological Attributes

2.2a. Flora

The principal forest vegetation categorised as Northern Indian Moist Deciduous Forest (Champion and Seth 1968) comprises a mosaic of dry and moist deciduous forests dominated by Sal *Shorea robusta* along with other characteristic tree species such as *Terminalia alata*, *Syzygium cumini*, *Anogeissus latifolia*, *Haldina cordifolia*, *Mallotus philippinensis Lagerstroemia parviflora* and *Cassia fistula* (Gautam et al 2008). Regional plantation forests are dominated by *Tectona grandis, Holoptelia integrifolia, Acacia catechu*, *Dalbergia sisoo*, *Ailanthus excelsa* and *Eucalyptus globules* (Kanjilal 1928). Dominant understorey vegetation includes *Ardesia Clerodendron viscosum, Carissa carandas, Colebrookia oppositifolia, Ehretia leavis*, *Murraya koenigii*, *Lantana camara*, and *Woodfordia fruticosa* (Babu 1977).

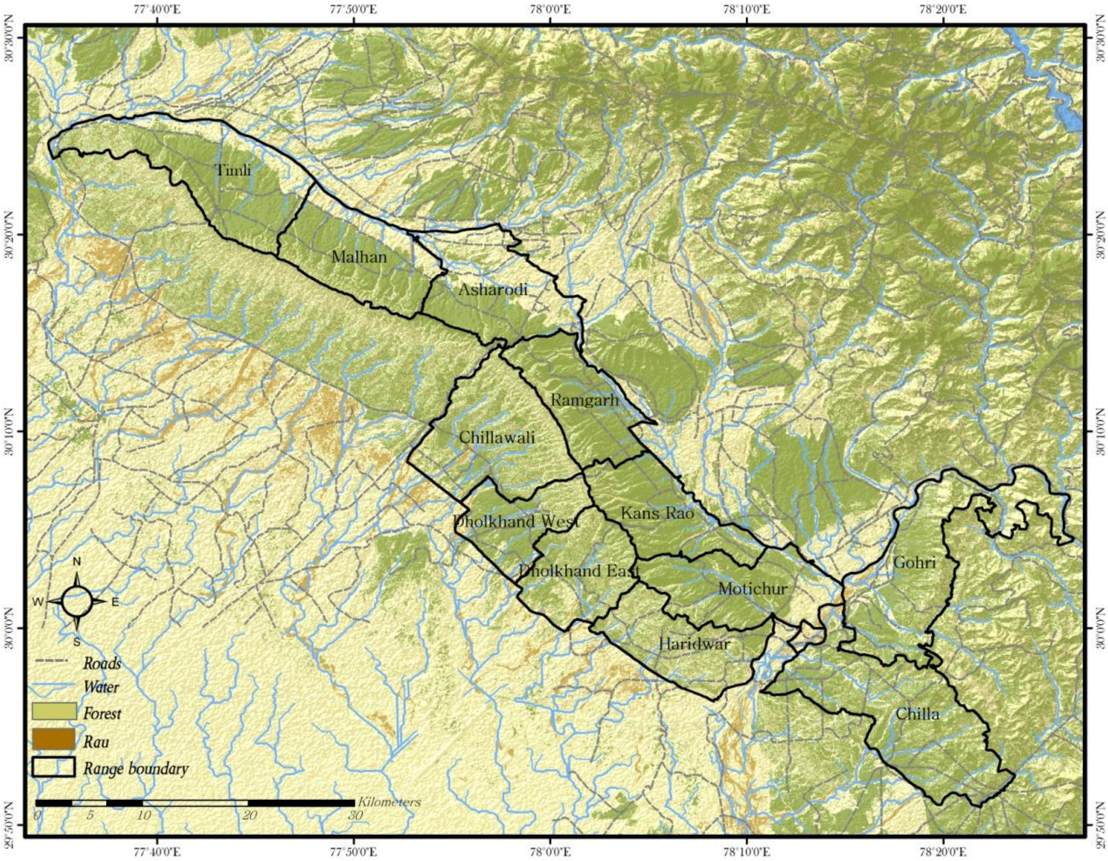
2.2b. Fauna

A number of globally threatened species are known from the area (Johnsingh et al. 2006, Harihar et al. 2009) which falls entirely within the Tiger Conservation Unit 1 (Wikramanayake et al. 2004). The area has rich avian diversity with 312 species reported (Pandey et al 1994), of interest are *Buceros bicornis*, *Anthracoceros albirostris* and *Chloropsis aurifrons* that are at the western most edge of their distribution; while *Gyps benghalensis*, *Gyps indicus*, *Gyps himalayensis*, *Gyps fulvus*, *Neophron percnopterus*, *Aegypius monachus* and *Sarcogyps calvus* are some of the threatened species (www.iucn.org access in 2014). The area is also listed as an Important Bird Area (Islam and Rahmani 2004). There are nearly 50 species of reptiles and amphibians (Vasudevan and Sondhi 2011) and nearly 125 species of icthyofauna are reported from the area (Sinha et al 1998).

2.3. Socio-Economics attributes

The area has experienced a varied history of landuse in the agriculture-forest frontier from the Mughal, colonial administration to present day urbanisation (Habib 1999, Flint 1998 and Tucker 1998). After the creation of State Uttarakhand in 2000 and Dehradun as the interim capital the area has reported a decadal increase of 33% since 2001, with Dehradun (16.78%) and Haridwar (19.04%) accounting for 35% of the states population of 1.01 cr with a density of 665.5 persons/km^2 that is much higher than the states density of 189 persons/km^2 (www.census2011.co.in). The presence of *Gujjars* a semi-pastoral community with high dependence on fodder for their livestock is a concern with efforts underway to resettle them outside the protected area. Another group called *taungyas* are settled on the south boundary of the Rajaji NP. The dependency of these ethnic groups and villages on the fringe for fuel wood is high.

Figure 2.1. Map of the study area.



CHAPTER 3

OCCUPANCY ACROSS SEASONS

3.1. Background

Establishing a protected area (PA) is considered as a cornerstone in biodiversity preservation (Naughton-Treves et al. 2005). But, despite the exponential increase of such institutional mechanism, along with additional supporting architecture (community based efforts and sensitisation regarding the values and essence of ecosystems) the decline in habitats or species have not abated (Boakes et al. 2010). In some cases, these declines have led species to reach their critical threshold levels (Brooks et al. 2006) and local extinctions without any apparent change in the habitat (Sodhi et al. 2004). This cognisance has been primarily attributed to the widespread dominance, colonisation and the synergistic effects of the human race (Rhymer and Simberloff 1996, Diamond 2002) where it has shaped the present patterns in landuse-landcover (Foster et al. 1998, Geist and Lambin 2002). Understanding how species have endured vital shifts (level of population), accompanied with correlated effects of potential drivers such as environment and habitat, predict the usage across space and time provide the basic foundation for managing and implementing strategies for conserving critical populations (Simberloff 1986).

Certainly, no species is equally abundant and evenly distributed across its distributional range. Suggesting there are ways wherein a species maximises the net energy intake for growth and reproduction while minimising its risk to predation. This spatio-temporal habitat-specific relationship (Luken 1990) is critical for the long term persistence of the species in any given area (Johnson 1980). Abundance or population size has traditionally been preferred, to study such relationships, but 'detection error' complicates the estimate as it is difficult to separate out the ecological and detection process in the analysis (MacKenzie et al. 2002). It is easier to incorporate 'detectability' a trait for the whole species (occupancy) than for the individual (abundance). Hence, occupancy has been studied as a surrogate for abundance (MacKenzie et al. 2006). By incorporating imperfect detections not only for the species but also related to habitat characteristics, observer and survey design, allows 'occupancy' to be used as a state variable for monitoring (Gurutzeta et al. 2010).

On large spatial scale terrain, structural vegetation and large infrequent disturbances are known to influence the observed patterns (Foster et al. 1998). Topographical heterogeneity influences the microclimate forming an important feature of the structural niche. It might influence positively the species presence or might preclude in utilising certain areas due to the higher associated energy costs or act as a thresholds between sympatric species (for eg. Sukumal and Savini 2009) and sometimes facilitating the use as escape cover (for eg. Kramer 1972). Initial studies on habitat occupancy emphasised the importance of ‘physiognomy’ namely vegetation structure and habitat configuration namely size, shape and distribution within an area (for eg. Hilden 1965, Wiens 1969). However, subsequent studies laid greater emphasis on plant species composition namely ‘floristics’ (for eg. Rotenberry 1985). The use of physiognomy or floristic are not independent of each other, but an issue of scale at which the patterns are analysed (Wiens 1989). At fine scale ‘floristics’ better predicts the bird-habitat relationship rather than physiognomy and conversely at coarser scale ‘physiognomy’, thus affirming to the hierarchical pattern of habitat selection (Lee and Rotenberry 2005). Lately, remote sensing technologies have been developed and are capable of delivering information in regards to structural attributes of forest strand and canopy cover (Innes and Koch 1998). Remotely sensed imageries, along with automated data processing techniques provide high quality time-series continuous data sometimes at no costs. This has considerable impetus in supplementing information for large geographic areas which are characteristics of the species habitat (Turner et al. 2003). The inter annual variability of Normalised Differential Vegetation Index (hereafter NDVI) is an index of habitat heterogeneity while, multi-temporal NDVI correlates with a number of vegetation related parameters (Pettorelli et al. 2005) and has been used to differentiate amongst forest vegetation types and to study effects of season on the forest vegetation (Lunetta et al. 2006). But habitat suitability can be altered due to large infrequent and intermediate disturbances making habitats less preferred.

Seasonal changes are known to have correlated effects on the resource levels and subsequent changes in the population (Ghosh et al. 2011). Species are known to have their life histories concomitant to these seasonal changes. Understanding how species respond to such changes of resources is necessary given the additional constraint of disturbances within the landscape. The ecological factors that influence the occurrence and abundance of the species at a landscape level have not been investigated. In the context of changing landscape and increase pressure on the existing protected area, there is a critical need to investigate species-habitat relationships and hence this chapter will deliberate on the pattern of occupancy in *G.gallus* in the western part of the Shiwalik landscape. The aim is to determine the occupancy and detection probabilities of *G.gallus* during non-breeding (2009) and breeding (2010) season and assess the influence of vegetation, topography and disturbance on *G.gallus*.

This study had the following objectives

1. To assess the patterns of occupancy of *G.gallus* and factors influencing it.
2. To determine the habitat covariates that influence occurrence and of *G.gallus* in the landscape

The study attempted to answer the following questions

a. What are the factors that affect occupancy and detection probability for *G.gallus*
b. Are these factors the same or different during the nonbreeding and breeding seasons?

3.2. Methods

3.2a. Study design

The study was conducted using a random systematic clustered based grid design. In order to achieve sampling independence, 2X2 km grid were realised within the study area and alternate grids were sampled. These grids were realised on the ground using a handheld Global Positioning System unit (Garmin 12XL). Each 2X2 km grid was further divided into 1X1 km grid and randomly three grids were sampled Fig 3.1. Within each 1X1 km grid a segmented 200 m, forest trail of 1km was selected (Fig 3.2). For both seasons, detection histories and covariate parameters were noted for each segment.

Occupancy studies involve a combination of 'sites' and 'surveys'. 'Sites' in this case would be each grid and 'surveys' will be each trail. The area will be covered within a short time interval (two/three months) during which occupancy status of the species is not expected to change. Data resulting from such a sampling scheme results in a history which is a sequences of 1's (detection) and 0's (non detection) (MacKenzie et al. 2006, Nichols et al. 2007) have been used for analysis.

Fig 3.1. Map of the study area showing the grids (1X1km) that were sampled.

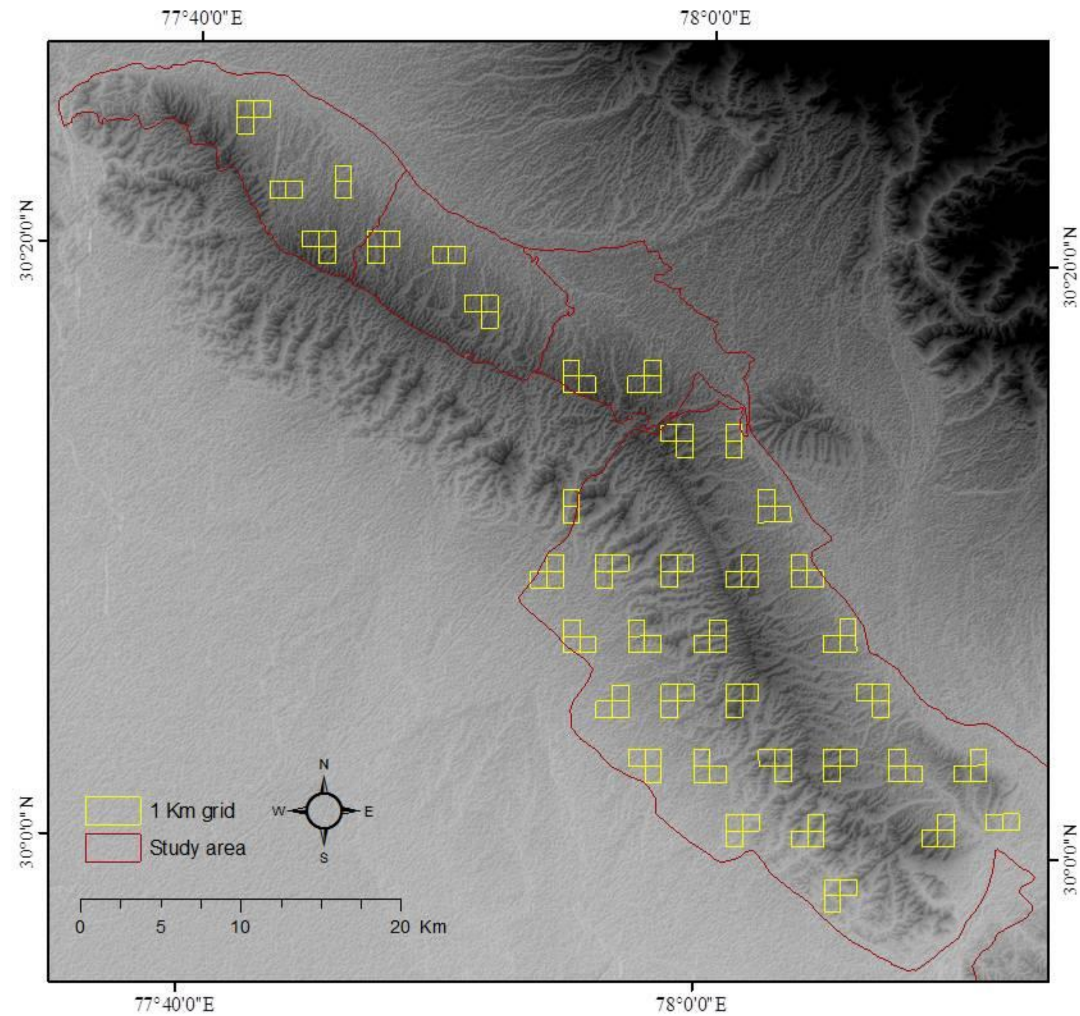
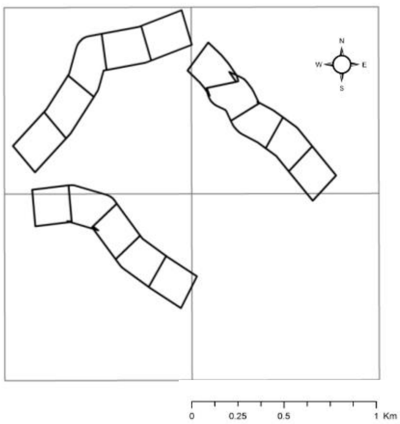


Fig 3.2. Graphical representation of the five, 200m segment in each 1km trail.



3.2b. Bird surveys

All surveys were conducted over two months within the non-breeding (November to January) and breeding (April to mid June) season, to minimise the likelihood of change in occupancy during sampling. These surveys total 105 kms, covering forest trails recording signs of disturbance and vegetative parameters. To ensure that the region is sampled under similar conditions the forest trails were sampled randomly during the most active time period of *G.gallus*, as reported from earlier studies in the region (Collias and Collias 1967; pers obs), omitting days of inclement weather in both seasons. For all surveys, aural and visual detections were used and data were recorded along segments as either detected (1) or non-detected (0).

3.2c. Identification of covariates

The relevant habitat covariates influencing the occurrence of the species were identified with the help of earlier work carried out within the landscape (Collias and Collias 1967, Kalsi 1992 and Harihar and Fernandes 2011) and from literature surveys (Javed and Rahmani 2000 and Subhani et al 2010). There were two form of measurable habitat covariates that were recorded namely ground based and remotely sensed covariate. Each segment of the trail was sampled for a ground based covariate, while mean values for the grid was used for the remotely sensed covariate. Variables were tested for multi-collinearity (Pearson's $|r| > 0.6$) and only one of the highly correlated variables was used for further analysis.

Ground based covariates: The available literature suggests that *G.gallus* is a forest generalist that prefers dense undergrowth and lower canopy cover hence parameters of undergrowth vegetation were recorded. The structural attributes were recorded within a 5m radii plot. The inventories include percentage of undergrowth cover, average shrub height, undergrowth denseness and resource availability. As vegetation attributes are known to change seasonally and this might have an influence on the species and resource availability hence the above said covariates were measured independently for each season.

The area experiences considerable anthropogenic pressures from the *dheras* (settlements of nomadic gujjars) that are situated within and villages located all along the periphery. Persistence of humans was calculated for each site by the summed value of sightings (settlements or traversing) and signs. The signs of human activities include tree lopping, presence of stumps, collection of fuel wood, clearance of the undergrowth vegetation, and other collection signs of non-timber forest produce (NTFP). During the dry season the landscape experiences large scale forest fires which, is also the breeding season for the species. Hence, this covariate was used for the breeding season only. The explanations for each of the habitat variables are given in Table 3.1.

Table 3.1 Habitat variables recorded as covariates for each 200m segment of the trail.

S.No	Variable Name	Description
1	Shrub height (S)	Average shrub height across four cardinal directions.
2	Undergrowth cover (C)	Average cover across four cardinal directions, measured at breast height with the help of a point frame spaced at 1cm.
3	Undergrowth denseness (U)	Average measure across four cardinal directions, measured with the help of a density board.
4	Resource (R)	Average number of insects and fruits available in the forest litter at four cardinal directions,
5	Disturbance (D)	Utilisation of sites for fuel-wood collection, collection of non-timber forest produce, livestock grazing and weed eradication. Forest fires during the breeding season are known to cause large scale disturbance. Hence for the breeding seasons locations burnt or burning were recorded as ‘1’ or not as ‘0’.

Remotely sensed covariates: For variables of topography and vegetation surrogates, remotely sensed data was used. Slope was calculated using ArcMap 9.3. Heterogeneity in the terrain was calculated as the coefficient of variance (cv) of slope. Grids with higher variance indicate an increase in the ruggedness of the area. The vegetation correlates of NDVI ie., mean NDVI (NDVIm), where NDVIm correspond to tree density and canopy cover irrespective of seasonal change is an indicator of the temporal variation in phenology. This measure is computed through the normalised difference in surface reflectance at visible red (VR) and near-infrared (NIR) wavelengths for each pixel. Both these variables are not correlated with each other.

Processing of remote sensed covariates

Topographical variable was derived from the Global Digital Elevation Model (GDEM) which was extracted from Advanced Spacehorne Thermal Emission and

Reflection Radiometer (ASTER) data (http://www.gdem.aster.ersdac.or.jp/). This was processed using the program System for Automated Geoscientific Analyses (SAGA). Optimisation for maximum terrain heterogeneity was carried out by using neighbourhood analysis and information was extracted for each sampled grid. Landsat data (LISS-III) was selected for processing the NDVI correlates for the non-breeding and breeding season, details methods are given as Appendix 1.

3.2d. Data Analysis

Spatial sub-samples as temporal replicates was used for estimation of occupancy within a season, to estimate seasonal changes in occupancy for the species. As detection histories were constructed for every 200 m segment of the 1 km trail, sub-sampling without replacement was undertaken where three segments from each grid was used as a 'replicate'(Kendall and White 2009). This was then imported as survey and site specific covariates into program PRESENCE 5.1 (Hines 2006).

The continuous variables were standardised such as NDVI, slope, shrub cover and shrub height using z-transformation and treated categorical variables such as protection status and disturbance as dummy variables with values of 0 or 1 (Donovan and Hines 2007). For the site covariates, mean values for topographic and habitat characteristic variable and for the sampling covariates the actual bird survey data was used. The two model parameters i.e., the probability that a segment is occupied by the species (ψ) and the detection probability (p) were estimated in the occupancy framework (MacKenzie et al., 2002) using PRESENCE version 5.1 (Hines 2006). Single-species single-season models were used to construct effect of occupancy and detection. The program PRESENCE uses the logit link and a maximum likelihood approach to linearise the relationships among independent covariates and the probability of occupancy (ψ) and the probability of detection (p) given the species is present. Site selection criteria are prominent during the breeding season, by keeping occupancy of the breeding season for each grid as a standard and checked for rate in change of occupancy between the breeding and non-breeding season.

3.2e. Modelling effects of covariates on detection probability and occupancy

In the analysis, a two-step approach to model parameters of interest. Modelling covariates on detection probability was the first step, where parameter was either assumed constant or allowed to vary with individual or additively combined covariates. Given that variation in abundance could influence detection rates (Royle and Nichols 2003), Next, effect of protection status, shrub height, ground cover and human sightings were incorporate to minimise unmodelled sources of heterogeneity in detection probability. By using the most complex model of *psi* (ψ) namely the global model as a constant model, detection probability (*p*) was modelled (MacKenzie et al 2006). Candidate models using the Akaike's Information Criterion (AIC) were compared following Burnham and Anderson (2002).

In the second step, season-wise influence of covariates on occupancy while incorporating covariates of detection probability that contained highest Akaike weight were modelled seperately. The logit link and a maximum likelihood approach were used. The amount of over-dispersion in the data was examined by assessing the global model fit using 1000 parametric bootstraps (MacKenzie et al 2006) and inferences were made using model averaged parameters (Burnham and Anderson 2002). Lastly, the model averaged estimates of cell specific occupancy by considering all plausible alternative occupancy models and variance were computed following Karanth et al 2011.

3.3. Results

A total of 105 sites covering 105 km of trails, were sampled during the non-breeding and breeding season. In the non-breeding season, RJF‘s were detected on 71 segments in 50 of the 105 surveyed grids, resulting in a naïve occupancy estimate of 0.48 while, the breeding season the species were detected on 73 segments in 46 of the 105 surveyed grids, resulting in a naïve occupancy estimate of 0.44. Resource availability was correlated (Pearson's $|r| > 0.9$) with undergrowth cover and hence undergrowth cover was selected for the analysis due to its ease of explanation. Slope and slope cv was correlated (Pearson's $|r| > 0.7$) hence slope was used for further analysis.

3.3a. Detection Probability

While constructing the effects of variables on detection probability for each season two of the variables namely protection status (*P*) and shrub height (*S*) had greater support (nonbreeding w = 0.42; breeding w =0.77) than compared to all plausible models constructed to assess the effect on detection probabilities. Therefore, protection status and shrub height as additive models were used in detection probability to examine habitat occupancy than compared to any other possible alternatives.

The mean segment level probability of detection *G.gallus* were higher during the breeding season 0.33(0.08) than compared to the non-breeding season 0.30(0.08) and protected sites had higher probability of detection than non-protected sites (Fig 3.3). For both seasons, shrub height (S) shows a strong influence on the detection probability indicating a strong preference by the species (Fig 3.4a,b)

Figure 3.3 Probability of detecting Red Junglefowl within protected (PA) and non-protected areas (NPA) as a function of seasonality.

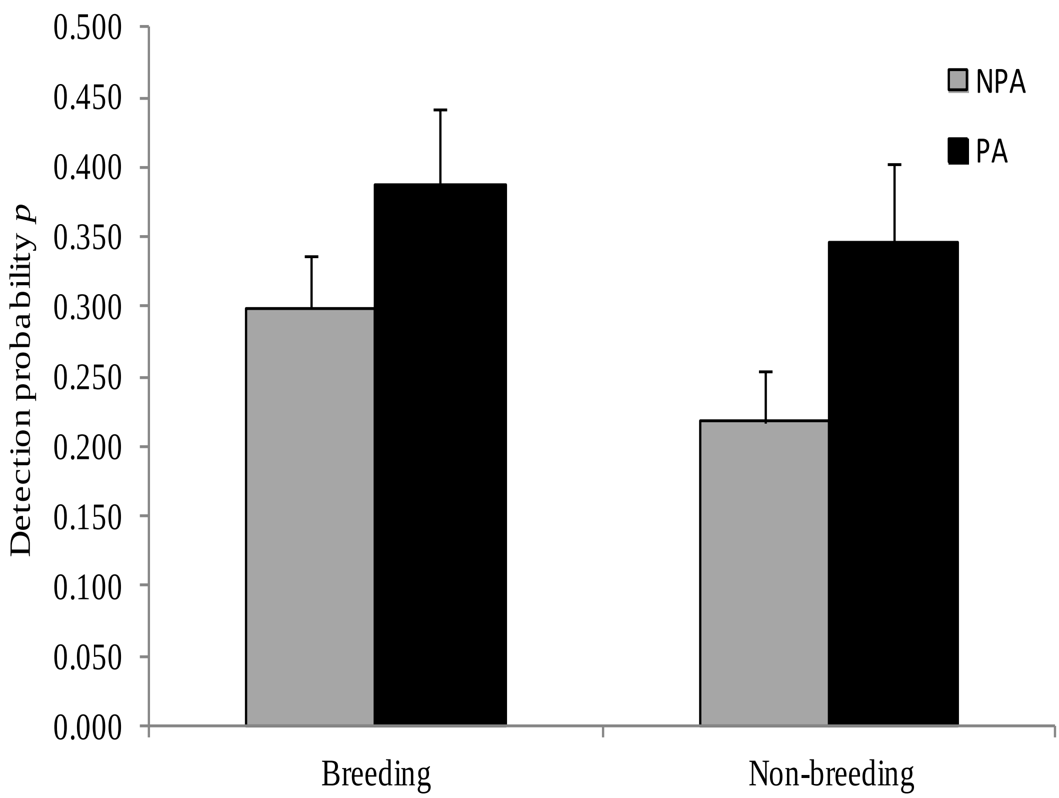
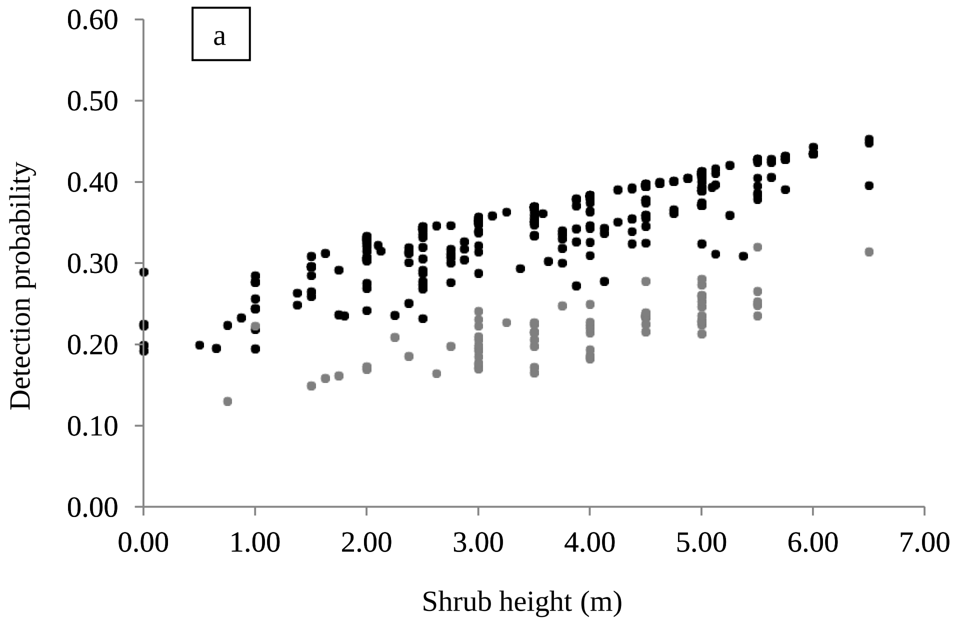


Figure 3.4 Probability of detecting Red Junglefowl during the nonbreeding (a) and breeding season (b) within protected ● and non-protected ● areas.



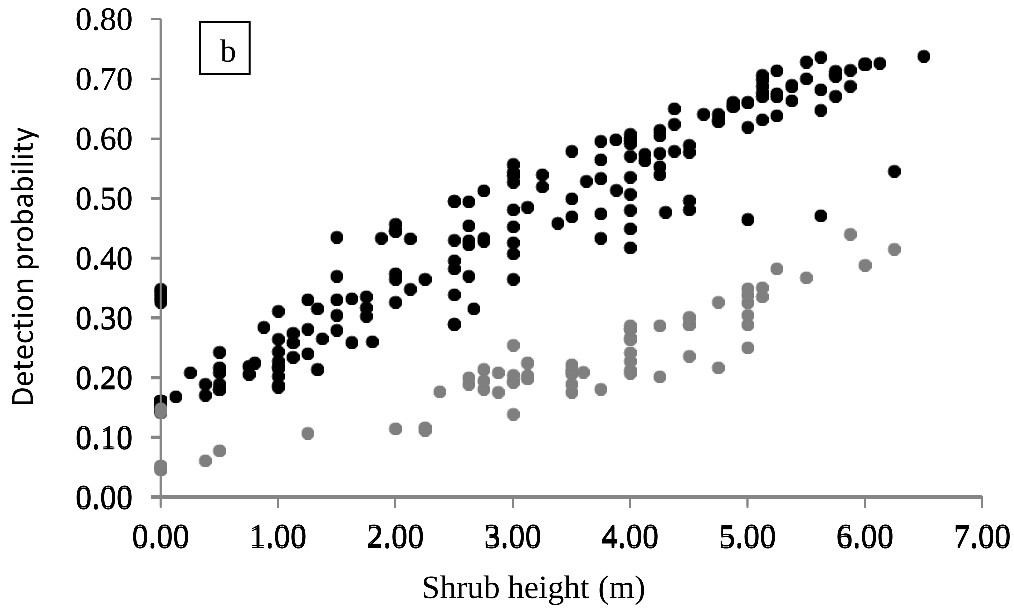


Table 3.2. Effect of covariates[1] on detection probability (*p*) of Red Junglefowl during non-breeding season

Model[2]	**ΔAIC**	**AIC_{wt}.**	**Parameter no**	**Deviance**
p (P+S)	0	0.185	8	308.82
p (S)	1.1	0.107	7	311.92
p (S +H)	1.5	0.087	8	310.32
p (P + S + H)	1.74	0.077	9	308.56
p (P + S +O)	1.82	0.074	9	308.64
p (P + O)	2	0.068	8	310.82
p (.)	2.14	0.063	6	314.96
p (P)	2.16	0.063	7	312.98
p (O)	2.31	0.058	7	313.13
p (S + O)	2.84	0.045	8	311.66
p (O + H)	2.89	0.044	8	311.71
p (S + O + H)	3.15	0.038	9	309.97
p (H)	3.38	0.034	7	314.2
p (P + S + O + H)	3.55	0.031	10	308.37
p (P + H)	4.08	0.024	8	312.9

[1] Covariates used to model detection probability (*p*) were protection status (P), shrub height (S), habitat openness (O) and human presence at the sight (H), '.' denotes that was held constant instead of being allowed to vary as a function of any covariate. [2] In all models probability of occupancy (ψ) was modelled on normalised difference vegetation index (N), terrain (T), undergrowth cover (C) and disturbance (D). '+' denotes covariates were modelled additively.

Table 3.3. Effect of covariates[1] on detection probability (*p*) of Red Junglefowl during breeding season.

Model[2]	**ΔAIC**	**AIC_{wt}**	**Parameters nos**	**Deviance**
p (*P* + *S*)	0	0.327	9	258.8
p (*P* + *S*+ *O*)	0.56	0.248	8	261.36
p (*P* + *O*)	1.6	0.147	8	262.4
p (*P* + *S* + *O* + *H*)	1.94	0.124	10	258.74
p (*P* + *S* + *H*)	2.52	0.092	9	261.32
p (*S* + *O* + *H*)	5.96	0.017	9	264.76
p (*S* + *H*)	6.58	0.012	8	267.38
p (*O* + *H*)	6.78	0.011	8	267.58
p (*O*)	7.47	0.008	7	270.27
p (*S* + *O*)	7.5	0.008	8	268.3
p (*S*)	7.73	0.007	7	270.53
p (*P*)	16.19	0	7	278.99
p (*P* + *H*)	18.19	0	8	278.99
p (.)	18.22	0	6	283.02
p (*H*)	19.02	0	7	281.82

[1] Covariates used to model detection probability (*p*) were protection status (P), shrub height (S), habitat openness (O) and human presence at the sight (H), '.' denotes that was held constant instead of being allowed to vary as a function of any covariate. [2] In all models probability of occupancy (ψ) was modelled on normalised difference vegetation index (N), terrain (T), undergrowth cover (C) and disturbance (D). '+' denotes covariates were modelled additively.

3.3b. Occupancy Patterns

Non-breeding season

For the non-breeding season, terrain (T) was selected as the final best fit model with AIC = 319.59 (Table 3.4). The occupancy estimates for the species during the non-breeding season revealed that terrain (T) accounted for the highest AIC weight of 0.96 added from all the model structures (Table 3.4). Therefore, terrain is the most important determinant of occupancy for the species during the non-breeding season.

Table 3.4. Effect of covariate(s)[1] on occupancy (ψ) during the non-breeding season for Red Junglefowl. Number of sites = 105.

Model[2]	**Δ AIC**	**AIC_{wt}**	**Parameter no**	**Deviance**
ψ (T)	0.00	0.42	5	309.59
ψ (N+T)	1.43	0.21	6	309.02
ψ (C+T)	2.00	0.16	6	309.59
ψ (N+C+T)	3.43	0.08	7	309.02
ψ(C+T+D)	3.47	0.07	7	309.06
ψ (N+C+T+D)	5.23	0.03	8	308.82
ψ(.)	8.04	0.01	4	319.63
ψ (C+D)	8.26	0.01	6	315.85
ψ (D)	9.16	0.00	5	318.75
ψ (C)	9.96	0.00	5	319.55
ψ (N)	10.02	0.00	5	319.61
ψ (N+C+D)	10.24	0.00	7	315.83
ψ(N+D)	11.10	0.00	6	318.69
ψ (N+C)	11.92	0.00	6	319.51
ψ(.), p(.)	14.66	0.00	2	330.25

Note: Model selection based on Akaike's Information Criterion (AIC).

[1]Covariates used to model detection probability were NDVI (N), Shrub cover (C), Terrain (T) and Disturbance (D). '.' Denotes that was held constant instead of being allowed to vary as a function of any covariate.

[2]In all models the probability of detection (p) was modelled as an additive of protection status (P) and shrub height (S) ie., '$P+S$' based on model selection result presented in Table 3.2.

I also examined the β-coefficient values for different covariates which were expected to influence the occupancy for the species (Table 3.5). There was a negative influence of terrain on the occupancy levels of the species while undergrowth cover had a positive influence on the species occupancy levels. Though the trends for NDVIm and disturbance suggest that the species prefer thicker vegetation with a close canopy and lesser disturbance the occupancy levels does not give a clear pattern in this regard (Fig 3.5A). The site occupancy for *G.gallus* during the nonbreeding season is shown in Figure 3.6.

Table 3.5. Model-specific β-coefficient estimates and associated standard errors for covariates determining the occupancy of Red Junglefowl during the non-breeding season. Number of sites = 105.

Model	**βo ($SE_{\beta o}$)**	**β_N ($SE_{\beta N}$)**	**β_C ($SE_{\beta C}$)**	**β_T ($SE_{\beta T}$)**	**β_D ($SE_{\beta D}$)**
ψ (T)	1.19 (0.63)			-1.21 (0.48)	
ψ (N+T)	1.22 (0.65)	-0.31(0.42)		-1.34 (0.56)	
ψ (C+T)	1.19 (0.69)		0.01 (0.51)	-1.21 (0.50)	
ψ (N+C+T)	1.22 (0.71)	-0.31(0.43)	-0.01 (0.49)	-1.34 (0.58)	
ψ(C+T+D)	1.16 (0.71)		-0.05 (0.54)	-1.24 (0.54)	-0.42 (0.54)
ψ(N+C+T+D)	1.16 (0.70)	-0.23(0.46)	-0.05 (0.52)	-1.32 (0.57)	-0.29 (0.60)
ψ(.)	1.14 (0.59)				
ψ (C+D)	5.55 (4.18)		2.32 (1.62)		-2.54 (1.80)
ψ (D)	1.15 (0.73)				-0.54 (0.58)
ψ (C)	1.00 (0.60)		0.15 (0.52)		
ψ (N)	1.14 (0.60)	-0.04 (0.38)			
ψ (N+C+D)	5.24 (4.08)	0.16 (1.12)	2.24 (1.56)		-2.47 (1.64)
ψ(N+D)	1.17 (0.77)	0.11 (0.47)			-0.61 (0.68)
ψ (N+C)	1.09 (0.60)	-0.09 (0.42)	0.19 (0.57)		

Covariates used to model detection probability were NDVI (N), Shrub cover (C), Terrain (T) and Disturbance (D). '.' Denotes that was held constant.

Breeding season

For the breeding season undergrowth cover (C) and disturbance (D) were selected as the final best fit model with AIC = 275.8 (Table 3.6). The occupancy estimates for the species during the breeding season revealed that undergrowth cover (C) accounted for the highest AIC weight of 1.00 and disturbance (D) accounted for an AIC weight if 0.56 added from the model structures (Table 3.6). Thus undergrowth cover is the most important determinant for the species during the breeding season. The β-coefficient values for different covariates which were expected to influence the occupancy for the species during the breeding season are given in Table 3.7. There is a strong influence of shrub cover on the occupancy of the species (Fig 3.5B). The site occupancy for *G.gallus* during the breeding season is shown in Figure 3.7.

Seasonal change in occupancy

While comparing the occupancy rates between the non-breeding and breeding season there were no changes in 47% of the grids, positive change in 16% and a negative change in 37% of the grids (Fig 3.8).

3.4 Discussion

Although the current global conservation status for *G.gallus* is 'Least Concern' (IUCN), making an assessment of their status requires reliable information on population size and distributions. To help ensure a viable population, management guidelines should provide for the full range of habitat conditions and quality needed to maintain a well distributed population. *G.gallus* has their life histories concomitant to seasons and the availability of resources (Johnsgard 1986) thereby, making it critical to understand factors that govern their abundance. In this study, after the incorporation of imperfect detection into occupancy estimates, the proportion of area occupied by *G.gallus* in western Shiwalik substantially increased for the nonbreeding (ψ = 0.72± 0.13, 50%) and breeding (ψ = 0.58 ± 0.12, 31%) season respectively. Studies on the breeding behaviour of *G.gallus* have indicated that during the breeding season male *G.gallus* become territorial and there is an enhanced presence of the species at display arenas, this behavioural trait may have an influence on the detection rates for the species when compared to the non-breeding season when the species is more sedentary (Collias and Collias 1967, Johnsgard 1986). Also an earlier study

(Harihar and Fernandes 2011), conducted within the region had a higher detectability during the breeding season (summer) when vegetation is denuded compared to the non-breeding season when the undergrowth is luxuriant.

Table 3.6 Effect of covariate(s)[1] on occupancy (ψ) during the breeding season for Red Junglefowl. Number of sites = 105

Model[2]	**Δ AIC**	**AIC_{wt}**	**Parameter nos**	**Deviance**
ψ (C+D)	0.00	0.30	6	263.80
ψ (C)	0.65	0.22	5	266.45
ψ (N+C+D)	1.97	0.11	7	263.77
ψ(C+T+D)	1.97	0.11	7	263.77
ψ (C+T)	2.40	0.09	6	266.20
ψ (N+C)	2.41	0.09	6	266.21
ψ (N+C+T+D)	3.96	0.04	8	263.76
ψ (N+C+T)	4.32	0.04	7	266.12
ψ (T)	10.57	0.00	5	276.37
ψ(.)	11.10	0.00	4	278.90
ψ (D)	11.91	0.00	5	277.71
ψ (N+T)	12.55	0.00	6	276.35
ψ (N)	12.59	0.00	5	278.39
ψ(N+D)	13.33	0.00	6	277.13
ψ(.), p(.)	49.60	0.00	2	321.40

Note: Model selection based on Akaike's Information Criterion (AIC).

[1]Covariates used to model detection probability were NDVI (N), Shrub cover (C), Terrain (T) and Disturbance (D). '.' Denotes that was held constant instead of being allowed to vary as a function of any covariate.

[2]In all models the probability of detection (p) was modelled as an additive of protection status (P) and shrub height (S) ie., '$P+S$' based on model selection result presented in Table 3.3.

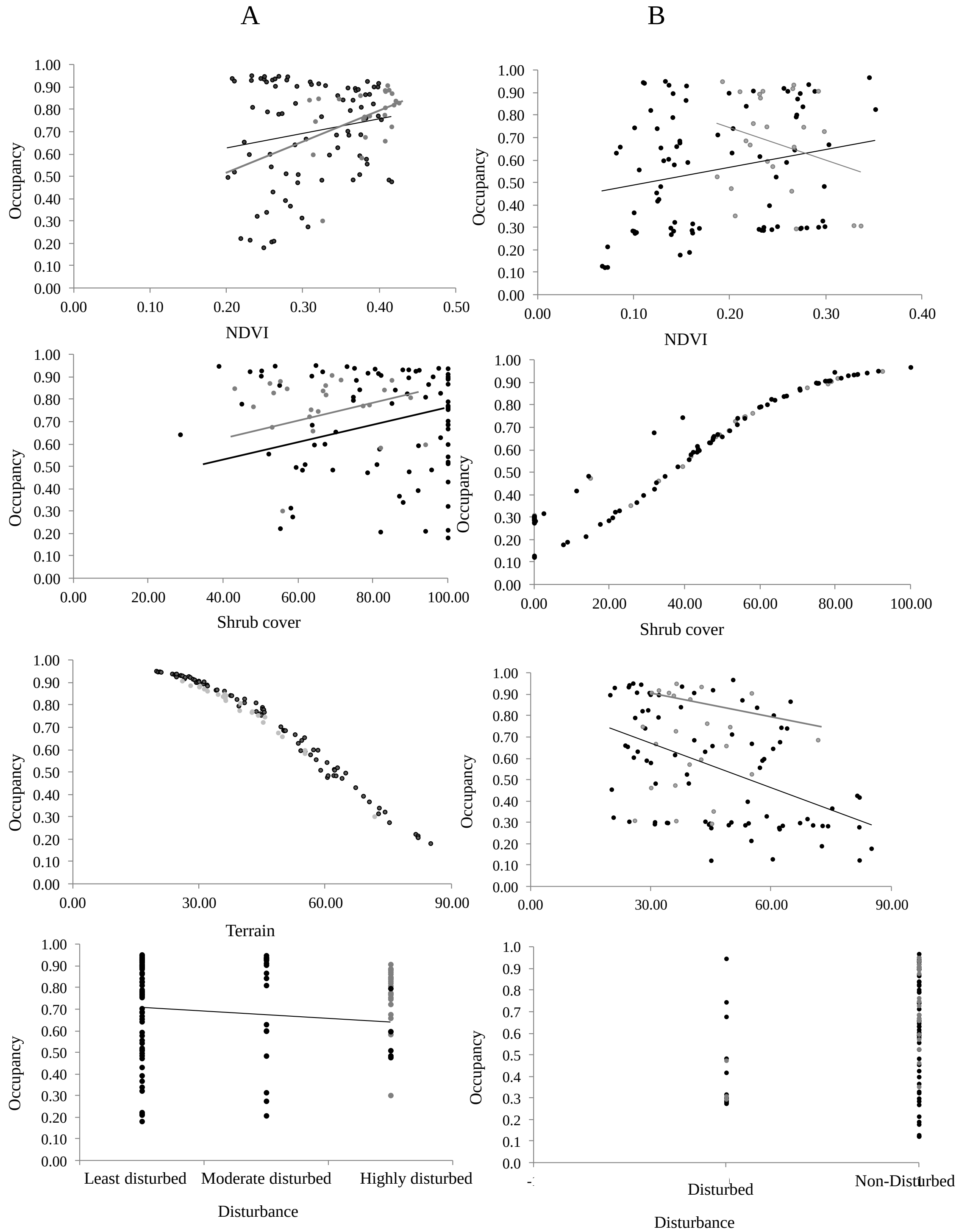
Table 3.7. Model-specific β-coefficient estimates and associated standard errors for covariates determining the occupancy of RJF during the breeding season. Number of sites = 105

Model	**β_o ($SE_{\beta o}$)**	**β_N ($SE_{\beta N}$)**	**β_C ($SE_{\beta C}$)**	**β_T ($SE_{\beta T}$)**	**β_D ($SE_{\beta D}$)**
ψ (C+D)	0.60 (0.43)		0.59 (0.23)		-0.59 (0.50)
ψ (C)	0.09 (0.21)		0.28 (0.11)		
ψ (C+T+D)	0.22 (0.16)		0.22 (0.08)	-0.01 (0.04)	-0.21 (0.18)
ψ (N+C+D)	0.22 (0.16)	0.01(0.04)	0.22 (0.09)		-0.21 (0.18)
ψ (C+T)	0.04 (0.09)		0.11 (0.05)	-0.01 (0.03)	
ψ (N+C)	0.04 (0.09)	0.02 (0.04)	0.11 (0.05)		
ψ (N+C+T+D)	0.08 (0.06)	0.00 (0.02)	0.08 (0.03)	0.00 (0.02)	-0.08 (0.07)
ψ (N+C+T)	0.02 (0.03)	0.00 (0.02)	0.04 (0.02)	0.00 (0.02)	
ψ (T)	0.00			0.00	
ψ (.)	0.00				
ψ (D)	0.00				0.00
ψ (N)	0.00				
ψ (N+T)	0.00			0.00	
ψ (N+D)	0.00				0.00

Covariates used to model detection probability were NDVI (N), Shrub cover (C), Terrain (T) and Disturbance (D). '.' Denotes that was held constant.

Implementation of institutionalised mechanisms such as protected area has shown to have a direct influence on the abundance of the species and thereby detectability especially when compared to non-protected sites. Such increases in abundance have been reported for other galliformes in the Himalayas (Miller, 2010). Like most galliformes, *G.gallus* are ground dwelling and hence structural components (height) of the undergrowth vegetation (shrub) will have a direct influence on the species preference such preferences for height of shrub cover were reported from Sage grouse (Gregg et al., 1994).

Figure 3.5.. Relationship between occupancy probability and covariates across protected ● and non-protected ● sites for the non-breeding (A) and breeding (B) season*.



*Note: Graphs on the left are for the non-breeding season while on the right are for the breeding season

Figure 3.6. Site occupancy of Red Junglefowl during the non-breeding season

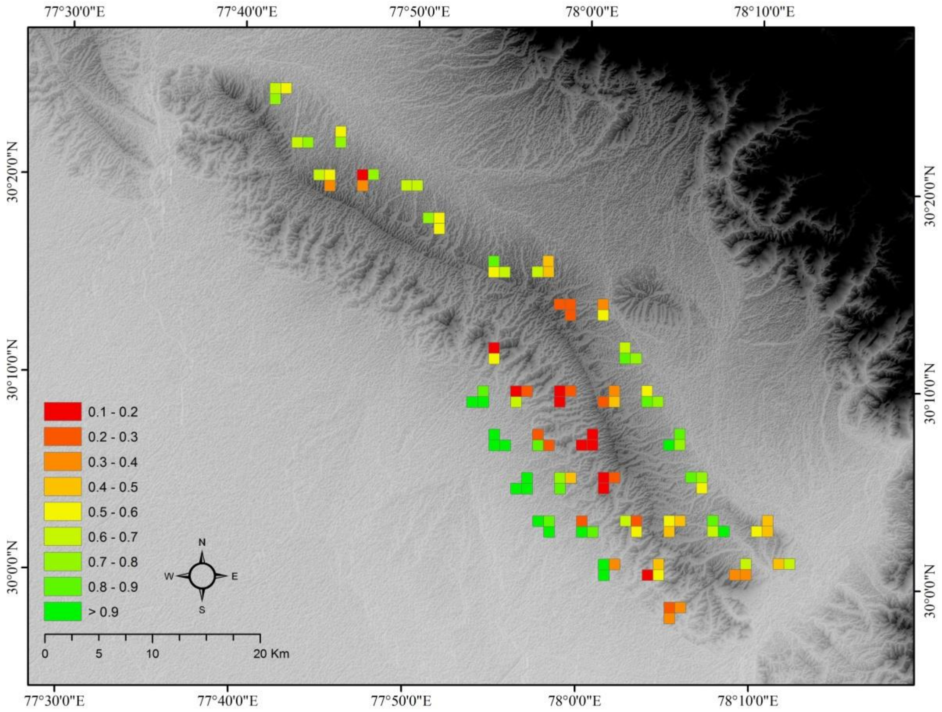


Figure 3.7. Site occupancy of Red Junglefowl during the breeding season.

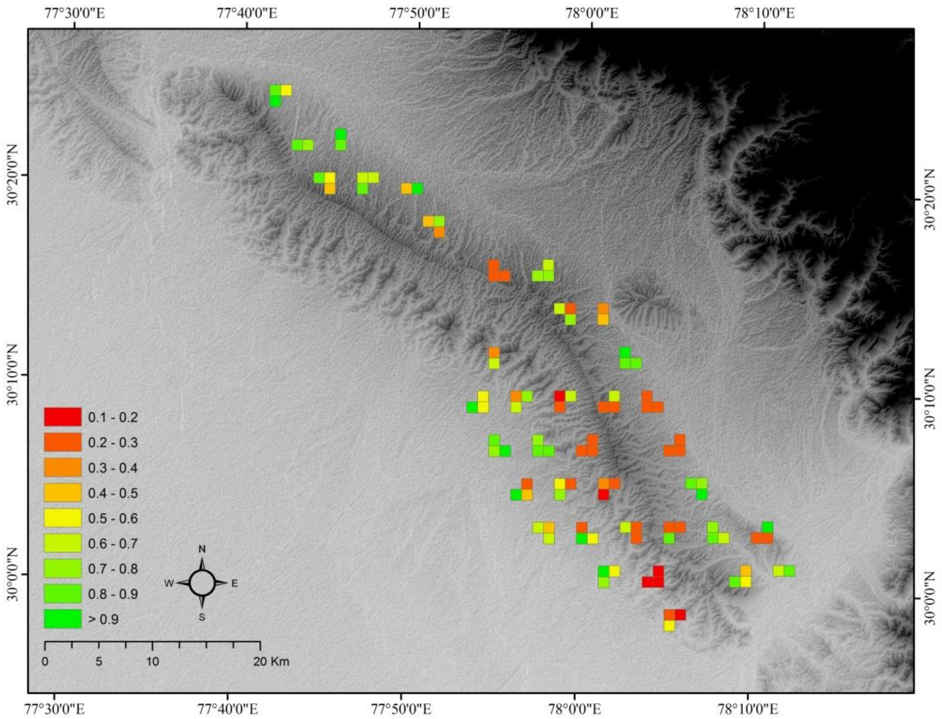
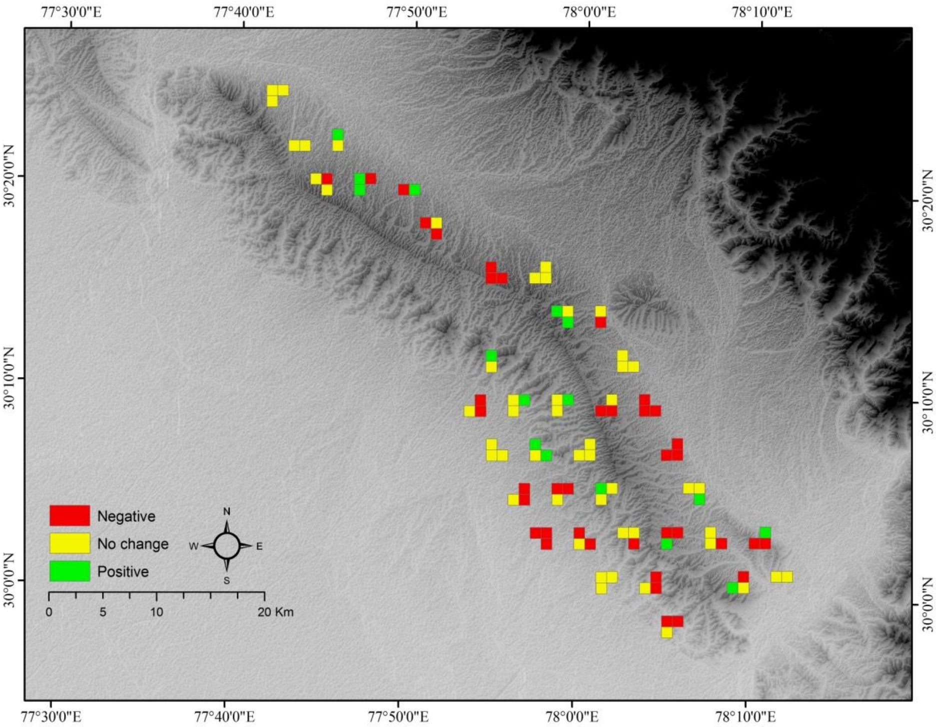


Figure 3.8. Change in site occupancy between nonbreeding and breeding seasons for Red Junglefowl within the study area



There were different variables that influenced *G.gallus* occupancy for the non-breeding and breeding season suggesting a seasonal response of the species. For the non-breeding season topography (slope) showed a strong negative response to *G.gallus* occupancy as indicated by the β-coefficient in Table 3.5 (Fig 3.4c) while for the breeding season though the effect were negative there was a weak relation to the slope. Though, pheasants are distributed across various elevation gradients in the western Himalayas, slope has never been reported to form a threshold for the species. Similar effects of slope forming a threshold between sympatric species of *Lophura* are reported for Siamese fireback *Lophura diardi* and Silver pheasant *Lophura nycthemera* (Sukumal and Savini, 2009).

G.gallus are ground dwelling and nesting species and hence undergrowth vegetation provides cover and resource to the species especially during the nesting and when the chicks are young. For the breeding season, there was a strong positive relation as indicated by the β-coefficient in Table 3.7 (Fig 3.5b). The relationship was not as strong as that for the non-breeding season (Table 3.5) but trends do suggest the preference for dense undergrowth. The non-breeding season sampling was undertaken post monsoon where vegetation growth is enhanced and resource is not a constraint hence the species could utilise sub-optimal sites.

There was no influence of NDVIm on the occupancy patterns of the species, during the non-breeding and breeding season. Though, *G.gallus* are forest surrogate species there are found in varied habitats (Ali and Ripley 1983) and earlier studies within the region suggested that they prefer mixed forest types with slight openness in the canopy or forested areas adjacent to agricultural fields (Collias and Collias 1967 and Kalsi 1992).

Though the β-coefficients for disturbance for non-breeding and breeding season were negative there were no particular trends in the occupancy for *G.gallus*. The species is known to be recorded near human habitations and agricultural fields (Hume and Marshall 1879). There are studies which correlate human activities to be the cause of decline in many galliformes, this is the first instance where fire, was used as one of the major disturbance factor. There are annual fires of varying intensity and frequency that affect large parts of the western Shiwalik forest. These fires occur are mostly during the summer season which is also the breeding season for *G.gallus*. In order to mitigate the scale of destruction, preventive measures are undertaken prior to the summer month. This was one of the reasons why there were negative changes in the occupancy of *G.gallus*. Measures such as creation and maintenance of fire-lines are needed as they are known to prevent large scale annihilation of the vegetation but the timing when these management activities are undertaken generally coincide with the breeding season of *G.gallus*, hence commencement of such management activities prior to the breeding season may lessen the disturbance to the nesting birds. Also, due to the spread of *Lantana camara* an invasive species, weed eradication were carried out in many parts of the western block of Rajaji NP. The removal of this undergrowth opens up the ground layer thus reducing the suitability of the site for *G.gallus*.

Decision making in conservation and applying management practices is frequently done, but mostly targeted at key charismatic species. Although, there are excellent models and practices undertaken in restoring fragmented habitats with native tree species, such management practices are completely lacking when restoring understorey after eradicating invasive species. Though, invasive species eradication is practiced annually as a measure to improve forage quality for the ungulates.

CHAPTER 4

ABUNDANCE AND FACTORS INFLUENCING ABUNDANCE

4.1. Background

Ecological processes are known to influence the distribution of a species, but these processes operate on various spatial scales (Wiens 1976, 1989). How a species perceives, reacts and responds to these ecological processes formulates a relationship with its environment (Cox and Moore 1993). This species-habitat relationship has been explained as a surrogate of abundance for the species. Traditionally, the most important processes assumed to be affecting this abundance-habitat association was at the fine (local) spatial scale (for eg. MacArthur and MacArthur 1961), but recent studies have elicited the relationship to differ in magnitude and direction, as scale changes (for eg. Wiens et al. 1987). There are several plausible ecological process that can generate the patterns in the frequency of local abundance across a particular area of interest or species range that are closely akin to those that are actually observed suggests that different processes may give rise to the observed patterns in different cases. For instance, mostly common species typify left skewed distribution of log-transformed abundance (Gregory and Blackburn, 1998). It would rather be interesting to note where different levels of abundance occur within the range. The most common way in addressing this issue has been the partial response of asking without any necessary explicit reference to the spatial relations of sample points, how the abundance of species responds to the environmental gradients (Fahrig 1992).

It is known that bird-habitat associations can be modelled into different scales wherein the variable may not be independent of the scale for instance the use of climatic models to understand geographic distribution of the bird which is done at coarse scale (Bibby et al 1992). At a slightly fine level of information with the use of patch metrics (McGarigal and McComb 1995), and at a micro scales where traditionally habitat factors are measured (for eg. the nesting tree) by traditional measures of habitat within plots (James 1971). Furthermore, although we know that there are correlations among these levels, we also frequently find independent contributions of each level to overall patterns of occurrence (Knick and Rotenberry 1995). When these correlations exist within a multi-scale habitat study, the relationships identified at different scales are not likely to be independent. Hence, the interest is to determine how multiple scales of variation combine to produce the observed pattern and not determine the correct scale at which the species operate.

Understanding the relative importance of different factors in producing the abundance pattern is essential for protecting the habitat in order to maintain the populations at the landscape level. It is in this regard that the chapter will delve on abundance of the *G.gallus* in Shiwalik landscape for the non-breeding and breeding season and during the breeding season which are the local and landscape scale variable that influence abundance.

4.2Methodology

4.2.1. Study design

For the abundance estimates of *G.gallus*, the study was conducted using a random systematic clustered based grid design. In order to achieve sampling independence, 2X2 km grid were realised within the study area and alternate grids were sampled. These grids were realised on the ground using a handheld Global Positioning System unit (Garmin 12XL). Each 2X2 km grid was further divided into 1X1 km grid and randomly three grids were sampled Fig 3.1. Within each 1X1 km grid a forest trail of 1km was selected which was sampled at 200m interval. Abundance estimates were derived using the larger landscape.

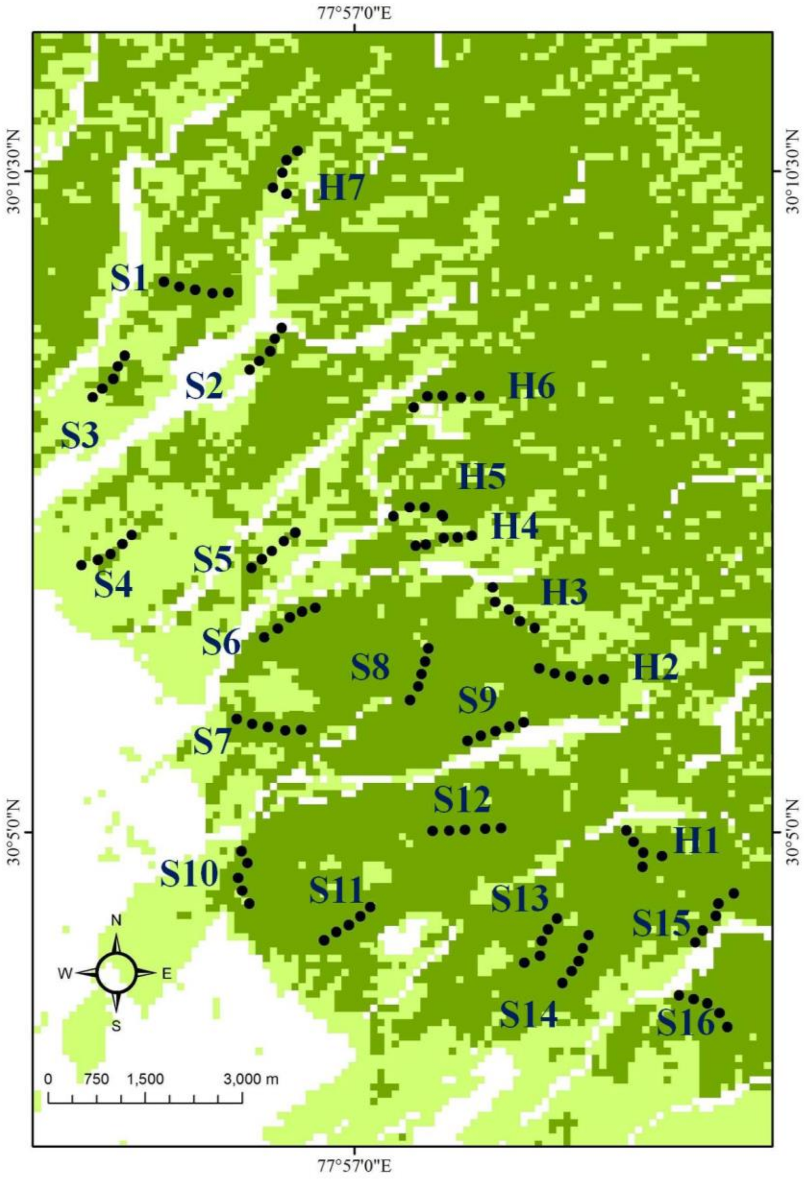
In order to study multiscale-factors influencing abundance an intensive study with a good representation of forest types, elevation and range of human use was selected in the south western region of Rajaji NP. This area was selected to address the species-habitat relationship at two scales *viz;* local and landscape. For the local scale seven trails (H) were laid in the hilly tracks while 16 transects were laid in the plain area (S) and each trail/transect was sampled at 200m interval (Figure 4.1). Each trail was traversed a minimum of five times to record detections of *G.gallus*. Encounter rates were used as the dependant variable, while using local and landscape scale factors as predictors.

4.2.2. Bird surveys

Initially, rapid assessment surveys involving aural and visual detections were conducted along the entire study area. The methods is described in 3.2b.

Following this sampling was undertaken in the intensive area where trails of 1 km were laid covering the various gradients of elevation, shrub height and human use. Bird surveys were done during the breeding season (April to June). All sampling was carried out at dawn an hour prior to sunrise only on day of sunshine. For the analysis the total birds encountered for the trail is used as a surrogate for abundance.

Figure 4.1. Map of intensive study site showing local scale sampling points● at every 200 m of the trail where H=hilly and S = plain.



4.2.3. Variables for multiscale

The variables were measured at two spatial scales *viz*; 200m and 1000m considered to be biological meaningful for *G.gallus*. these were labelled as local and landscape scale respectively to correspond to the areas and factors sampled at each scale. During the breeding season, *G. gallus* is known to be territorial and hence 200m may be considered as the local scale, while the home range for *G.gallus* is not known from these forested areas, I assume that it would not be more than 1000m during the breeding season.

Sampling of vegetation was undertaken at every alternate 200m on the trial where habitat covariates were measured. In order to measure habitat covariates a 10m radii circular plot was laid to enumerate tree height (TH) and canopy cover (cc), nested within the 10m radii a 5m circular plot was used to measure shrub height (SH), shrub cover and undergrowth denseness. Shrub cover (0.1-1 scale, where 0.1=no cover and 1= full cover) and undergrowth denseness (0.1-1 scale, measured using a density board) was multiplied to give the undergrowth vegetation index measure undergrowth vegetation parameters and 1m for resources availability was measured as an index (0 = low, 1 = high).

From remote sensed data NDVI and terrain parameters were generated for 1000m, using the 200m point as a centre. Moderate Resolution Imaging Spectro radiometer (MODIS) dataset gives a 16 days composite index of NDVI. This index correlates to vegetation heterogeneity and the average seasonal NDVI (N) is an indicator of the temporal variation in phenology. The Global Digital Elevation Model (GDEM) was extracted from Advanced Spacehorne Thermal Emission and Reflection Radiometer (ASTER) data and topographic wetness index and terrain ruggedness (henceforth TR) was created. TR is a physiographical measure of each mountain in a 3-dimensional dispersion of vectors normal to the planar facets on a landscape. TR is a dimensionless unit ranging from 0 (flat) to 1(most rugged). In order to capture the heterogeneity of the landscape the neighbourhood analysis was carried out. TR has been built in GIS (Sappington 2007) using Hobson's techniques (Hobson 1972) that incorporates heterogeneity of both slope and aspect. TR gives a better measurement of the terrain heterogeneity and performs better than other available terrain topologies. Topographic wetness index (W) is an index to generate subsurface water. The wetness index is a dimensionless unit ranging from -1 (dry) to 1(wet). This measure can be directly correlated for presence of water in a landscape.

4.2.4 Analysis

4.2.4a. Abundance estimates:

To assess the abundance patterns of *G.gallus* populations, field sampling was carried out during the non-breeding (November 2009 to January 2010) and breeding (April to June 2010) season respectively. During each period, trails with lengths varying from 0.95 to 1.2 km, laid in different parts of the study area covering all vegetation associations were surveyed once each (105km of survey/season). On survey, walks were carried out by two observers during 0615 hrs and 0915 hrs (non-breeding) and between 0530 hrs and 0830 hrs (breeding). When *G.gallus* was encountered parameters such as group size, detection (sighting or aural) angle and distance measured by a laser range finder or an approximation to the nearest 5m. We estimated the population density using program DISTANCE 5.0 Release 2 (Thomas et al., 2006).

To model detection functions, the data was examined for signs of evasive movement and peaking at great distance from the line transect for *G.gallus*. Following this, the data were either truncated at great distances or re-classed to ensure a reliable fit of key functions and adjustment terms to the data. Akaike Information Criterion (AIC) and goodness-of-fit (GOF-p) tests were used to judge the fit of the model. Using the selected model, estimates of half strip width (μ) expressed in metres, encounter rate (ER) expressed as the number of groups/km, mean group size (GS) and individual density (Di) expressed as number of individuals/km^2 were derived.

To test for difference across seasons on the estimates of ER, GS and D Mann-Whitney U test was carried out. As group size distributions are rarely normal (Reiczigel et al. 2008), hence median group size (GSm) was computed and test for differences in grouping across seasons using the Mood's median test (FLOCKER 1.1; Reiczigel and Rozsa 2007) was carried out. For the purpose of the analysis we used the spatial replicates per season as our basic sampling unit.

4.2.4b. Factors influencing abundance

The variables measured at different scales were used independently to access the influence on abundance. In order to access the influence of a factor on the dependent variable a series of generalized linear models (GLM) with gaussian errors and logit link was used. The technique involved fitting models to produce estimates of the variation explained by different sets of variables (Borcard et al. 1992). A forward stepwise procedure for each levels was preferred starting from a full model that includes all predictors. This was done in order to exclude within each group variables that did not contribute significantly ($P > 0.05$) to the explained deviance. Thus, final best models included only significant predictable variables using the Akaike's information criterion (AIC) differences in all steps of the models (Burnham and Anderson, 2002). These models were run in R with glmulti package (Calgano and Mazancourt 2010) and the likelihood ratio test was used to compare full and reduced models using an ANOVA.

4.3 Results:

Based on a total effort of ~208.4 km of trail walks across the two seasons, a total of 260 individuals during the breeding season and 251 individuals during the non-breeding season were encountered. There were 22 sightings (each season) of *G.gallus* accounting for 44 individuals for the non-protected areas against 95 sightings (207 individuals) during the non-breeding and 96 sightings (216 individuals) during the breeding season. There were significant differences in the mean encounter rates (number/kilometre) for the protected (2.66 birds/km) compared non-protected areas (1.9 birds/km).

In the intensive study area, from a total effort of 129 km there were a total of 64 segment detections out of 115, with 283 individuals being detected. For the trails in the plain area, out of 80 segments there were detections in 44 segments where 207 individuals were encountered, while trail in the hills, 20 out of 35 segments had detections, where 76 individuals were detected.

4.3a. Abundance estimates of *G.gallus*:

The results of the Mann-Whitney U test indicate that there were no significant difference in the ER ($Z= 0.0649$, $P = 0.95$), Gs ($Z = 0.129$, $P= 0.44$) and D*i* ($Z= 0.136$, $P = 0.76$). The median group size (G*sm*) was 2 (CI 95% = 1-2 breeding, 1-3 non-breeding) and Mood's median test for differences in grouping indicated that no such differences across the season for *G.gallus* ($\chi^2 = 1.64$, P=0.121). The estimated half-strip width (μ) for the non-breeding season (17.8 ± 5.4) was smaller than compared to the breeding season (19.5 ± 3.2) (Table 4.1).

Table 4.1. Densities (#/km^2) of Red Junglefowl during non-breeding and breeding seasons in western Shiwalik Landscape November 2009 to June 2010.

Season	Non-breeding	Breeding
ER	0.35 ± 0.05	0.41 ± 0.04
Dg ± SE	2.56 ± 0.96	3.08 ± 0.52
GS ± SE	2.96 ± 0.81	2.04 ± 0.11
Di ± SE	5.64 ± 4.88	6.54 ± 4.33
CV%	34.12	28.64
GOF-p	0.744	0.824

Note= Estimated encounter rates (ER), density of groups (Dg; no.groups/km^2), group size (Gs), density of individuals (Di; no. individuals/km^2), CV% = percent coefficient variation on D*i*, the probability of Chi-square goodness of fit (GOF-p) with associated standard error (SE). Total effort of 208.4 km.

4.3b. Factors influencing abundance

Issues of multi-collinearity were address by eliminating correlating variables where Pearson's $|r| > 0.5$ (Appendix 4.1) one variable from the group of highly correlated variables was selected for each of the scale types. The final selected variables for the local and landscape scale are shown in Table 4.2, the global model was run with all variables.

Global model: The global model has all variables from the local and landscape scale. The coefficients estimated suggest that an interactive variable of canopy cover with tree height and terrain ruggedness were significant at 0.05 level (Table 4.3). The null deviance of 68.086 on 114 df and residual deviance estimated was 58.694 df 109, with an AIC 263.01.

Local scale: There were three variables selected, the best model included an interactive model with canopy cover with shrub height and canopy cover with tree height (Table 4.4). The coefficients estimated for each of the variable were significant at 0.05 level (Table 4.5). The residual deviance estimated were 63.058, df= 112 with an AIC of 265.2553. The ANOVA results suggest that the interactive variable of canopy cover and tree height was the most significant (Table 4.6). The predictor variables are plotted against the dependent in Figure 4.2.

Table 4.2. Variables measured at two spatial scales to assess habitat associations of Red Junglefowl.

Variable name	Description
Local level	
Tree height	Average height of tree in meters
Shrub height	Height of shrubs in meters.
Canopy cover	Measured using a densitometer
Landscape level	
NDVI	Average NDVI for the season
Terrain Rugged	Terrain ruggedness index
Wetness	Topographic wetness index
Disturbance	Human settlements/ weed eradication/fire was scored as 1

Landscape scale: There were four variables selected to the best model (Table 4.2). The best model with an AIC 268.456 was with an interactive variable N:TR along with N and TR (Table 4.4). The coefficients estimated for each of the variable were significant at 0.05 (Table 4.5). The residual deviance 63.720, df = 111. The ANOVA results suggest that the interactive variable of ruggedness and NDVI are significant (Table 4.6). The predictor variables are plotted against the dependent in Figure 4.3.

Table 4.3. Model specific β-coefficient estimates and associated standard errors with P- values with $\alpha = 0.05$* for the global model.

Variables[1]:	β	β (SE)	tvalue	Pr(>|t|)
(Intercept)	1.5852	0.44	3.599	0.00048*
cc_SH	0.0019	0.00098	1.91	0.0587
cc_TH	-0.0005	0.00019	-2.914	0.0043*
N	-3.6839	2.0328	-1.812	0.0727
TR	-1.014	0.0429	-2.362	0.0199*
TR_N	0.4556	0.2657	1.715	0.0892

[1]=Shrub height (SH), Tree height (TH), Canopy cover (cc), NDVI (N), terrain ruggedness (TR) and topographic wetness index (W).

Table 4.4. Effects of variables on abundance estimates for Red Junglefowl

Variables[1]	AIC	Weight
Null	270.08	
Local*		
cc:SH + TH:cc	265.2553	0.081
SH + TH:cc	266.0174	0.055
SH	266.2831	0.048
SH + cc: SH + TH:cc	266.6552	0.040
SH + cc + TH:cc	266.9856	0.034
cc: SH + TH: SH + TH:cc	267.0175	0.033
Landscape**		
N + TR + TR: N	268.456	0.040
TR + W + TR: N + W: N	269.218	0.027
N + TR + TR: N + W:TR	269.441	0.024
N + TR + W + TR: N	269.457	0.024
N + TR + TR: N + W: N	269.714	0.021
W + TR: N + W: N + W:TR	269.762	0.021

*= Total number of Local models = 50, only top six models are given here.

**= Total number of Landscape models = 1050, only top six models are given here

[1]=Shrub height (SH), Tree height (TH), Canopy cover (cc), NDVI (N), terrain ruggedness (TR) and topographic wetness index (W). N=115

Table 4.5. Model specific β-coefficient estimates and associated standard errors with P- values for the best model in the local and landscape scale with $\alpha = 0.05$*.

	Variable[1]	β	β (SE)	tvalue	Pr(>|t|)
Local	(Intercept)	0.646354	0.10048	6.43	3.19e-09 *
	cc_SH	0.002089	0.00099	2.11	0.03701 *
	cc_TH	-0.00054	0.00019	-2.9	0.00495 *
Landscape	(Intercept)	1.61472	0.45435	3.55	0.000559 *
	N	-4.4857	1.99633	-2.24	0.026620 *
	TR	-0.1127	0.04397	-2.56	0.011712 *
	TR:N	0.56202	0.26786	2.09	0.038160 *

[1]=Shrub height (SH), Tree height (TH), Canopy cover (cc), NDVI (N), terrain ruggedness (TR).

Table 4.6. Likelihood ratio test of model for the local and landscape scale

		Df	Deviance residuals	Df	Residual deviance	Pr(>Chi)
	NULL			114	68.086	
Local	cc_SH	1	0.3992	113	67.687	0.39974
	cc_TH	1	4.6285	112	63.058	0.0041**
Landscape	N	1	0.00229	113	68.084	0.94959
	TR	1	1.8366	112	66.247	0.07367
	TR:N	1	2.52714	111	63.72	0.03589*

** = 0.01, * = 0.05 significance

4.4. Discussion

Pattern of abundance:

The present estimates for *G. gallus* abundance are from the protected area of western part of Rajaji NP, there were few detections (22) over an effort of 23.2 at Dehradun FD. Incorporating the same would have biased the estimates for the landscape as management practices and protection status differ for the forest division compared to the protected areas.

Figure 4.2. Response of predictor variables plotted against encounter rate for the local scale model with 95% CI for an interactive variable between canopy cover and shrub height[1] and interactive variable between canopy cover and tree height[2]

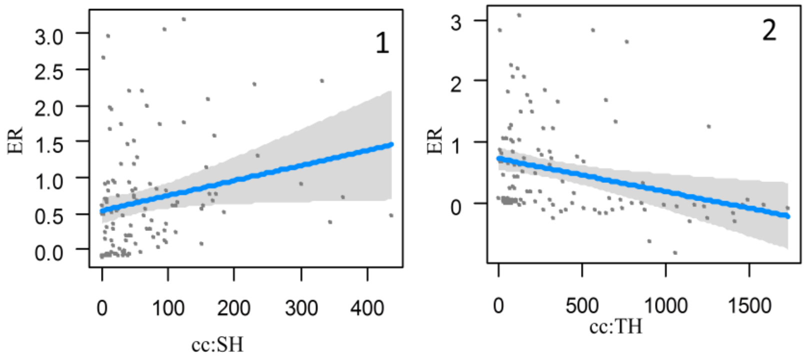
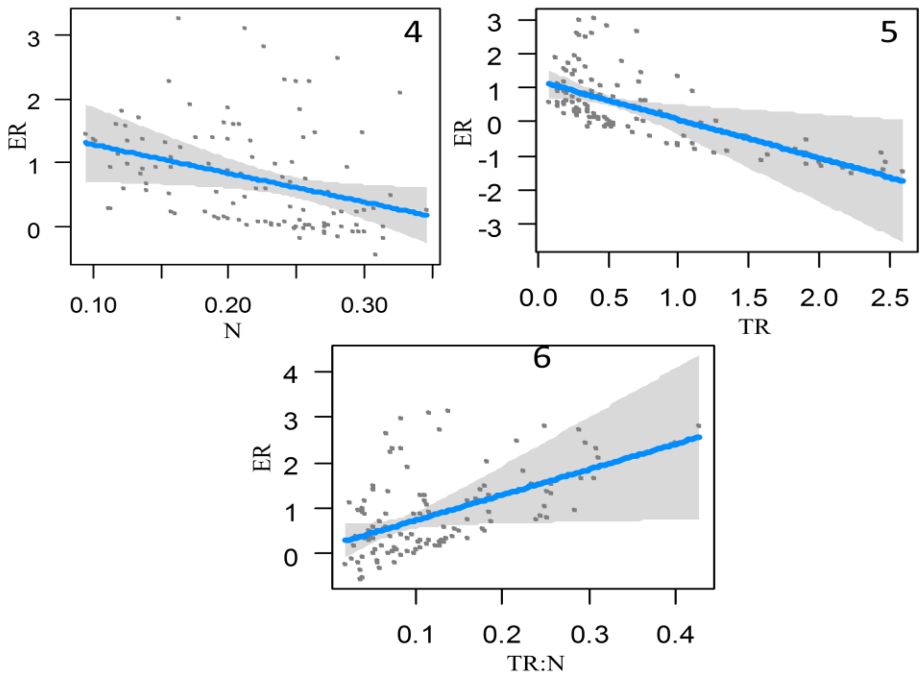


Figure 4.3. Response of predictor variables plotted against encounter rate for the landscape scale model, NDVI[4] (N), Terrain ruggedness[5](TR) and interactive variable of ruggedness with NDVI[6] (TR:N).



Despite being a common species within the Shiwalik landscape there are few comparable estimates for *G.gallus* (Subhani et al., 2010 [7.87/km] and Harihar and Fernandes 2011). The present estimates of density (5.64 and 6.54) lie within the range reported for the chilla range (4.30, 13.54) of Rajaji NP (Harihar and Fernandes 2011). As the *Gujjar* settlements are still present within the western part of Rajaji NP there are pressures are still prevalent in this area. Studies have shown the recovery of species when the anthropogenic pressure has been removed (Harihar et al., 2009). This may have been the reason for the higher densities in the Chilla range.

The sampling from the non-breeding was undertaken during the winter season as previous work within the region suggested that the detection rates and not significantly differ from the breeding season which is the summer months (Fernandes *pers comm*). The estimates of μ was smaller (17.8 ± 5.4) than compared to the breeding season (19.5 ± 3.2), suggesting certain influence of the undergrowth cover on detectability. This may be due to the fact that the undergrowth vegetation is mostly denuded due to poor precipitation as the season progresses from winter to summer and also management practices carried out for the undergrowth begin during early summer.

Factors influencing abundance:

The general guiding principle is that preference coevolves with qualities of the environment and species positively respond through their survival and reproductive success. These preferences are accentuated during the breeding season (Orians and Wittenberger 1991) hence the sampling and factors that influence the species abundance are in context of the breeding season. The quantification of spatial heterogeneity is necessary to elucidate the relationship between ecological processes and spatial patterns thus the measurement, analysis and interpretation of spatial pattern receives attention.

It is important to note, however, that even relatively weakly correlated variables can produce large shared components of variation; hence explanatory variables were chosen where correlation were < 0.50 (Appendix 2). Residual and sequential regression and structural equation modelling all require *a priori* knowledge of the relative functional importance of, or the functional relationships among, the factors in the analysis, they are of little use if this information is unavailable. Here, the knowledge of the natural histories of *G.gallus* provided me with some clues as to the relative importance of some of the factors but did not allow me to clearly prioritise the factors at the given spatial scales. Hence along with pure components interactive components have been used to interpret the relative strengths of associations in explaining the pattern.

G.gallus are ground dwelling and nesting birds hence variable such as shrub height are important parameters as it provides structural cover for the species. *G. gallus* being ground dwelling forage on a wide variety of resources available on the forest floor. The resource availability data was strongly correlated to shrub height (r= 0.67). Like most other galliformes species, *G. gallus* is known to roost in the trees at night and during the day operate on the ground. Hence, the importance of strong canopy cover is essential along with undergrowth vegetation. In older matured forest of Sal *Shorea robusta* the undergrowth vegetation is stunted and hence such habitats would be less preferred, than compared to young forest with dense undergrowth. The interactive variable of canopy cover with tree height showed a negative relationship suggesting avoidance to tall trees with canopy cover. The older forest strands of Sal show a huge gap between the undergrowth vegetation and the canopy cover which may make the species vulnerable to predation during the night.

At the landscape scale the species seems to be negatively influenced by the terrain ruggedness suggesting the preference for plain areas than compared to the hilly tracks. The negative relationship with NDVI suggest the preference of the species for less dense vegetation, while most of the hill track have sparse vegetation or species that are deciduous (Champion and Seth 1968). The more dense vegetation is seen in the plain areas being dominated by pure strands of Sal or mixed Sal forest with *Terminalia* sps and the lower canopy being *Mallotus philippinensis* which are less deciduous (Champion and Seth 1968). The species also shows a positive association with plain areas than compared to rugged terrain.

When the local habitat is abundant and well connected, the influence at the landscape scale has been reported to be less (Andren 1994), although the importance of the landscape has been shown to be important at multiple scales for many taxa (e.g. Lindenmayer et al. 1999, Steffan-Dewenter et al. 2002, Stoner & Joern 2004).

Future studies should address the issue of temporal scale this study is based on data collected in one breeding season, but species-habitat relationships can fluctuate (Wiens 1989), especially for species breeding within the area may occupy different parts of their territory ranges. *G.gallus* can be at greater risk from landscape-scale forest management strategies aimed at controlling invasive species or from large scale disturbances for eg forest fire, which is prevalent within the region.

Plate 2: Red Junglefowl habitats in India.

1. MixedSal Forest
2. Moist evergreen Rain Forest
3. Riverine Forest
4. Bamboo mixed forest
5. Grasslands and Woodlands
6. Mangrove Forest

CHAPTER 5

SPECIES DISTRIBUTION MODELLING

5.1. Background

Proactive response to ameliorate impacts on a species have been proposed keeping in mind scenarios of habitat loss, diseases and climate change (Pimm and Raven 2000). But the fundamental question is from where do we start and from whom? Given this predicament, one way is by knowing where what species are found (Chape et al. 2005) and scenarios with probably predictions of impacts are made such responses have become public policy in many jurisdictions (Stern 2006). Given the predictive outcomes, critical decisions in formulating policies or guiding government investments can be thus made to address the impending problem. Therefore strategies targeted at a landscapes level is aimed at species persistence (Margules and Pressey 2000), despite the lack of population and ecological parameters. Hence, often species distributions are supplemented to calculate the capacity of the landscape (Hanski and Ovaskainen 2000).

In order to obtain species distributions, climate envelopes have been used as it provides an essential conceptual tool for understanding range limits (Gaston 2009), and formulate an useful autecological construct to test ecological and evolutionary theories that underpin the fields of biogeography. Though independent of its implications for inter and intra-specific interactions and community organisation (Soberon 2007). Utilisation of climate envelope to formulate a distribution range is an example of species distribution modelling (henceforth SDM). There are acknowledged technique to determine the distribution range (for eg. Franklin 1995, Anderson et al. 2002, Raxworthy et al. 2003) and have been widely used for various applications such as resource management (for eg., Bradbury et al. 2000, Nams et al. 2006) conservation planning (Ferrier 2002), biodiversity assessment (for eg. Nagendra 2001), reserve designs (Moilanen 2005), habitat management (Carroll et al. 1999), invasive species management (for eg., Peterson 2003), community and ecosystem modelling (Guisan and Thuiller 2005) and modelling climate change scenarios (Sinclair et al. 2010) accompanied with range shifts (Elith et al., 2010a).

Though, niche models that include ‘species’ or ‘environment’ as prefixes to explain niche theory and habitat suitability, predicted in geographical space use ‘predictive habitat distribution’ or ‘spatially explicit habitat suitability’ these are all synonyms that have been used to describe SDM. Often distribution models misinterpret the very concept of ‘*niche*’ (Hutchinson 1957) that requires an understanding of the effects of biotic and abiotic factors on the fitness of the organism (Kearny 2006), while the concept of habitat suitability is similar to resource selection function (Manly et al. 2002) as they explain the proportion of the probability of use, they differ to a SDM where it explains the likelihood (probability) of a species presence (Franklin 2010). But these correlative statistical models in combination with presence data are able to simulate projections of a species distribution in geographical space and not provide a description of the species niche (Austin 2002, Peterson 2007, Soberon 2007). The distributional maps resulting from the application are often referred to as predictions of (species) geographical range (Graham et al. 2004). These models have variously been described as estimating the fundamental (potential) or realised (actual) distribution (Rotenberry et al. 2006, Peterson et al. 2008). While both (potential and realised) distribution refer to the geographical space at a discrete time period, the realised distribution models need ‘absence’ data from environmentally suitable localities ie., true absences caused by scenopoetic (Grinnell) or bionomic (Elton) models (Lobo 2008), but confirming “true absence” for any given site is an arduous task (Gu and Swihart 2004) especially considering issues of detectability. And though these presence-absence framework for eg., Logistic regression, Generalized additive model (Hastie and Tibshirani 1990), Generalized linear model (McCullagh and Nelder 1989), Boosted regression trees (Friedman 2001), Multivariate adaptive regression splines (Friedman 1991) are statistically robust are less suited for a SDM. In order to circumvent the above issues presence-only methods are preferred as they incorporate background or pseudo-absence points as data from location where a species was/is known to occur to create the distribution (Rota et al. 2011). A number of presence-only methods are available for modelling patterns of species distribution (Guisan and Zimmermann 2000, Scott et al. 2002, Guisan and Thuiller 2005) and evaluating the relative performance of different methods still remains an ongoing challenge in ecology and conservation biology (e.g. Loiselle et al. 2003, Thuiller 2003, Ottaviani et al. 2004, Vaughan and Ormerod 2005, Elith et al. 2006, Pearson et al. 2006, Tsoar et al. 2007).

SDM's expresses a quantitative association/relationship between occurrence probability of a species and one or more variable (Dorazio 2011). But the main concern is at what scale are these ecological patterns being studied? Presently, this is being guided by the scale of the data that is being made available (Levin 1992). There is no single straight 'right' strategy or 'sufficient' scale for conservation and management of whatever is left. A change from the pluralist point of view driven by charisma or endangerment criteria for the species is warranted, it is within this ambit that species common will and can be used. But, the information generated through intensive field research on one or more species not only provide useful insight of their ecology, but also helps in generating spatial distribution maps, which could be used to address conservation issues both at the local and landscape levels.

The Red Junglefowl *Gallus gallus* is listed in the "Least Concern" category of IUCN (www.iucn.org, accessed on Dec 20, 2013) with an extent of occurrence (breeding/resident) approx 5,100,000 km2 (www.birdlife.org accessed on Jan 1, 2014) with the subspecies *G g murghi* having a distribution range of 354,978 km^2 (Sathyakumar et al. 2012). Most of the information gathered on the species is either present as a checklist for the area which has been compiled in various avian guide books for the region (Hume and Marshall 1879, Baker 1920, Beebe 1922, Ali and Ripley 1983, Madge and McGowan 2002, Rasmussen and Anderton 2005). While, several techniques have been developed to aid research work in the field, it would take several years to gather the required information. In an attempt to overcome this shortfall for the species presence-only methods are developed that deals with sight location wherein information can be collated using primary, secondary or both types of data to generate probable distribution for the species. Hence this chapter would deliberate on the following objectives

a. Characterise the distribution of the Red Junglefowl in the Indian subcontinent
b. Representation of the Red Junglefowl distribution within the present protected areas network

5.2. Method

Ethics Statement

All research (surveys and sample collection) was conducted with relevant permissions granted from incumbent administrators. Published literature and archived database were used as information for the species.

5.2a. Species Data

Data on the distribution of the species was obtained from primary and secondary sources. The primary source of information came through surveys that were conducted in the different *G. gallus* range states. The secondary source of information was mainly through literature review, questionnaires, and personal communications, and from a network of non-governmental organisations (NGO's) and local villagers. Apart from this museum specimen records (Smithsonian Natural History and Bombay Natural History Society) and archived databases at Global Biodiversity Information Facility (www.gbif.net accessed on September 3, 2009 [GBIF]) and Boakes et al. (2010) were sought to supplement information regarding past records for the species. Records within GBIF were cross-checked for duplication and removed. Geographical biases were corrected for all secondary source of information.

5.2b. Environmental layers

The environmental layers were made up of Bioclimatic, topographical and landcover variables, details of which are listed in the Table.5.1.

Bioclimatic variables

Bioclimatic data was extracted from the worldclim data set at 30 arc seconds Version 1.4 available at http://www.worldclim.org/bioclim.htm (Hijmans et al. 2005). This dataset, ranging over a 50 year period (1950 to 2000) and collected over several globally located weather stations, uses annual trends, extremes and seasonality of temperature and precipitation to derive 19 biologically variables that were used in the analysis.

Topographical variable

Elevation was derived from the Global Digital Elevation Model (GDEM) which was extracted from Advanced Spacehorne Thermal Emission and Reflection Radiometer (ASTER) data. (www. http://www.gdem.aster.ersdac.or.jp/)

Table 5.1. Predictor variables used to estimate potential distribution of Red Junglefowl.

	Variable [units]	Spatial resolution	Data description	Source
Climate	Temperature [^{0}C*10] 11 variables	1 km	1990-2000 monthly	www.worldclim.org
	Precipitation [mm/year] 8 variables	1km	1990-2000 monthly	www.worldclim.org
Topography	Elevation	30mts	AGDEM	www.gdem.aster.ersdac.or.jp
Forest cover	Forest area	30mts	MSS & TM	www.glovis.usgs.gov
Landcover	Landcover types	1km	2009	www. ies.jrc.ec.europa.eu.
Disturbance	Human footprint	1 km	2009	
Productivity	NDVI- 16 days composites	250m	2009	www.glovis.usgs.gov

Landcover variables and Protected Area Network

Global cover type data was extracted from http://ies.jrc.ec.europa.eu/global-land-cover-2009 (Bartholomé and Belward 2005; henceforth referred as GLC 2009). The present protected area database that includes all designated National Parks (NP), Wildlife Sanctuaries (WLS), Conservation Reserves (ConRes) and community reserves (ComRes) was obtained. All protected area within the distribution model was used while eliminating areas such as reservoirs and marshlands. As these habitats are not preferred by the species while inclusion of such areas would affect the measures of vegetation-cover heterogeneity for the region.

Satellite data

Landsat data was selected for the present study because of its high temporal resolution and continuous availability. Images with high cloud cover were not considered for the present study and therefore images from October-June onwards was used in the present study. Landsat Multispectral Scanner (MSS) and Thematic Mapper (TM) images were downloaded from Earth Resource Observation and Science (EROS) data centre (http://glovis.usgs.gov). Details of images used, processing methods and accuracy estimation are given in Appendix 1.

5.3 Presence-only Modelling

Wide ranging vagile organisms may potentially use or occupy the location but not at the time of the survey (issues of detection) and hence in some situations we cannot estimate a sample of unused sites, including georeferenced natural history collections records. The general approach to modelling habitat suitability when data on species presence are available is to calculate some measure of similarity to the presence location by using environmental variables (Elith and Leathwick 2009). There are many modelling algorithms that have been formulated using this approach (Franklin 2010). Presence-only modelling is one type of approach, the advantage of using this approach is that as it does not rely on requiring data of confirmed absence from the specified areas (Li and Guo, 2011; Rota et al., 2011). From an array of presence-only models, the predictive consistence is comparatively better in MaxEnt as it produces robust results with minor errors in location data and irregularly sampled data (Elith et al., 2006; Dudik et al., 2007).

MaxEnt is a machine learning program that uses the principles of maximum entropy to estimate a species niche and potential geographical distribution by relating presence-only data to environmental variables (Phillips et al. 2006). The MaxEnt algorithm *a priori* assumes a uniform distribution and performs a number of iterations in which the weights associated with the environmental variables and functions are adjusted to maximize the average probability of the presence localities expressed as the training gain. These weights are then used to compute the distribution over the entire area (Elith et al., 2010b). The modelled probability is a ‘Gibbs’ distribution (ie., exponential in a weighted sum of the feature (Phillips et al., 2006). The raw output is an exponential function that assigns a probability to each site, where the value is dependent on the number of background and occurrence sites (Phillips et al. 2006). The modelled logistic outputs have a natural probabilistic interpretation representing degree of habitat suitability where '0'indicate less suitable and '1' more suitable (Phillips et al., 2006, Phillips and Dudik 2008)

5.3a. Modelling Procedures

An assumption of SDMs in general is that the entire area of interest has been systematically or randomly sampled (Phillips et al.,2009; Royle et al., 2012) else it leads to a sampling bias. Sampling bias is a concern resulting in over representation of certain environmental features leading to inaccurate models (MacKenzie, 2005; Phillips et al., 2009). Accounting for the independence of the record is also a logical step in MaxEnt models. Hence, to reduce sampling bias and spatial clumping, I use a spatial filter (for eg., Kramer-Schadt et al., 2013) of having a presence record within a radius of 1km as a conservative approach to account for various home range sizes reported in different habitats (Arshad and Zakaria 2011) and ensure spatial independence. A sampling bias grid was created for altitude of more than 3500m and water bodies. These were designated as having "no data" and were automatically excluded by MaxEnt (Phillips et al, 2009). Issues of multi-collinearity, were address by eliminating correlating variables where Pearson's |r | > 0.75 one variable from the group of highly correlated variables was used for the analysis based on the biological significance of the species.

MaxEnt ver 3.3.3.k (http://www.cs.princeton.edu/~schapire/maxent/) with linear and quadratic feature class was selected based on number of presence localities (Phillips and Dudik, 2008) and reduced regularisation value (0.05) was selected to reduce overfitting. These were selected as they constrain the mean, variance and covariance of the respective variables to match their empirical values (Phillips et al 2006). For convergence threshold, maximum iterations (500), background points (10,000). Testing or validation forms an important part to assess the predictive performance of a model. In this case, there was no independent data set for testing the model performance. Therefore I have used the 50-fold partitioning replacement method to create 'training' and 'testing' data this ensures quasi-independent data for model testing (Fielding and Bell 1997, Elith et al., 2010b) and repeated 100 times, thus 100 models were calibrated for the species. Inference was based on average estimates of AUC, predictor importance and prediction maps calculated as the mean probability of occurrence from the 100 models. The threshold was set using minimum training presence and Jackknife estimates was used to compute variable importance.

5.3b. Model Validation

Validation was carried out for the testing and training dataset using sensitivity, specificity analysis for which mean and standard deviations was used.

MaxEnt's heuristic estimated of the relative contribution of environmental variables was considered to the models and the result of jackknife analysis for each environmental layer (Phillips and Dudik, 2008). There are suggested methods to assess the contribution of environmental factors to models namely percentage contribution and permutation importance and Jackknife test. The permutation importance measure depends only on the final model and not the path used to obtain it. The contribution for each variable is determined by randomly permuting the values of that variable among the training points (presence and background) and measuring the resulting decrease in training AUC. A large decrease indicates that the model depends heavily on that variable. Values are normalised to give percentages.

For the Jackknife test, a number of models were created. Each variable was excluded in turn and a model created with the remaining variable. Then the model was created using each variable individually. In addition, a model was created using all variables. For the variables with highest predictive value, response curves show how each of these environmental variable affects predictions.

The ROC curves illustrate how the logistic predictions change as each environmental variable is varied, while keeping all other environmental variables at their average sample value. The curves thus represent the marginal effect of changing exactly one variable. Models were rerun after selecting those variables that contribute at least 5% to the result. This also reduces the total number of variables used in the analysis.

5.4. Result

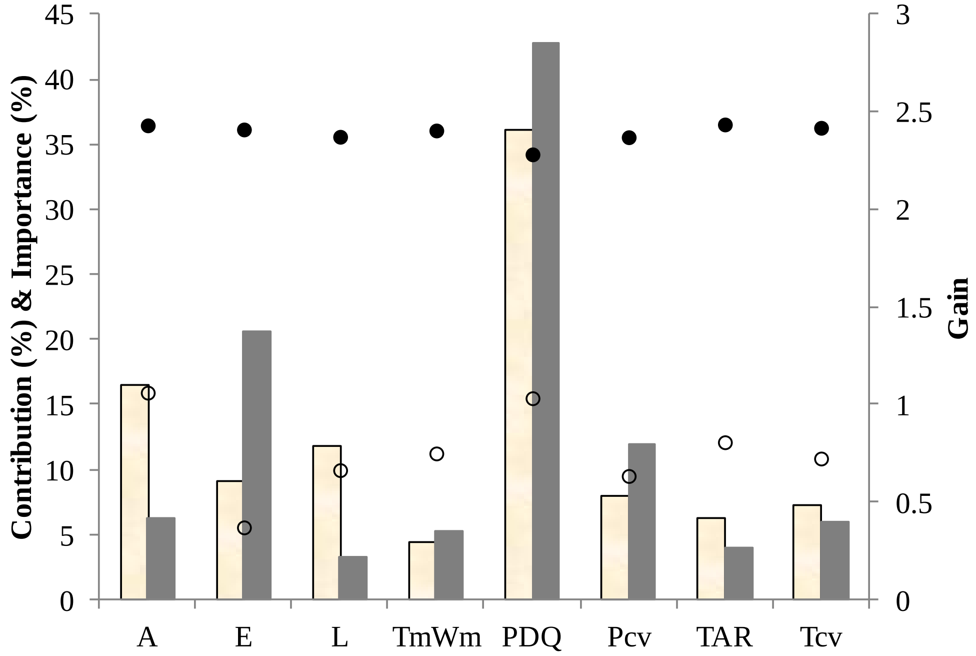
A total of 511 presence records were used for the analysis of which 357 records were used for training and the remaining for testing. There were eight variables that were used to run the final models (Table 5.2). For all models, the area under the curve of the receiver operating characteristics plot (ROC) was high for the training data ranging from 0.968 to 0.978 and test data, ranging from 0.950 to 0.978. AUC values in this range are considered to be informative and indicative of good accuracy (Fielding and Bell 1997).

In *G .gallus* models, based on average of 100 models, precipitation in the dry quarter was the most important variable having a contribution of 36.2% followed by annual mean temperature having a contribution of 16.6%. Based on permutation importance precipitation in the dry quarter was the most significant variable followed by elevation (Figure 5.1). The model internal jackknife test of variable importance in *G. gallus* showed the highest gain when a variable was used in isolation was for precipitation in the dry quarter thus suggesting that this variable contains the most information when used alone, while elevation decreased the gain the most when it was omitted and therefore contained information not present in any other variable (Figure 5.1).

The response curve for precipitation in the dry quarter (PDQ) showed a positive relationship with the logistic output of suitability for *G. gallus*, while elevation had a negative influence with the suitability for the species (Figure 5.2).

Within the land cover classes categories 1 (cultivated and managed terrestrial areas), 2 (Mosaic of areas with natural or semi-natural primary terrestrial vegetation and croplands), 8 (forest with broadleaved, evergreen/deciduous with natural or semi-natural primary terrestrial vegetation), 9 (broadleaved deciduous forest with natural or semi-natural primary terrestrial vegetation having herbaceous vegetation height of 5to >0.5m), 11 (broadleaved deciduous forest/woodland with main layer being natural or semi-natural primary terrestrial vegetation), 17 (broadleaved or needle-leaved, evergreen or deciduous, shrub <5m) and 20 (deciduous forest, sparse vegetation < 15%) had a strong influence on the habitat suitability for the species.

Figure 5.1. Jackknife analysis[1] of variables[2] incorporated in the development of the full model for Red Junglefowl.



Note: Only for test data
1. The variable contribution □, permutation importance ▮ and gain with only variable ○ and without ● the variable.
2. The variables included are, A= annual mean temperature, E= Elevation, L= Land cover, TmWm= maximum temperature in the warm month, PDQ = precipitation in the dry quarter, Pcv= precipitation seasonality, TAR = temperature annual range and Tcv = temperature seasonality.

Predicted distribution

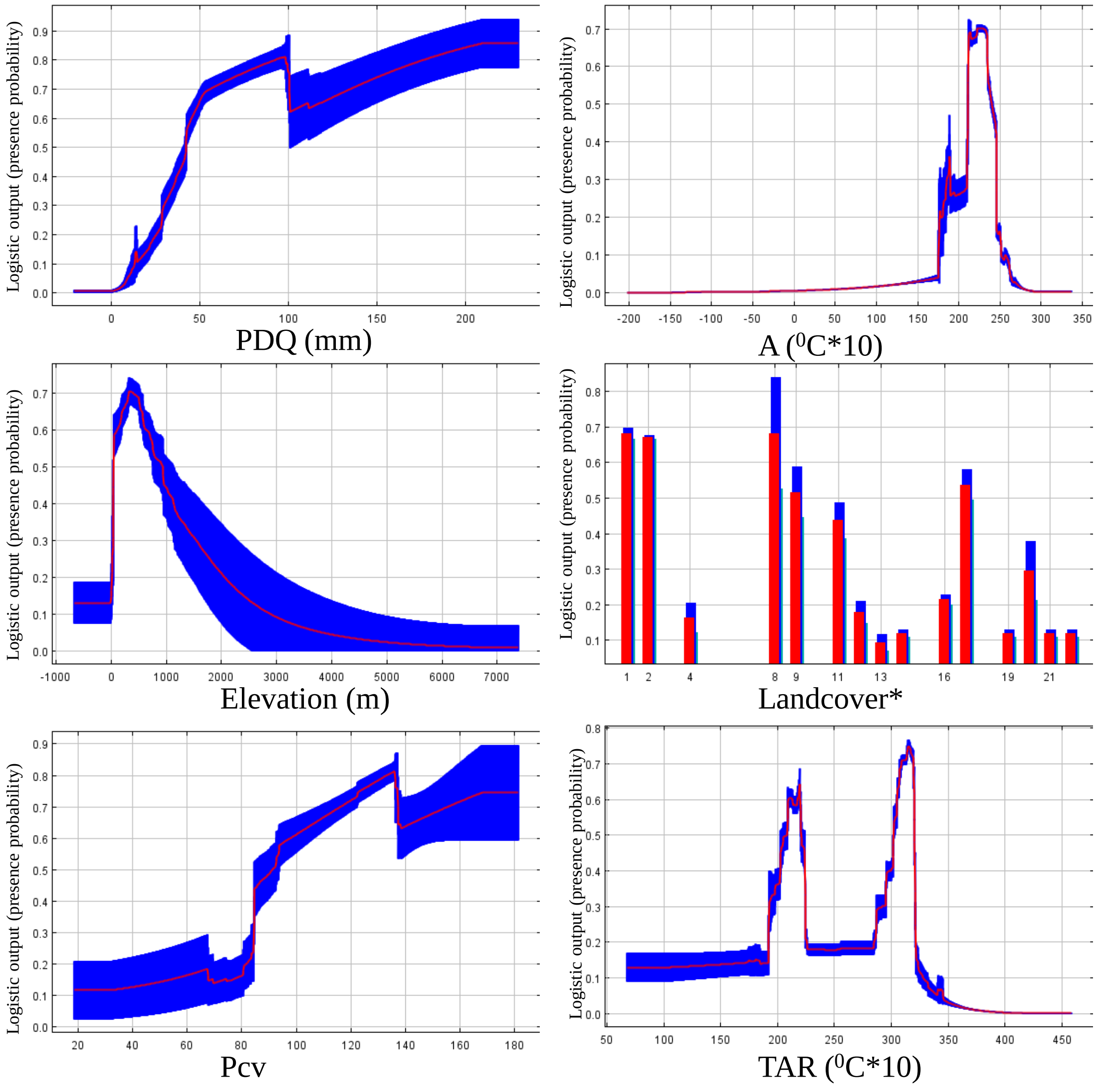
The MaxEnt model generated a map of predicted probabilities of occurrence showing potential suitable areas (Figure 5.3a) , the predictive model estimates the total extent of occurrence as *ca*. 219,312.90 km^2 out of which the range area predicted for India is *ca*. 189,492 km^2 (Table 5.3). The linear function model set the lower threshold to 0.229 and predicted potential suitable habitat for *G. gallus* with an average AUC value of 0.965 ± 0.005. The high suitability area (probability of presence is ≥ 0.6) accounted for 23867.13 km^2 these areas are within the administrative states of Himachal Pradesh, Haryana, Uttarakhand and Uttar Pradesh in the north, in northeast the plains and hill forest of Assam and Arunachal Pradesh and in central India forested areas in Madhya Pradesh and Orissa (Figure 5.3b).

Table 5.2. Bioclimatic profile of Red Junglefowl based on 511 presence locations and summary of environmental variables included in the predictive distribution.

Variable	% contribution	Mean	AUC (with – without) variable
Precipitation in the dry quarter (mm)	36.2	55.34	0.867 – 0.956
Annual mean temperature (^{0}C)	16.6	23.23	0.868 - 0.965
Precipitation seasonality (CV)	8	115.9	0.789 – 0.963
Temperature seasonality (CV*100)	7.4	5051.8	0.815 – 0.964
Temperature annual range (^{0}C)	6.4	27.33	0.835 - 0.965
Maximum temp in warm month (^{0}C)	4.5	35.8	0.823 – 0.964
Elevation (m)	9	440	0.730 - 0.963
Land cover	11.9	-	0.788 - 0.961

From the field surveys, questionnaires survey, literature reviews, informal interviews and databases and the present study reports *G.gallus* from 21 States in India which is in accordance to Hume and Marshall (1879), Ali and Ripley (1983) and Madge and McGowan (2002). It is reported to be 'present' in 205 districts out of the 270 districts of India and in 34 NP and 135 WLS (Appendix 3). The PA network in India, accounts for nearly 13% of the area (47,648.98 km^2), where the NPs represent 4.32% (15,335.84 km^2) and the WLSs 8.52% (30,257.13 km^2).Within the high suitability area nearly 19% of the total area lies within the PA network of India, while the rest of the area is outside the purview of the PA network (Figure 5.4a).

Figure 5.2. Response curves[1] for the most significant predictor variable[2] of potential distribution of Red Junglefowl.



[1].The 95% CI from 100 replicate runs is shown as a blue band.

[2]. The variables are A (⁰C*10) = annual mean temperature, Elevation (m), landcover (based on GLC 2000), PDQ (mm) = precipitation in the dry quarter, Pcv= precipitation seasonality, TAR (⁰C*10) = temperature annual range.

* 1 Cultivated croplands and managed terrestrial areas; 2 = Mosaic of cropland (50-70%) / vegetation (grass/shrub/forest) (20-50%); 4= Mosaic vegetation (grass/shrub/forest) (50-70%) / cropland (20-50%); 8= Closed to open (>15%) broadleaved evergreen or semi-deciduous forest (>5m); 9= Closed (>40%) broadleaved deciduous forest, herbaceous vegetation (5->0.5m); 11= Open (15-40%) broadleaved deciduous forest/woodland (>5m); 12= Closed (>40%) evergreen forest (>5m), trees (main layer); 13= Closed to open (>15%) mixed broadleaved leaved forest (>5m), shrub (5- >0.5m); 14= Mosaic forest /shrub (50-70%) / grassland (20-50%), tree height (>30-3m); 16= Mosaic grassland (50-70%) / forest or shrub (20-50%) undergrowth vegetation 3-0.03m; 17= Closed to open (>15%) (broadleaved or needle-leaved, evergreen or deciduous) shrub (<5m); 19= Closed to open (>15%) herbaceous vegetation , evergreen forest; 20= Sparse (<15%) vegetation, deciduous forest; 21= Closed (>40%) broadleaved forest /shrub flooded with saline/brackish or fresh water and 22= Artificial surfaces and associated areas (Urban areas >50%).

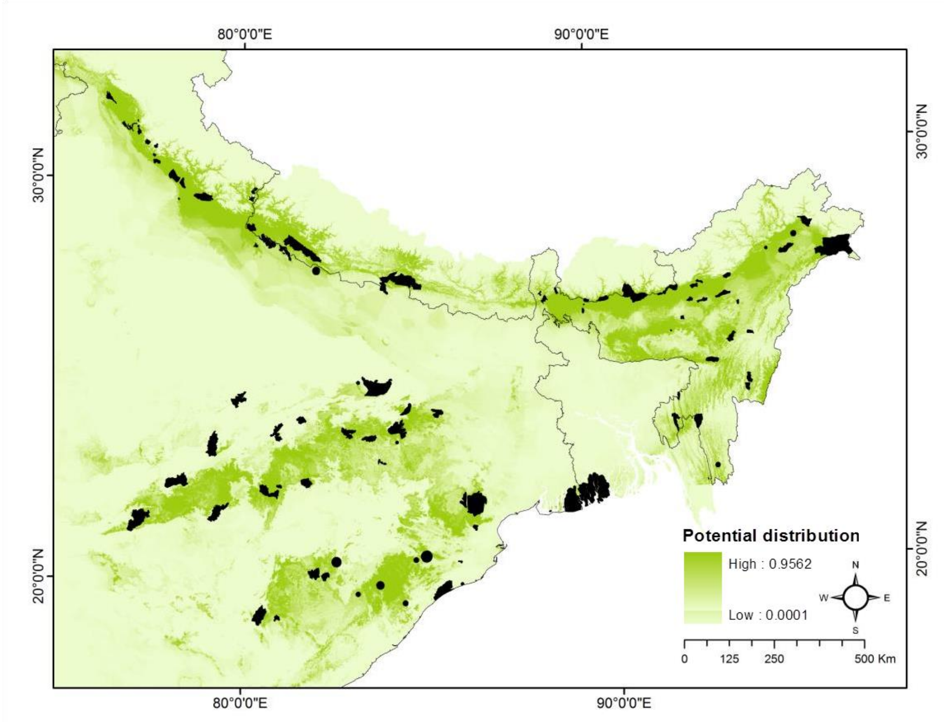
Table 5.3: Area representation for the various land cover class that is available for the Red Junglefowl.

Code[1]	Land class	Global Area (km^2)	India Area (km^2) North	Central	Northeast
1	Cultivated croplands and managed terrestrial areas	25140.60	3192.86	17713.10	1621.94
2	Mosaic of cropland (50-70%) / vegetation (grass/shrub/forest) (20-50%) *	81899.10	8132.83	54682.58	6582.69
4	Mosaic vegetation (grass/shrub/forest) (50-70%) / cropland (20-50%),	5229.00	693.04	1273.05	1316.21
8	Closed to open (>15%) broadleaved evergreen or semi-deciduous forest (>5m)*	821.70	5.19	19.10	142.22
9	Closed (>40%) broadleaved deciduous forest, herbaceous vegetation (5- >0.5m) *	10787.40	1187.55	8434.94	967.80
11	Open (15-40%) broadleaved deciduous forest/woodland (>5m) *	8640.00	0.00	0.00	7380.90
12	Closed (>40%) evergreen forest (>5m), trees (main layer)	12295.80	2533.46	465.89	9118.25
13	Closed to open (>15%) mixed broadleaved leaved forest (>5m), shrub (5- >0.5m)*	2422.80	366.57	449.79	982.74
14	Mosaic forest /shrub (50-70%) / grassland (20-50%), tree height (>30-3m)	1.80	0.73	0.01	0.16
16	Mosaic grassland (50-70%) / forest or shrub (20-50%) undergrowth vegetation 3-0.03m	60912.90	36945.44	567.27	15229.09
17	Closed to open (>15%) (broadleaved or needle-leaved, evergreen or deciduous) shrub (<5m)	8854.20	569.88	2076.50	4766.02
19	Closed to open (>15%) herbaceous vegetation , evergreen forest	69.30	3.10	13.78	43.42
20	Sparse (<15%) vegetation, deciduous forest	1803.60	0.00	1591.20	0.00
21	Closed (>40%) broadleaved forest /shrub flooded with saline/brackish or fresh water	20.70	0.03	8.97	0.00
22	Artificial surfaces and associated areas (Urban areas >50%)	414.00	113.48	178.65	121.87
	Total Area (km^2)	**219312.90**	**53744.14**	**87474.93**	**48273.32**

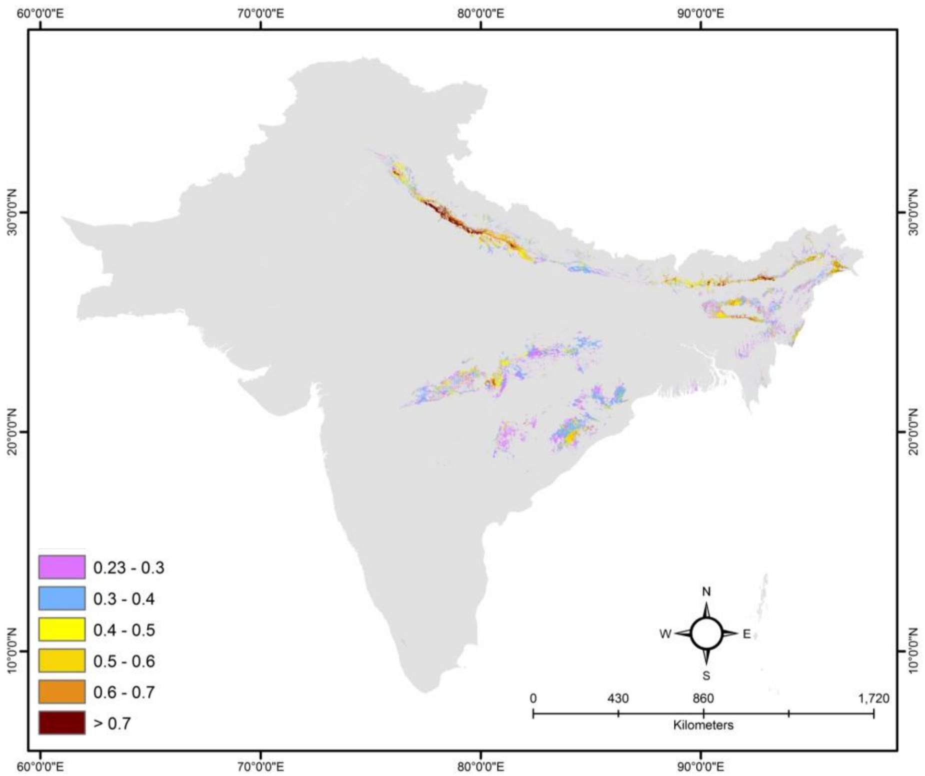
Note .The land class type as per the Global landcover 2000 report.

1= Lancover codes are represented in figure 5. 2; * = Main layer is a Natural and semi-natural primary terrestrial vegetation

Figure 5.3. Predicted distribution of Red Junglefowl in the Indian subcontinent along with protected area network[a] and lower threshold[b].

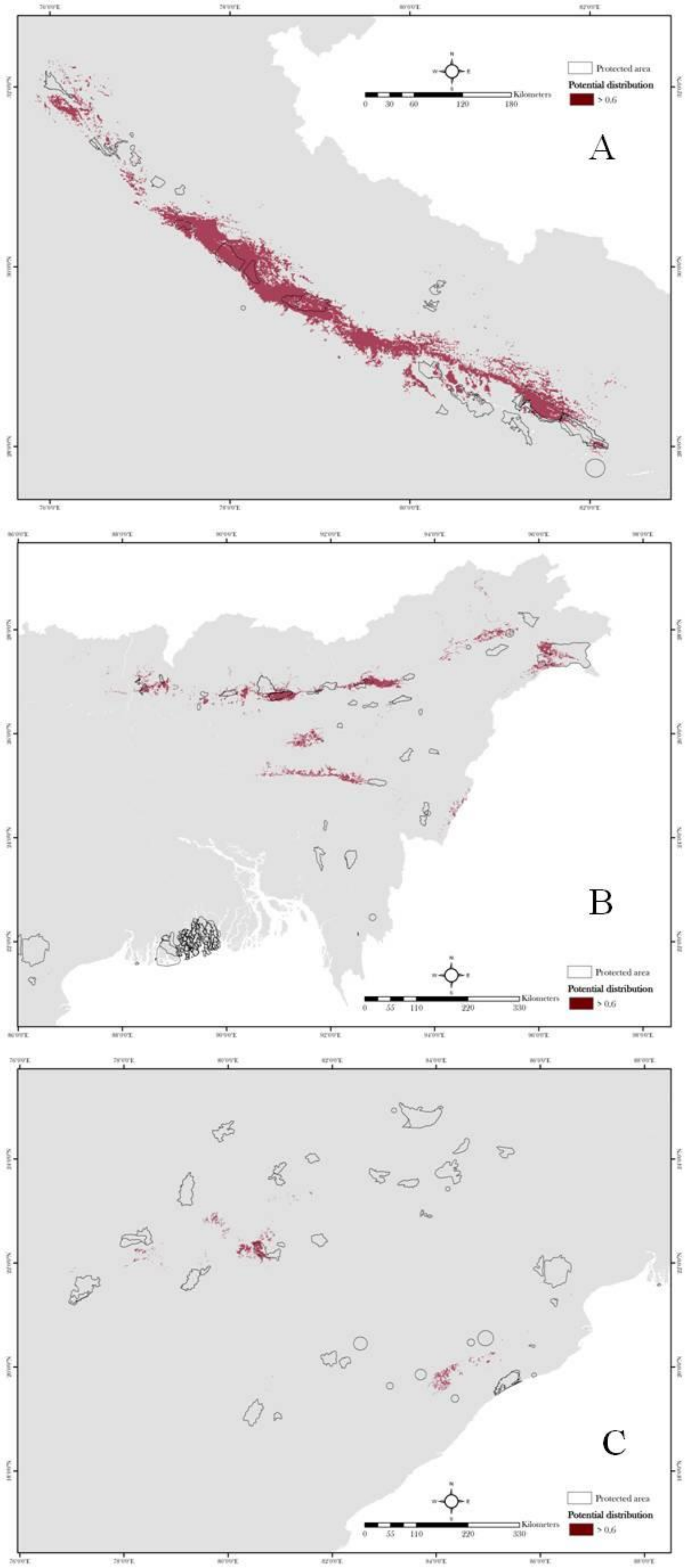


a= predicted potential distribution with PA network.



b = predicted distribution with lower threshold (0.229)

Figure 5.4. Map of high potential distribution (>0.6) of Red Junglefowl along with protected area network for A= north, B=Northeast and C=central India.



5.5. Discussion

The present distribution provide substantial improvement over earlier available distribution datasets (for eg Ali and Ripley 1983; Madge and McGowan 2002) and serves as a representation of the species predicted area of occupancy for practical conservation planning. The parts of the distribution predicted as <0.229 can be interpreted as very low suitability or 'unsuitable' for *G.gallus*. The models depict patterns and provide an understanding of the relevant predictors (natural and anthropogenic features) that have a functional relationship with the ecology and distribution of *G. gallus*. The species being a generalist is able to adapt to a wide range of environmental conditions, despite this precipitation in the dry quarter seems to influence the distribution positively. It is known that *G.gallus* breed during the dry season and precipitation during the dry season will replenish the dried water resources and also influences the seasonal availability of insects that form a major source of food resource for *G.gallus* during the breeding season (Bump and Bohl 1961). Also, burnt forest patch areas will get replenished with younger shoots after precipitation, as a habitat management tool forest burning is practiced in most protected areas to rejuvenate grasslands, control forest fires and weed eradication. *G.gallus* is recorded upto an elevation of 2200m, there are few records of the species in Himachal Pradesh above 2500m elevation, while a few birds are kept in captivity in two centres namely Padmaja Naidu Zoological Park at Darjeeling and the Himalayan Zoological Park at Kufri, Himachal Pradesh. The altitude is known to influence temperatures and hence being a species operating at lower elevations, the annual temperature is within the range of 15-25^{0}C, which is reflected in the annual mean temperature.

Being a forest surrogate species their distribution would be limited with the forest cover. Broadleaved deciduous forest and evergreen forest with herbaceous undergrowth vegetation seems to have a higher probability for the species presence. These forests are characteristics within the Indian subcontinent. Most of the forest landscapes are surround by croplands/agricultural land that provides food resources hence the higher probability for *G. gallus* to utilise these areas. Such behaviour of foraging in croplands adjacent to forested landscape has been reported by many authors. The grasslands with forest and scrub areas have shown the highest areas for the distribution of the species, most of this landcover lies along the foothills of the Shiwalik and Himalayan range from north India to northeast.

Although intitutive, testing climatic determinants for ranges is challenging, owing to the impossibility of experimentation across its broad spatial extent and drawing only correlative evidence to support or refute the idea that these environmental factors are determinants

Predicted distribution

The potential available area of occurrence was estimated to be *ca*. 652,000 km^2 (Fernandes et al., 2009) while the present area of occurrence is *ca* 219,312.90 km^2. Though, the species distribution is predicted to cover 21 administrative states as previously mentioned (Ali and Ripley 1983) the distribution suggests that the once continuous distribution of the species is presently divided into two major landscapes, namely the northern and central. The distribution in central part of India is disconnected from that of the north and northeast. The northern landscape is continuous to the northeast through Nepal and Bhutan (Figure 5.3b).

Within the central landscape, in the state of Madhya Pradesh, presently *G.gallus* is not reported beyond the west bank of the Pench River (R. Jayapal pers comm. 2009) though reported records for the species are from Satpura Hills (Ali and Ripley 1983). Ali and Ripley (1983) demarcate the southern distribution of *G.gallus* to be near Rajahmundry and River Godavari being the barrier, but Nagula et al. (1997) reported the presence of the species at Eturnagaram WS in the Warangal District, while surveys in 2007 did not record the species. Additionally, the species was recorded on the west bank of the River Godavari at Pollavaram WS, in the West Godavari District. In the state of Maharashtra, Gadchiroli District records of the species are from Nawegaon National Park and Nagzira Wildlife Sanctuary. Within the coastal environment broadleaved forest with saline and brackish water at Bhittarkanika in Orissa and Sunderbands of West Bangal and Bangladesh, have the species. It is the only system where *G.gallus* has been recorded. The distribution in north India covering the states of Jammu & Kashmir, Himachal Pradesh, Punjab, Haryana, Uttarakhand and UttarPradesh is linear, with most of the distribution restricted to the forest patches in the foothills of the Himalayas and the Shiwalik range. There are a ten PA within this distribution, it will be important to maintain this connectivity within the forest patches and prevent further isolation of the PAs

(Seiferling et al., 2011) while taking into account the level of changes that have occurred at the molecular level in *G.gallus* for northern India (Mukesh et al. 2011).

Being a forest surrogate species the range of distribution is restricted in certain states due to geographical barriers, environmental elevation gradients and unavailability of forest cover. The later being the main factor that has affected the geographic range for *G.gallus* and hence the distribution is discontinuous. As suggested by McKinney (2007), that a wide ranging species with a large geographic range sizes are more resistant to global extinctions, and persist for longer periods of time than those with smaller ranges this attribute might have assisted in the species to be reported within all earlier reported states. Creation of PA's as a mandate of the state and the ability of a generalist to inhabit wide ranging habitats from mangrove forest to mixed-Pine dominated forest may have also facilitated *G.gallus* to persist. Though *G.gallus*, is a widely distributed species the absence in certain PA's such as Barnawapara WS (Chhattisgarh) seems to be a result of continuous management practices that were undertaken but, addressing and accounting for such human induced interventions in the form of habitat restoration, is difficult (Rouget et al. 2003, Leemans and Serneels 2004). Similarly, the non-detectability of the species in small size PA's of Assam (Berijan, Borajan and Padumoni) seem to suggest that human-induced disturbance may have affected the species, though they are reported from the surrounding tea-estates of the above mention PA's near Digboi and also reported from another smaller PA of the state the Hollangapar Gibbon Sanctuary. Another underlining effect may be the prevalence of hunting that is still practiced and common by the indigenous ethnic groups (Aiyadurai 2007). This may all potentially result in declines in abundance of individuals in parts of the range of the species without necessarily leading to local extinction. The intensity and effects of such activities might have an impact on the local abundance of the species, but due to absence of such baseline data consideration of such plausible effects on the abundance is speculative. Though, these changes may occur and these may pass unnoticed due to apathy for common species especially in areas outside the purview of the PA network there are strong indications that these changed patterns of habitat fragmentation and connection are known to have at least as large an impact on the range of the species (Jetz et al. 2007) therefore the landcover surrounding protected areas (>0.6 probability) is crucial (Lindenmayer et al.,2008, Franklin & Lindenmayer 2009, Arponen et al., 2012) especially while considering *G. gallus* which is known to prefer open spaces and forest edges.

Conservation implications and conclusions

This is the first study where the distribution of a common avian species has been mapped within India. The distribution modelling could enable to identify suitable areas where anticipation of some conservation measures is of huge importance. Since, Maxent is mapping the fundamental niche (different from occupied niche) of the species using bioclimatic variables the suitable habitat for *G. gallus* may be overpredicted in some areas (Pearson 2007). However, the information produced during this study is timely and highly relevant given the potential threats to *G. gallus* (Brisbin 1995). The need and importance for such an objective can be inferred from other common species decline such as Gyps species, when such measures were undertaken when species records had fallen to abysmal low level (Prakash et al. 2003). Prioritising sites for conservation on the basis of coarse-resolution analyses, generally is followed by more detailed analyses within the most valuable coarse-resolution cells (Larsen & Rahbek 2003). Though the present result may seem inadequate such strategies are needed when only coarse-resolution data exist and surveys to increase the resolution of data are feasible only locally.

Although, *G.gallus* is assigned as 'Least Concerned' status by IUCN, there seems to have lost much of their distributional range due to rapidly changing landscapes and given the fact of hunting and hybridisation where the species may become locally extinct. There are some inherent limitations with the methods that have been used. These limitations are mostly with the resolution of the available data. Most of the data used, provide only coarse scale descriptions of the habitat, small-scale features are difficult to capture hence, finer scale interpretations were not feasible. The identification of environmental conditions associated with fine scale habitat variables unlikely to be captured at the landscape level such as predation, nesting sites, refuge habitat could be used to generate predictive spatial models by incorporating intermediate factors such as inter/intra species competition is essential in future studies. Further studies should focus on areas outside the present purview of the PA network to understand how this species copes, especially when there in reserved forests in the country, as a basis for recording rigorous distributional data and updating population status but more essentially keeping the common species common.

CHAPTER 6

RECOMMENDATIONS

There is considerable archaeological evidence to suggest that *G.gallus* were probably domesticated as early as 5400 BCE (West and Zhou, 1988) suggesting that *G.gallus* plays a major role as a progenitors for modern day breeds (Hillel et al. 2003). The close proximity of feral/domestic breed and the effective gene flow to and from the wild counterparts has raised concerns in regards to the purity, possibly loss of the progenitor (Brisbin 1995) and the effect of hybridised individuals in the system as a result of human intervention is of concern (Randi 2008). Apart from this, exotic high yielding breeds have replaced our native breeds. Poultry epidemics, such as the one in Hong Kong in 1998 and the 'bird flu' in India and other parts of SE. Asia in the recent past, could spell doom to the poultry industry and the only fall back option for mankind would eventually be the progenitor *G.gallus*. It is in this background that a project was initiated to understand the genetics diversity, introgression and distribution of *G.gallus*. This study was part of the project and mainly focused in northern India, as distribution was linear (Fernandes et al., 2009) with high genetic diversity (Mukesh et al., 2011) and pure strains (Kanginakudru et al., 2008). Making it imperative to understand factors influencing abundance and prioritising sites where effective conservation management for the species could be undertaken.

Though, this species is still widely common, with high occupancy level in the Shiwalik areas. The understorey was one of the important factors that determined habitat usage. There was no preference to floristic but the structural components played an important part. The structural components are known to provide cover to *G.gallus,* and other gallinaceous species that dwells and nests on the forest floor. Presently, as part of the invasive management strategy which is practiced as a management, emphasis is on eradication by using mechanical ways especially targeted removal of *Lantana camara* as a result the native species of the understorey is also annihilated. This practise thereby creates large empty spaces with no cover for these gallinaceous birds, which is a prime requisite. Hence there should be strategies or plans which may look at regeneration as a possible alternative but keeping in mind that apart from structural component, resource use and availability is also strongly correlated to the understorey.

The timing of the removal should take into consideration the needs and requirements of other ground nesting species other than *G. gallus* as there are many other avian groups apart from galliformes that nest on the ground (for eg. Nightjars).

This study also corroborates the genetic population structure in India (Sathyakumar et al., 2013), and emphasis should be within the northwestern parts of India as the heterogeinity levels are different in Himachal and Haryana.

As this study could not survey all areas within the distribution range, there is a need to increase efforts to understand whether the species is prevalent within forested tracts outside the PA network, where the present distribution is highly fragmented or restricted and also curb hunting and egg collection of this species which continues unabated, especially in central and northeast states. While, states that have restricted distribution *viz;* Sikkim, Andhra Pradesh, Maharashtra and Punjab and fragmented distribution *viz;* Haryana, Bihar and Uttar Pradesh there needs to be increased efforts to understand whether the species is prevalent within forested tracts outside the dominion of the PA network.

Sikkim: *G. gallus* is present in two districts and one PA. The species is reported only from Kitam WLS and forest patches adjoining it (present study). The species was recorded all along the riverine forest of the Rangit River from Naya Bazar upto Melli and in the forest patches leading to Kitam village.

Andhra Pradesh: Papikonda WS was the only PA where this species was recorded. In Papikonda WS, *G.gallus* is known to occur in certain areas and forest tracts (forest between Rampachodavaram and Maranmalli and at Maranmalli the road that leads to Badrachalam), also locals have reported the species at Krupan and Wangasar areas. The present study did not record any presence for *G.gallus* at Rajahmundry, Eturnagaram WLS and Kawal WLS.

Maharashtra: Districts of Chhindwara and Gondiya have reports for this species, but due to logistic constraints surveys were not conducted within this area.

Punjab: There are unconfirmed reports of the species presence in three PA of Punjab (appendix 3), these reports need to be validated.

States of Andhra Pradesh, Jammu & Kashmir and Maharashtra, extensive field surveys should be carried out to ascertain the presence/absence and exact distribution limits of the species as these states encompass the limits or edges of the distribution range of this species. Special focus surveys/studies are required at range overlaps between *G.gallus* and Grey Junglefowl *G. sonneratii*.

Though *G. gallus* is reported from a wide range of habitats the presence of *G.gallus* in mangrove habitats of Sunderband and Bhittarkanika are unique for any pheasant species.

The main threat to *G. gallus* is hybridisation and hence there is an urgent need to assess ranging patterns, survivorship and other basic demographic parameters, population status, genetic variability and purity of *G.gallus* in areas adjoining PAs where most of these forests have multiple uses and feral fowls are in close proximity to *G.gallus.* As ranging parameters and population demographics will provide the necessary understanding that is required to understand the social interactions of *G.gallus* with the feral/domestic fowl. Though, morphology has been used to differentiate between *G.gallus* and feral/domestic fowl there is still a debate as to what are the reliable traits that need to be considered apart from genetic screening. Finally, the heterogeneity levels of *G.gallus* in the landscape need to be checked before releasing the captive bred individuals as this was beyond the scope of the study. But, was part of the earlier recommendation for screening of captive bred individuals.

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APPENDIX 1: Processing of NDVI covariate for non-breeding and breeding season.

Landsat Multispectral Scanner (MSS) images were downloaded from Earth Resource Observation and Science (EROS) data centre (http://glovis.usgs.gov). Images from November-January (non-breeding) and March–May (breeding) with low (<10%) cloud cover were only considered details of images used in this study are given in Table 1A. Acquisition dates of satellite images used has a strong impact on the total area available due to the fact that large sun angle causes the increase in areas with deep shadow that cannot be included for vegetation classification. Hence the images were further converted to an eight-band image for further analysis. From the relevant layer remote sensed information for each sampled grid was extracted using the software ArcMap 9.3.

Table 1A. Landsat images used for the study.

Date of Acquisition	Sensor	Path/Row
November 2009	MSS	148/39
December 2009	MSS	148/39
January 2010	MSS	148/39
March 2010	MSS	148/39
April 2010	MSS	148/39
May 2010	MSS	148/39

Landsat data set contains eight multispectral bands including thermal and panchromatic band that is mostly avoided for vegetation classification. In order to attain a good discrimination among different vegetation classes the Landsat data were corrected for radiometric distortions by converting raw digital number (DN) to at satellite reflectance. For the radiometric correction raw digital value of image was first converted to at-sensor spectral radiance ($L\lambda$) using standardized rescaling factors (Chander et.al, 2009) and finally to top-of-atmospheric (TOA) reflectance (Markham and Barker 1986).

$$L\lambda = Grescale \times Qcal + Brescale$$

$$\rho\lambda = \pi * L\lambda * d2 / ESUN * \sin(\theta)$$

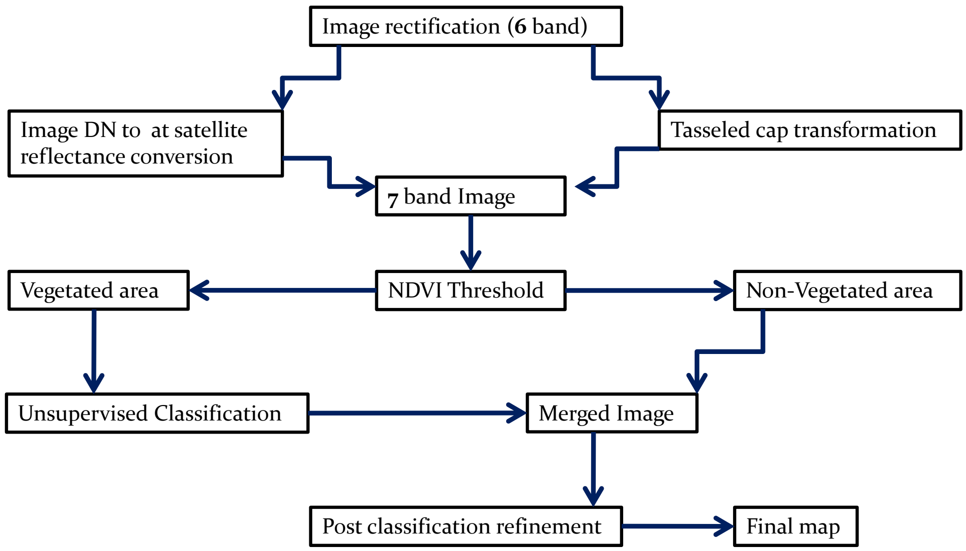
Where, $L\lambda$= TOA radiance, Qcal= Quantized calibrated pixel value, Grescale= Band-specific rescaling gain factor and Brescale= Band-specific rescaling bias factor, $\rho\lambda$= TOA reflectance, d is the normalised Sun-Earth distance, ESUN is Solar irradiance and θ is sun elevation angle. Parameters required for the equation can be found in the header file provided with the image. The Grescale and Brescale have been calculated according to Chander et.al, 2009.

Since, mountain ranges of are known to cause very high spatial variation in plant species assemblages through the impact of elevation, slope and aspect, an Advanced Spacehorne Thermal Emission and Reflection Radiometer (ASTER) global digital elevation model (AGDEM) with multispectral bands was downloaded (http://www.gdem.aster.ersdac.or.jp/) and processed. In order to avoid the impact of diffused sunlight on steep slopes, normalized difference vegetation index (NDVI) calculated from reflectance band along with tasseled cap derived brightness and wetness index were used as additional layers with four multispectral bands. Finally the eight bands composed of four multispectral bands, tasseled cap brightness and wetness index, NDVI and resampled AGDEM at float scale were stacked together for vegetation classification. The methodological framework of classification is given as a schematic representation in Figure 1a.

In the absence of training samples or ground truth data for supervised classification, an unsupervised classification approach through isodata clustering was adapted in the present study. However, water and bare land class is based on supervised classification for which training samples were collected from image itself. First non-vegetated surface were removed from the eight band stacked image with the help of mask function generated from the NDVI. Threshold value for vegetation was selected based on visual inspection of pseudo colour NDVI image overlaid over colour composite of the same area using swipe function. Data collected during field work was used to sort out the clusters into landcover classes.

The most common accuracy assessment elements include overall accuracy, producer's accuracy, user's accuracy and Kappa coefficient. Accuracy of the thematic map was assessed using ground truth data collected during winter and summer (2009-2011). Cover classes with very less and very large representation were not included in the assessment to avoid underestimation or overestimation of the overall accuracy. Kappa tool provided with the Arcview 3.2 was used to perform the accuracy assessment. Error matrices with cross tabulation of the classified (change/no-change) pixels and the reference pixel (actual ground condition) were generated to interpret the accuracy of classification.

Figure 1A. Schematic representation of classification framework.



References:

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http://glovis.usgs.gov.

http://www.gdem.aster.ersdac.or.jp/

APPENDIX 2. Correlations within variables[1] (Pearsons Correlation) N=115

	SH	**UVI**	**CC**	**TH**	**R**	**N**	**E**	**TR**	**A**	**S**	**D**	**W**
SH	1.00											
UVI	0.44	1.00										
CC	-0.23	-0.06	1.00									
TH	-0.11	-0.08	0.14	1.00								
R	0.67*	0.64*	-0.15	-0.06	1.00							
N	0.16	0.35	0.30	0.10	0.16	1.00						
E	-0.25	-0.35	-0.24	0.00	-0.26	-0.44	1.00					
TR	-0.26	-0.45	-0.29	-0.08	-0.31	-0.48	0.85*	1.00				
A	0.09	0.12	0.07	-0.07	0.06	0.09	-0.17	-0.06	1.00			
S	-0.22	-0.39	-0.30	-0.05	-0.28	-0.47	0.70*	0.89*	-0.06	1.00		
D	-0.19	-0.16	0.35	0.19	-0.03	0.02	-0.37	-0.26	-0.01	-0.27	1.00	
W	0.08	0.08	0.31	-0.02	0.17	0.35	-0.57	-0.44	-0.07	-0.48	0.16	1.00

[1]= Variable include SH= Shrub height, UVI = Undergrowth vegetation Index, CC = Canopy cover, TH = Average tree height, R = Resource, N= Average NDVI during the breeding season, E = Elevation, TR = Terrain ruggedness index, A = Aspect, S= Slope, D = disturbance and W= Topographic wetness index

*. Correlation is significant at the 0.05 level.

APPENDIX 3. List of protected areas in India reported with Red Junglefowl *Gallus gallus*

Name of State and Protected Area	**Area (sq km)**	**District (s)**	**Biogeographic Zone / Biotic Province**
Andhra Pradesh			
Papikonda NP	1012.86	East & West Godavari, Khammam	6 (06D)
Arunachal Pradesh			
Mouling NP	483.00	Upper Siang	2 (02D)
Namdapha NP	1807.82	Changlang	2 (02D)
D'Ering Memorial (Lali) WLS	190.00	Upper Siang	2 (02D)
Dibang WLS	4149.00	Dibang Valley	2 (02D)
Eagle Nest WLS	217.00	West Kameng	2 (02D)
Itanagar WLS	140.30	Papum Pare	2 (02D)
Kamlang WLS	783.00	Lohit	2 (02D)
Kane WLS	31.00	West Siang	2 (02D)
Mahao WLS	281.50	Dibang Valley	2 (02D)
Pakke (Pakhui) WLS	861.95	East Kameng	2 (02D)
Sessa Orchid WLS	100.00	West Kameng	2 (02D)
Tale Valley WLS	337.00	Lower Subansiri	2 (02D)
Yordi-Rabe Supse WLS	397.00	West Siang	2 (02D)
Assam			
Dibru-Saikhowa NP	340.00	Tinsukia,Dibrugarh	9 (09A)
Kaziranga NP	858.98	Golaghat, Nagaon, Sonitpur	9 (09A)
Manas NP	500.00	Barpeta, Bongaigaon	9 (09A)
Nameri NP	200.00	Sonitpur	9 (09A)

Amchang WLS	78.64	Kamrup	9 (09A)
Barail WLS	326.25	Cachar Karimgang	9 (09A)
Barnadi WLS	26.22	Darrang	9 (09A)
Burachapori WLS	44.06	Sonitpur	9 (09A)
Chakrashila WLS	45.56	Dhubri	9 (09A)
Dihing Patkai WLS	111.19	Dibrugarh, Tinsukia	9 (09A)
East Karbi Anglong WLS	221.81	Karbi-Anglong	9 (09A)
Garampani WLS	6.05	Karbi-Anglong	9 (09A)
Hollongapar Gibbon WLS	20.98	Jorhat	9 (09A)
North Karbi Anglong WLS	96.00	Karbi-Anglong	9 (09A)
Lawkhowa WLS	70.14	Nagaon	9 (09A)
Marat Longri WLS	451.00	Karbi-Anglong	9 (09A)
Nambor WLS	37.00	Karbi-Anglong	9 (09A)
Nambor Doigrung WLS	97.15	Karbi-Anglong	9 (09A)
Porbitora WLS	38.81	Marigaon	9 (09A)
Sonai Rupai WLS	220.00	Sonitpur	9 (09A)
Bihar			
Valmiki NP	335.65	Pashchim Champaran	7 (07B)
Bhimbandh WLS	681.99	Munger	6 (06B)
Gautam Budha WLS	138.34	Gaya	6 (06B)
Kaimur WLS	1342.00	Rohtas	6 (06A)
Udaipur WLS	8.87	Pashchim Champaran	7 (07B)
Valmiki WLS	545.15	Pashchim Champaran	7 (07B)
Chhattisgarh			
Indravati (Kutru) NP	1258.37	Dantewada	6 (06C)
Kanger Valley NP	200.00	Bastar	6 (06C)
Achanakmar WLS	551.55	Bilaspur	6 (06A)

Badalkhol WLS	104.45	Jashpur	6 (06B)
Bhairamgarh WLS	138.95	Dantewada	6 (06C)
Bhoramdev WLS	163.80	Kawardha	6 (06A)
Sarangarh-Gomardha WLS	277.82	Raigarh	6 (06C)
Pamed Wild Buffalo WLS	262.12	Dantewada	6 (06C)
Semarsot WLS	430.36	Surguja	6 (06B)
Sitanadi WLS	553.36	Dhamtari	6 (06C)
Tamor Pingla WLS	608.53	Surguja	6 (06B)
Udanti Wild Buffalo WLS	247.59	Raipur	6 (06C)
Haryana			
Kalesar NP	46.82	Yamuna Nagar	4 (04A)
Abubshehar WLS	115.30	Sirsa	4 (04A)
Kalesar WLS	54.06	Yamuna Nagar	4 (04A)
Morni Hills (Khol-Hi-Raitan) WLS	48.83	Panchkula	4 (04A)
Himachal Pradesh			
Simbalbara NP	27.88	Sirmaur	2 (02B)
Chail WLS	109.00	Solan	2 (02B)
Churdhar WLS	66.00	Sirmaur	2 (02B)
Daranghati WLS	176.12	Shimla	2 (02B)
Darlaghat WLS	6.00	Solan	2 (02B)
Dhauladhar WLS	1154.86	Kangra	2 (02A)
Gobind Sagar WLS	100.00	Bilaspur	2 (02A)
Majathal WLS	57.55	Solan	2 (02B)
Pong Dam Lake WLS	307.00	Kangra	2 (02A)
Renukaji WLS	4.50	Sirmaur	2 (02B)
Shilli WLS	2.00	Solan	2 (02B)
Shri Nainadevi WLS	123.00	Bilaspur	2 (02B)

Jammu & Kashmir			
Jasrota WLS	25.75	Kathua	4 (04A)
Nandni (Nandini) WLS	33.34	Jammu	4 (04A)
Ramnagar Rakha WLS	31.50	Jammu	4 (04A)
Surinsar Mansar WLS	55.50	Jammu	4 (04A)
Trikuta WLS	31.77	Jammu	2 (02A)
Bahu Con R	19.75	Jammu	2 (02A)
Jharkhand			
Betla NP	226.33	Latehar	6 (06B)
Dalma WLS	193.22	East Singhbhum, Saraikela	6 (06B)
Hazaribagh WLS	186.25	Hazaribagh	6 (06B)
Gautam Budha	121.14	Koderma, Hazaribagh	6 (06B)
Koderma WLS	177.35	Koderma	6 (06B)
Lawalong WLS	211.03	Chatra	6 (06B)
Mahauadanr WLS	63.26	Latehar	6 (06B)
Palamau WLS	752.94	Latehar	6 (06B)
Palkot WLS	182.83	Gumla, Simdega	6 (06B)
Parasnath WLS	49.33	Giridih	6 (06B)
Topchanchi WLS	12.82	Dhanbad	6 (06B)
Madhya Pradesh			
Bandhavgarh NP	448.85	Umaria,Katni	6 (06A)
Kanha NP	940.00	Mandla, Balaghat, Dindori	6 (06A)
Madhav NP	375.22	Shivpuri	4 (04B)
Pench (Priyadarshini) NP	292.85	Seoni, Chhindwara	6 (06D)
Sanjay NP	466.88	Sidhi	6 (06A)
Bagdara WLS	478.00	Sidhi	6 (06A)

Panpatha WLS	245.84	Umaria	6 (06D)
Pench WLS	118.47	Seoni, Chhindwara	6 (06A)
Phen WLS	110.74	Mandla	6 (06A)
Sanjay Dubri WLS	364.59	Sidhi	6 (06A)
Maharashtra			
Nawegaon NP	133.88	Gondia	6 (06D)
Nagzira WLS	152.81	Bhandara	6 (06D)
Manipur			
Yangoupokpi Lokchao WLS	184.40	Chandel	9 (09B)
Meghalaya			
Balphakram NP	220.00	South Garo Hills	9 (09B)
Nokrek Ridge NP	47.48	East, West & South Garo Hills	9 (09B)
Nongkhyllem WLS	29.00	Ri Bhoi	9 (09B)
Siju WLS	5.18	South Garo Hills	9 (09B)
Mizoram			
Murlen NP	100.00	Champhai	9 (09B)
Phawngpui Blue Mountain NP	50.00	Lawngtlai	9 (09B)
Dampa WLS	500.00	Mamit	9 (09B)
Khawnglung WLS	35.00	Serchhip	9 (09B)
Lengteng WLS	60.00	Champhai	9 (09B)
Ngengpui WLS	110.00	Lawngtlai	9 (09B)
Pualreng WLS	50.00	Kolasib	9 (09B)
Tawi WLS	35.75	Aizawl	9 (09B)
Thorangtlang WLS	50.00	Serchhip	9 (09B)
Tokalo WLS	250.00	Saiha	9 (09B)
Nagaland			
Intanki NP	202.02	Dimapur	9 (09B)

Fakim WLS	6.41	Tuensang	9 (09B)
Puliebadze WLS	9.23	Kohima	9 (09B)
Rangapahar WLS	4.70	Dimapur	9 (09B)
Orissa			
Bhitarkanika NP	145.00	Kendrapara	8 (08B)
Simlipal NP	845.70	Mayurbhanj	6 (06B)
Usakothi WLS	304.03	Sambalpur	6 (06B)
Baisipalli WLS	168.35	Nayagarh	6 (06C)
Balukhand Konark WLS	71.72	Puri	8 (08B)
Bhitarkanika WLS	525.00	Kendrapara	8 (08B)
Chandaka Dampara WLS	175.79	Khurda, Cuttack	6 (06B)
Debrigarh WLS	346.91	Sambalpur	6 (06B)
Hadgarh WLS	191.06	Keonjhar, Mayurbhanj	6 (06B)
Karlapat WLS	147.66	Kalahandi	6 (06C)
Khalasuni WLS	116.00	Sambalpur	6 (06B)
Kotagarh WLS	399.50	Phulbani	6 (06C)
Kuldiha WLS	272.75	Balesore	6 (06B)
Lakhari Valley WLS	185.87	Gajapati	6 (06C)
Nandankanan WLS	14.16	Khurda	6 (06C)
Satkosia Gorge WLS	745.52	Angul, Boudh, Cuttack	6 (06B)
Simlipal WLS	1354.30	Mayurbhanj	6 (06B)
Sunabeda WLS	500.00	Nuapada	6 (06C)
Punjab			
Abohar WLS	186.50	Firozpur	4 (04A)
Kathlaur Kushlian WLS	7.58	Gurdaspur	4 (04A)
Takhni-Rehampur WLS	3.82	Hoshiarpur	4 (04A)
Sikkim			

Kitam WLS	6.00	South Sikkim	2 (02C)
Tripura			
Clouded Leopard NP	5.08	West Tripura	9 (09B)
Bison (Rajbari) NP	31.63	South Tripura	9 (09B)
Gumti WLS	389.54	South Tripura	9 (09B)
Rowa WLS	0.86	North Tripura	9 (09B)
Sepahijala WLS	13.45	West Tripura	9 (09B)
Trishna WLS	163.08	South Tripura	9 (09B)
Uttar Pradesh			
Dudhwa NP	490.00	Lakhimpur-Kheri	7 (07A)
Katerniaghat WLS	400.09	Bahraich	7 (07A)
Kishanpur WLS	227.00	Lakhimpur-Kheri, Shahjahanpur	7 (07A)
Sohagibarwa WLS	428.20	Maharajganj	7 (07A)
Sohelwa WLS	452.47	Shravasti,Balrampur	7 (07A)
Uttarakhand			
Corbett NP	520.82	Nainital, Pauri Garhwal	7 (07A)
Rajaji NP	820.42	Dehradun, Pauri Garhwal, Haridwar	7 (07A)
Sonanadi WLS	301.18	Pauri Garhwal	7 (07A)
Asan Wetland Con. Res.	4.44	Dehradun	7 (07A)
Jhilmil Jheel Con. Res.	37.84	Haridwar	7 (07A)
West Bengal			
Buxa NP	117.10	Jalpaiguri	7 (07B)
Gorumara NP	79.45	Jalpaiguri	8 (08B)
Sunderban NP	1330.10	North & South 24-Paraganas	8 (08B)
Haliday Island WLS	5.95	South 24-Paraganas	7 (07B)
Jaldapara WLS	216.51	Jalpaiguri	8 (08B)

Lothian Island WLS	38.00	South 24-Paraganas	7 (07B)
Mahananda WLS	158.04	Darjeeling & Jalpaiguri	8 (08B)
Sajnakhali WLS	362.40	South 24-Paraganas	2 (02C)
Senchal WLS	38.88	Darjeeling	4 (04A)

Note: There is no confirmation from the highlighted protected areas regarding the species presence.

Conservation of red junglefowl *Gallus gallus* in India

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Abstract The red junglefowl (RJF) is one of the most important species for mankind, due to economic and cultural reasons. Recently, fears have been expressed that the wild RJF may be genetically contaminated, leading to an inference that there may not be any pure RJF left in the wild. In order to assess the distribution of RJF in India, field surveys were carried out and secondary information was collated. Historically, RJF occurred in 270 districts in 21 states across India, but now it is found in 205 districts in the 21 states. Of the 255 Protected Areas (PAs) that occur within the RJF's distributional range in India, 190 PAs (31 National Parks [NPs] and 159 Wildlife Sanctuaries [WSs]) have reported its presence. A composite set of trait characters that are presumed to be indicators of wild RJF was used for characterising RJF in the field. A total of 563 (293 males and 270 females) RJF were characterised of which 7% of birds in the central region had reports of white ear patch. Eclipse plumage was observed in wild and captive birds. Ninety-two RJF samples and twenty five domestic chicken samples were collected and processed for DNA extraction. Thirty highly polymorphic microsatellite markers were utilised for RJF and domestic chicken genotyping. Preliminary studies showed polymorphism within RJF at these microsatellite loci.

Keywords DNA extraction, hybridisation, microsatellite markers, PCR, red junglefowl, traits.

Introduction

The red junglefowl *Gallus gallus* (RJF), the ancestor of the domestic chicken, is one of the most important species for mankind, due to its economic and cultural significance. Liu et al., (2006) suggested that there are distinct distribution patterns and expansion signatures suggesting that different clades may originate from different regions, which support the theory of multiple origins in South and Southeast Asia. The present day, multi-billion dollar poultry industry is based on the RJF and may have to depend on it in the future. Andersson et al. (1994) stated that 'populations of domestic animals and their wild ancestors provide a valuable source of genetic diversity that may be exploited to develop animal models for quantitative traits of biological and medical interest'. Hence conservation of genetically pure wild forms or their representatives have great potential to make a significant contribution to the study of some economically important genetic traits (Brisbin et al., 2002).

The RJF is widely distributed and its five sub-species are spread from the Indian sub-continent eastwards across Myanmar, South China, Indonesia to Java (Johnsgard, 1986). In India, two sub-species occur, the type specimen, *Gallus gallus murghii* and *Gallus gallus spadiceus* (Ali & Ripley, 1983). While the former is found in the north and central part of India, extending eastwards to Orissa and West Bengal, the latter is confined to the north eastern parts of India. Recently, fears have been expressed that the wild RJF populations may be genetically contaminated with domesticated chickens, leading to an inference that there may not be any pure RJF populations in the wild (Peterson & Brisbin, 1998), causing introgression of domestic genes into wild birds. An analysis of skins by Peterson & Brisbin (1998) showed a lack of phenotypic traits, which characterise true wild RJF, as described by Morejohn (1968).

This study investigated the status of RJF in India and aimed to identify ways to safeguard remaining pure wild birds. In 2006, a collaborative research project on the

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RESEARCH ARTICLE

Genetics Driven Interventions for *Ex Situ* Conservation of Red Junglefowl (*Gallus gallus murghi*) Populations in India

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Genetics driven interventions (GDI) are imperative for *ex situ* conservation to exhort long-term sustenance of small and isolated populations in captivity as they are more prone to an increased extinction risk due to inbreeding and genetic drift. We investigated constitutive genetic attributes of four captive Red Junglefowl (RJF) populations in India, to facilitate the prioritization of the birds to formulate an effective breeding action plan. All the four RJF populations were found to be evident of significant inbreeding but none of them had exhibited any signature of bottleneck footprints in the recent past. Bayesian cluster analysis revealed three distinct groups among the four captive RJF populations. Interestingly, birds of Kufri population were assigned together with Gopalpur as well as with Morni populations, indicating their shared genetic ancestry. Among the four populations, Morni population displayed the richest genetic attributes and was therefore presumed as a key source of genetic variation. Nine birds of Morni population were relatively pure (q-value >0.98) and carried about 50% of the total private alleles of Morni population. Thus, being the foremost reservoir of allelic diversity, these nine birds may be selected for launching alien alleles to other RJF populations to rescue their loss of genetic diversity arising from inbreeding. Zoo Biol. XX: XX–XX, 2013.

Keywords: ***ex situ*** **conservation; microsatellite marker; genetic diversity; inbreeding; bottleneck signature; red junglefowl** ***Gallus gallus murghi***

INTRODUCTION

Genetic assessment for captive populations is the key step towards effective formulation and prioritization of breeding policies in any conservation action plan. A few studies have provided direct evidence for an association between genetic variability and reproductive performance in natural populations [Keller et al., 1994; Briskie and Mackintosh, 2004]. Inbreeding often arises in small and genetically isolated or closed populations which tend to lose genetic variability over time and thus may in turn increase the probability of extinction or reduce the opportunity for future adaptive change [Jiménez et al., 1994; Meffe and Carroll, 1994]. Inbreeding is therefore a natural, unavoidable process, and accumulates over time in the genetically isolated or closed populations [Frankham, 2005]. Certain events such as selection and genetic bottleneck can accelerate the rate of inbreeding due to the limited number of individuals that contribute to future generations. Populations that have undergone a genetic bottleneck tend to lose rare alleles quickly and show a temporary excess in observed heterozygosity to the heterozygosity expected from the observed number of alleles at a locus under mutation-drift equilibrium (Heq), which forms the basis for detection of bottleneck signature [Cornuet and Luikart, 1996]. In the past, many populations in zoos died out due to reduced fitness associated

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