

**ASSESSMENT OF LANDSCAPE PATTERN FOR MODELLING HABITAT
SUITABILITY FOR LIONS AND PREY SPECIES IN GIR PROTECTED
AREA, GUJARAT**

THESIS
SUBMITTED TO THE
FOREST RESEARCH INSTITUTE UNIVERSITY
DEHRADUN, UTTARAKHAND

For

THE AWARD OF THE DEGREE OF
DOCTOR OF PHILOSOPHY IN FORESTRY
(WILDLIFE SCIENCES)



By
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WILDLIFE INSTITUTE OF INDIA
DEHRADUN, UTTARAKHAND
2013

DECLARATION

I hereby declare that the dissertation "Assessment of landscape patterns for modeling habitat suitability for lions and prey species in Gir Protected Area, Gujarat" is original research conducted by me under the supervision of Shri Qamar Qureshi and Dr. Yadvendradev V. Jhala of the Wildlife Institute of India. The thesis has been submitted to the Forest Research Institute University for the award of the degree of Doctor of Philosophy in Forestry (Wildlife Science), and has not formed the basis for the award of any other degree. It embodies my own work and observations, and in that respect the investigation appears to advance knowledge on the subject.

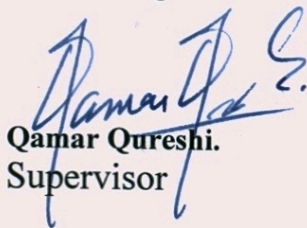
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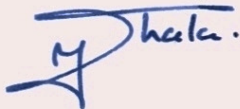


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CERTIFICATE

This is to certify that the dissertation "Assessment of landscape patterns for modeling habitat suitability for lions and prey species in Gir Protected Area, Gujarat", submitted to the Forest Research Institute University, embodies original research conducted by Miss. Parabita Basu under our supervision. This work has not been submitted, in part or full, for any other degree. The dissertation fulfills the partial requirements for the award of a Ph.D. degree in Forestry (Wildlife Science) as specified in the ordinance of FRI University.

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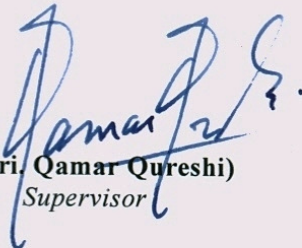
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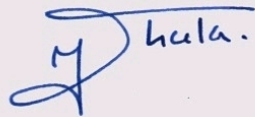
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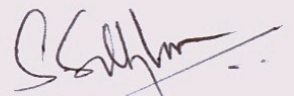
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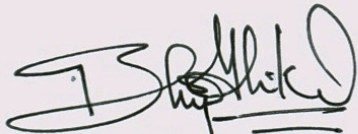

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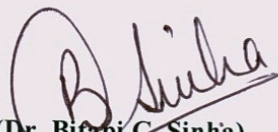

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7-9 February, 2012

**THEME: Geo-Budget: Enabling
Sustainable Growth**

Paper Reference Number: PN-40

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Assessment of the future of potential dispersal corridors of Asiatic Lions (*Panthera leo persica*) across the agro-pastoral landscape between Gir & Girnar Wildlife Sanctuary.

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Abstract

Wide ranging species are susceptible to fragmentation effects and small isolated populations become unviable due to environmental stochasticity & inbreeding effects. The recently colonized Girnar lion population is a typical example where the expansion of human settlements along the dry river channels and conversion of scrub patches into agricultural & urban areas results in substantial impacts on the functionality of dispersal corridors between the lion source population of Gir and Girnar. Monitoring these changes and planning urban development can be successfully achieved using multitemporal remotely sensed data, spatial metrics, and modeling. Three land cover maps (1998, 2002 and 2009) were used to study the patterns and quantities of land use dynamics. This agro-pastoral landscape is now being modified to towns, dams and roads. Results indicate increasing decline in vegetative patches from 1998-2002 (14%) and 2002-2009 (57%) with an increase in areas under barren land & settlements (90%) from 1998-2009. Land use change along the dispersal corridors was modeled using a Cellular Automata based approach for the year 2015. The predictive power of the model was successfully validated using Kappa variations. Projected land cover changes show a growing tendency in urban land use, threatening areas used by animals as passageways and day refugia. There is an urgent need for protecting the remaining natural, undisturbed vegetation patches thereby facilitating dispersal of lions between Gir and Girnar wildlife sanctuary in order to ensure long-term viability of the small Girnar lion population.

Keywords: Fragmentation, corridors, land-use dynamics, land-cover change modeling, dispersal.

Introduction

Gir remained connected with Girnar, Mitiyala, Barda, Alech hills, Dhank and Chorwad by corridors of rough semi-wooded forests, grasslands, and sparsely populated villages till the early part of the 19th century. This enabled lions to move freely in this region. Lions become locally extinct from Barda and the Alech hills towards the later half of the last century (Singh 2007). They disappeared from Girnar and Mitiyala by 1963 and 1955. By 1965, lions were found only in the compact forest of the Gir. The agro-pastoral landscape of Saurashtra that covers most of the distributional range of Asiatic lions (*Panthera leo persica*) have been vulnerable at the expense of human driving forces like demand of land for agriculture, pasture, logging, limestone mining and urban sprawl. Soil

and drainage conditions made this land more suitable for a variety of agriculture and urban land uses. As a result, expansion of urban, residential, agricultural land uses in this landscape have eliminated and fragmented the habitat available to Asiatic lions.

The correlated processes of habitat loss and fragmentation are probably the most important threats to global biodiversity. Historically, studies of fragmented ecosystems have relied heavily upon the conceptual model of island biogeography theory (MacArthur & Wilson 1967). In due course of time, the simple Island biogeography theory has been superseded by landscape ecology (Forman & Godron 1986), which further emphasizes factors such as fragment shape, spatial configuration of fragments, movement corridors, and structure and composition of the surrounding matrix. Metapopulation theory was also helpful in understanding the responses of subdivided populations to fragmentation (Hanski & Gilpin 1996).

Studies have yielded important insights into the responses of various taxa to fragmentation and into the effect of fragment size, shape, connectivity and other landscape features on species assemblages and ecological processes (Debinski & Holt 2000; Saunders et al. 1991; Laurance & Bierregaard 1997). Because lions have extensive home ranges, large contiguous forested areas are required to sustain viable populations. The most recent estimate of the total lion population of 411 individuals (Gujarat Forest Department, 2010) exist as one large (297) population in Gir PA and other small populations of 10-50 individuals including Girnar (24). These smaller populations can only persist in a metapopulation framework (Banerjee et al. 2010). Therefore, understanding the patterns of movement of individuals between subpopulations is essential managing the dynamics of the metapopulation structure to ensure long-term survival of the species (Gilpin and Hanski, 1991; Hanski and Gilpin, 1997).

In Gir, the population of lions remained almost constant during last censuses (267 in 1990, 262 in 1995 and 297 in 2010) (Singh 1995; Gujarat Forest Department, 2010). The population of important ungulates has increased at the rate of 14.2% per year during the last three decades and food is not a major limiting factor (Singh 1995). Lions being territorial, are socially regulated (Schaller 1972) resulting in dispersal out of the Gir protected area (PA). Natural dispersal of lion started after the last draught in 1987. The emigrant lions take advantage of the riverine habitat that exists between Gir and the other satellite populations. Some of the groups intermingle with the parent population in Gir whereas others remain confined to the isolated sites (Singh 1995).

Gir-Girnar corridors provide an important biological linkage between Gir Wildlife Sanctuary (WLS) and the Girnar WLS. Discontinuous cover and the scattered distribution of shrubs play an important role in lion's use of this landscape though it has been observed that there is a persisting problem of unrestrained encroachment and urban sprawl in this landscape disrupting the functionality of corridor habitats. However, the extent of land use changes has not yet been quantified. The present study identified different land use changes overtime in this landscape.

Study Site

The study area comprises Girnar WLS and the adjacent human dominated landscape connecting it with the Gir WLS. (Figure 1). The landscape on the south-eastern fringe of the Girnar WLS includes the revenue lands of approximately 90 villages extending in east from Malinda to Ravani Mundy and in western side from Khadiya to Nataliya. This human governed landscape matrix consists of broken topography, small drainage systems, Government and private owned wastelands, fallow lands and is predominantly agricultural with seasonal crops and fruit (mango) orchards. Climate is generally dry with two main seasons' viz. summer and winter. The monsoon which has

few rainy days fuses with the summer period. The principal rivers flowing through the area are Ozat and Gundajali. Gundajali merge with the river Ozat, together forming the main water source of the area. Several small rivers, joining these two main rivers, are used by lions as passage ways.

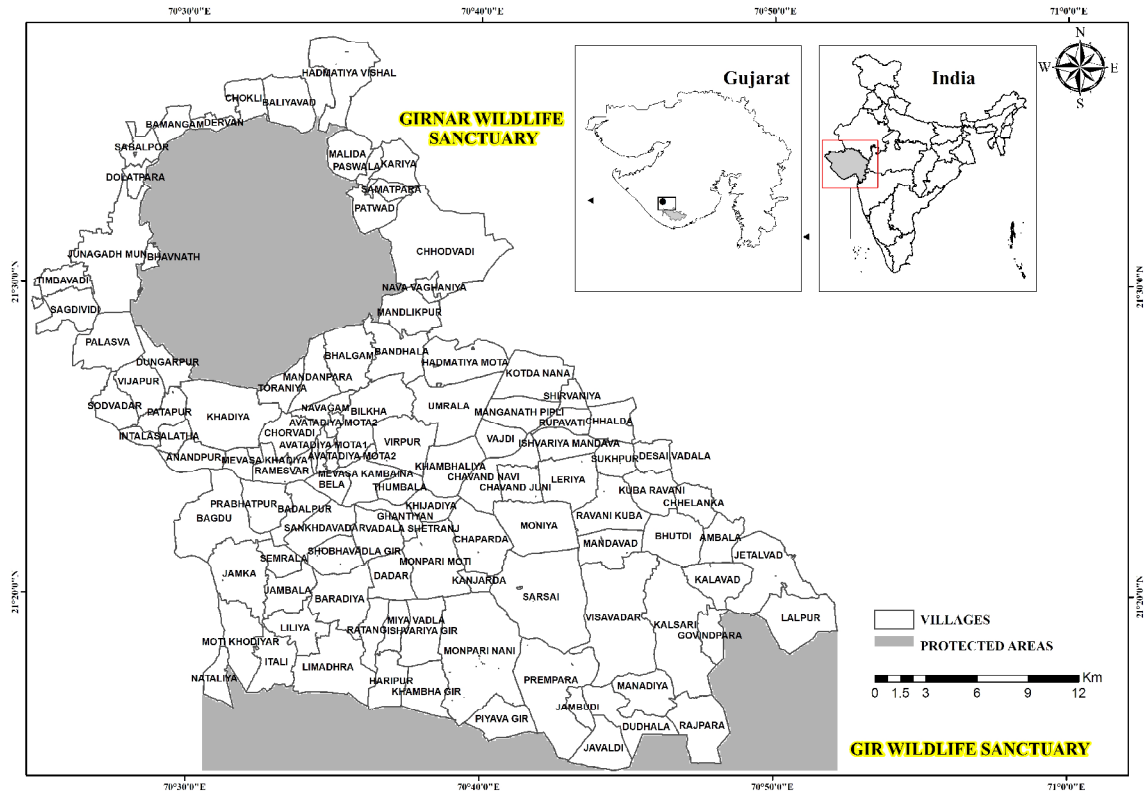


Figure 1: Location of Girnar WLS, Gir WLS and villages within the corridor landscape between Girnar and Gir WLS

The area is interwoven with road and rail networks. There are four major state highways and three metre gauge railways going through our study area. The main crops cultivated in this area are bajri (*Pennisetum glaucum*), jowar (*Sorghum bicolor*), wheat (*Triticum spp*), paddy (*Oryza sativa*), sugar-cane (*Saccharum officinarum*), ground nuts (*Arachis hypogaea*), cotton (*Gossypium spp*), sesame (*Sesamum indicum*), onions (*Allium cepa*), chillies (*Capsicum frutescens*) etc. Ground nut, cotton, jowar and wheat considered as the major crops of the district.

Methodology:

Image Classification & Calculation of Spatial Metrics:

Landsat5 TM imagery of the year 1998, 2002 and 2009 of resolution of 30 m X 30 m (Jensen, 1996) were used for the land use land cover classification. A field survey was carried out in November-January 2010 to assess the variation in land use/cover in this landscape. Fifty vegetation plots were sampled of 10 meter and 5 meter radius to collect phytosociological data of plants. Other descriptive parameters like: topography, precipitation, hydrology, vegetation structure, and human land use, cultivable lands, presence of roads etc. were also used to differentiate habitats within the habitat matrix. Unsupervised classification in ERDAS 9.2 was used to create land cover map describing four major land use classes present in this landscape such as cropland, sparse vegetation,

waterbodies, and barren land including bare areas, road networks, and settlements (Figure 4).

The changes in areas under sparse vegetation & urban structures were measured and analyzed using the FRAGSTATS (McGarigal et al. 2002) and the classified map. Five spatial metrics (Class area, Number of patches, Edge density, Largest patch index, Euclidian mean nearest neighbor distance, Area) that showed significant difference were used for analyzing transformations. These indices are described in table 1.

Table 1: Spatial metrics used in change detection analysis of the corridor landscape

Metrics	Description	Units	Range
Class Area (CA)	CA measures total areas under different land use classes in the landscape.	Hectares	CA>0, no limit
Number of Patches (NP)	NP is the number of patches of all land use classes in the landscape	None	NP=0, no limit
Edge Density (ED)	ED equals the sum of the lengths (m) of all edge segments involving the patch type, divided by the total landscape area (m ²)	Meters per hectare	ED=0, no limit
Largest Patch Index (LPI)	LPI percentage of the landscape comprised by the largest patch	Percent	0<LPI=100
Euclidian Mean Nearest Neighbor Distance (EMN-MN)	Equals the distance (m) to the nearest neighboring patch of the same type, based on the shortest edge-to-edge distance	Meters	EMN_MN>0, no limit

Land use change modelling using Cellular Automata (CA)-Markov

Cellular Automata (CA)-Markov Model Description

For this study, we used Cellular Automata (CA)-Markov Chain analysis modeling technique, embedded in IDRISI Andes software (Eastman 2000). This incorporates two techniques: Markov Chain analysis and CA. The Markov Chain analysis explains the likelihood of land cover alteration from one period to another by developing a transition probability matrix between t1 and t2. The Markov module is based on the first law of Geography by using a contiguity rule (Cabral & Zamyatin 2006). According to the rule a pixel that is near one specific land cover category is more liable to become that category than a pixel that is farther. CA is integrated into the Markovian approach to add the spatial character to the model. CA-Markov models the change of several classes of cells by using a Markov transition matrix; a suitability map and a neighborhood filter.

In this study, the Markov Chain analysis model was executed using the Markov module. The first step in the model was to develop a transition probability matrix for each of the land cover classes between the years 2002 and 2009, and this in turn was used as

an input for modeling land cover change. CA analysis was carried out with the CA-Markov module, which uses the output from the above stated Markov Chain analysis and transition suitability image collection, and applies a contiguity filter.

Implementing and validating the model

The predictive power of the model was evaluated by comparing the result of the simulation t2 (2009) to a real classified map of t2 (2009) by means of using three different Kappa statistics (Pontus 2000, Dushku et al 2003): Kappa for no information (K_{no}), Kappa for location ($K_{location}$), and Kappa for quantity ($K_{quantity}$). K_{no} describes the overall accuracy of the simulated map. $K_{location}$ validate the model's ability to predict location. $K_{quantity}$ is a measure of validation of the simulations to predict quantity perfectly. If the predictive power of a model is considered strong then it is used to make future projections assuming that the transition mechanism verified between t1 and t2 is going to be repeated.

Results & Discussion

Land cover areas of 1998 were dominated by cropland (57.9%) followed by sparse vegetation 23.7%, barren land and settlements 17.4%, while Waterbodies and earth dam occupied 0.98% of the total area. On the other hand, in 2009 though cropland continued to occupy the largest area of 57.5% there was an increase in barren land and settlements to 33.1%, followed by sparse vegetation (8.79%) and the lowest cover was of the waterbodies and earth dam occupied.

During this period (1998-2009) sparse vegetation decreased in area by 63% while barren land and settlements increased in area by 90%. Furthermore, in year 2002 sparse vegetation decreased in terms of coverage from previous 24% to 20% of the total area but it continued to take the second lead in coverage of the area followed by barren land and settlements (14%) and waterbodies, earth dam occupied 0.7% of the total area. The calculation of amount of percentage change was based on initial values of individual land use classes for year 1998 and 2002. For instance, in 1998-2002 cropland increased 11% with the decrease in area of sparse vegetation (13%) and barren areas and settlements (18%). In 2009 there was no significant change in area under cropland but area under barren land and settlements increased by 134% with a decrease in areas under sparse vegetation (57%) (Figure 2).

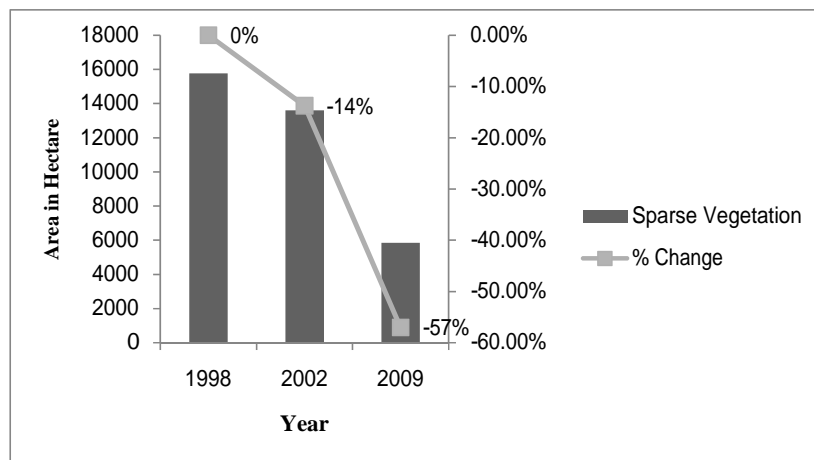


Figure 2: Change in sparse vegetation land cover class between 1998 to 2009 in the corridor landscape between Gir and Girnar WLS

Spatial metrics and their variation were calculated for the barren land and settlement areas (Table 2), which increased by 90% and for areas under sparse vegetation which decreased by 63% in the landscape between 1998 and 2009.

Table 2: Landscape indices & percentage of changes in areas under Barren land-settlement land-use in the corridor landscape between 1998 to 2009

Metrics	Year			Changes in areas under Barren Land-Settlement land use	
	1998	2002	2009	$\Delta\%$ 1998-2002	$\Delta\%$ 2002-2009
CA	11575.34	9415.11	22038.81	-0.19	1.34
NP	9610	6634	8432	-0.31	0.27
LPI	1.08	1.03	8.21	-0.05	6.95
ED	69.01	51.18	71.29	-0.26	0.39
ENN_MN	86.37	91.75	79.16	0.06	-0.14

* See Table 1 for abbreviation

The NP in a landscape analysis indicates the aggregation or disaggregation in the landscape, while LPI measures the proportion of total landscape area comprised by the largest urban patch. The NP (i.e., urban blocks and patches of barren area) decreased considerably (31%) between 1998 and 2002 (First Period), and increased (27%) between 2002 and 2009 (Second Period). This suggests an urbanization made by agglomeration of pre-existing urban patches in the first period while, in the second period, urbanization was characterized by dispersion. The development of a number of isolated fragmented or discontinuous built-up areas occurred in the second period. LPI increased by 7% between 1998 and 2009, thus indicating considerable growth within the historical urban core. ED decreased by 26% in the first period indicating urban sprawl around historical urban core while increased by 39% in the second period thus indicating an increase in the total length of the edge of the urban patches, as due to land use fragmentation. The decrease in ENN_MN by 14% after 2002 reveals a reduction in the distance between the built-up patches, thus suggesting coalescence.

Table 3: Landscape indices & percentage of changes in areas under sparse vegetation in the corridor landscape between 1998 to 2009

Metrics	Year			Changes in areas under Sparse Vegetation	
	1998	2002	2009	$\Delta\%$ 1998-2002	$\Delta\%$ 2002-2009
CA	15765.69	13596.42	5846.24	-0.14	-0.57
NP	10281	8777	14614	-0.15	0.67
LPI	1.62	0.95	0.02	-0.41	-0.98
ED	103.22	80.61	45.20	-0.22	-0.44
ENN_MN	77.44	84.33	83.67	0.09	-0.01

* See Table 1 for abbreviation

Observation of changes occurred in areas under sparse vegetation (Table 3) revealed that the NP (i.e., patches of sparse vegetation) decreased considerably (15%) indicating removal of forest patches in the first period and increased considerably (67%) in the second period showing fragmentation in existing patches. The decrease in class area in first and second period (14% and 57%) assured that there was no addition of new vegetation patches and the increase in number of patches in second phase was surely due to fragmentation of existing patches. The loss of vegetation patches and encroachment on the existing patches were more evident by the decrease in LPI (41%) in the first period and (98%) in the second period. Decrease in ED (22%) in 1998-2002 and (44%) in 2002-2009 indicates decrease in shape diversity of patches and rounding up of the existing patches, clearly showing the loss of forest habitat.

Increase of agriculture, changes in farming practices and urban sprawl impedes wildlife movement from one protected area to another. Saurashtra has been the agricultural bowl of for the past 800 years. Very fertile lands historically farmed but intensity of agriculture has changed. The current land uses are not compatible with wildlife and biodiversity conservation. Land use changes along wildlife corridors caused negative impacts on Gir-Girnar corridors. Study of land use changes and their impacts on wildlife corridor between Gir and Girnar wildlife sanctuary revealed that settlements and agriculture expanded into wildlife grazing and dispersal areas which reduced the area under sparse vegetation patches within the corridor (Figure 3).

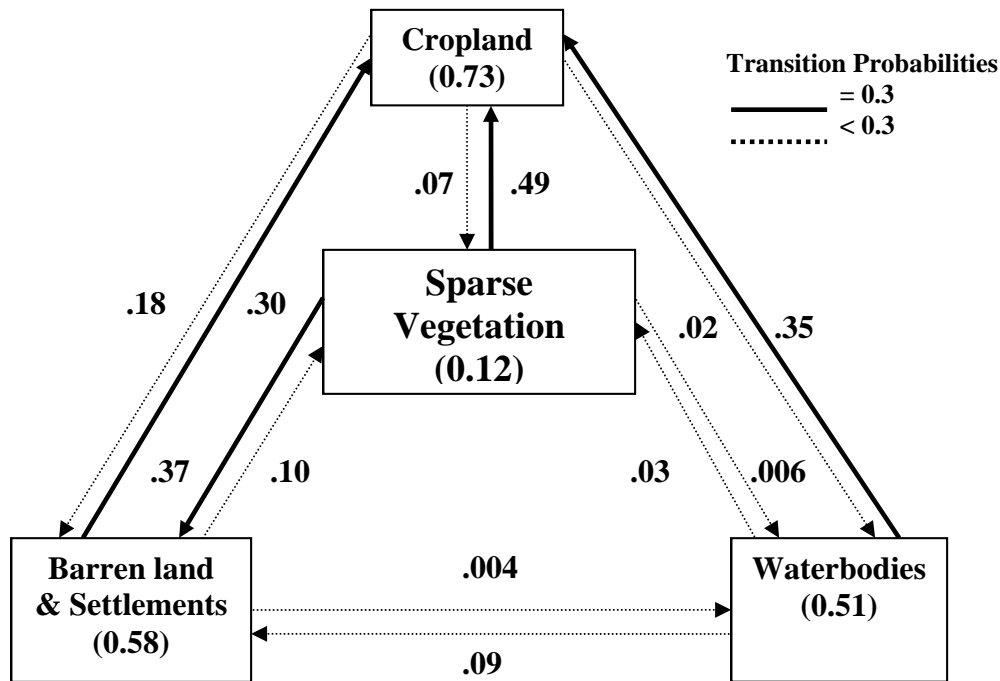


Figure 3: Box and arrow diagram illustrating land cover transition probabilities for the corridor landscape 2002-2009. Values within boxes are self replacement probabilities, whereas values positioned on arrows are transition probabilities

Land Cover Modeling and Validation

Visual interpretation of the modeling results shows that the simulated map for the year 2009 is reasonably similar to the real map for that year. A more detailed analysis was accomplished using the Kappa variations. The closer the values of these indices are to 100%, the stronger the agreement is between two maps. The K_{no} , which also gives the

overall accuracy of simulation, is calculated to be 66%. The model performed well in the ability to specify location correctly ($K_{location} = 64\%$), and also in the ability to specify quantity ($K_{quantity} = 64\%$). Thus the classified map of 2009 was used for the future projection of 2015 (Figure 5).

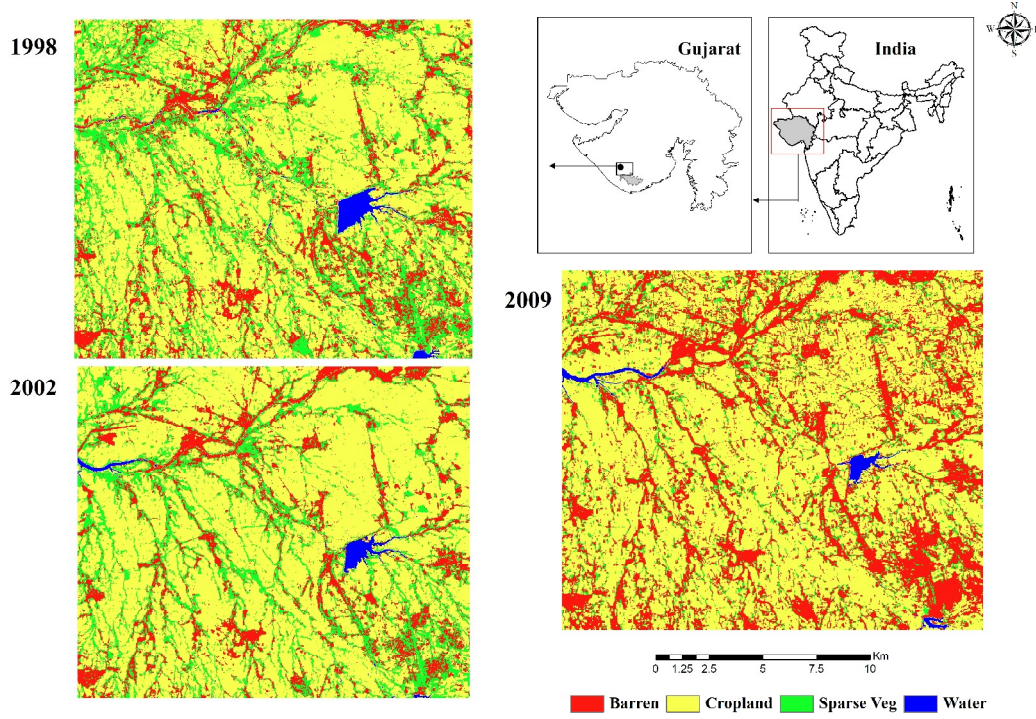


Figure 4: Three classified images (1998, 2002 and 2009) of the landscape

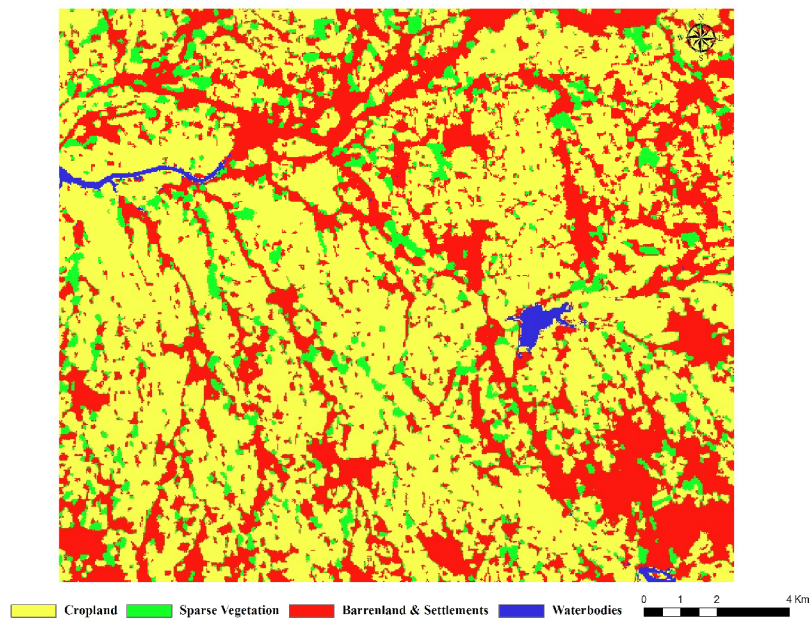


Figure 5: Simulated land cover map of 2015

A cross-tabulation that describes the changes in land cover classes included in the study is given in Table 4. The table shown below demonstrates the number of pixels that are expected to change from 2009 to 2015. The diagonal of the matrix indicates the number of pixels that have persisted during the simulation, while the off-diagonal shows the number pixels that changed class.

Table 4: Cross tabulation of simulated land cover map of 2015 with actual land cover map of 2009 (number of pixels)

		Actual Land Cover Map 2009				
Simulated Land Cover Map 2015	Land Use Classes	Cropland	Sparse Vegetation	Barren land & Settlements	Waterbodies	Total
	Cropland	174620	3783	3165	302	181870
	Sparse Vegetation	5084	17097	4011	0	26192
	Barren land & Settlements	11871	5444	78907	0	96222
	Waterbodies	15	48	169	2084	2316
	Total	191590	26372	86252	2386	307622

Conclusions

Gir protected area is at its carrying capacity for lions (Singh 1995), positive changes in the habitat may improve this figure marginally, but for better management of the increasing lion population, habitat improvement in new areas naturally colonized by the lions and maintaining the linkages in between these areas are necessary. The human dominated landscape surrounding Gir protected area is fragmented. Analyses of land use and cover change processes within the Gir-Girnar corridors suggest that riverine patches are vanishing at an alarming rate and the remaining patches are likely to be converted to agriculture. Sustainable management of these forest fragments huddled along the riparian corridors of tributaries of the main river Ozat and other small rivers is urgent with a focus on biodiversity.

Comparison of the 1998 Landsat TM classified image with the classification derived from the 2009 Landsat TM data shows that forests have been drastically impacted by conversion to agriculture and settlements during the period. Consequently, the area experienced extensive conversion to urban land cover in the last few decades. The continuous augment in land use changes is also reflected in an increased area under settlements. The occurrence of land use changes in areas adjacent to the Gir-Girnar wildlife corridor has not only reduced wildlife habitat but also may result into obstruction of wildlife movement from either part in future if the situation is left unattended. Results of the simulated land cover map of 2015 indicate that urban growth might continue to expand further in the future, and might have an irrefutable impact on land resources, unless some conservation policy is enacted.

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Acknowledgements

Every long-term work could be accomplished only with the help of several people. This study would have never been possible without the constant guidance, support, encouragement from Dr. Y.V. Jhala. His constant effort for perfection, valuable comments and getting the best thing out of mine are appreciated for whatever the good work, I have done so far in my 6 years of research career. Though it seems a bit emotional, but his inspiration, appreciation and encouragement for small but important breakthroughs have always given the strength to be strict on my principle, even going through difficult phases in my life. Not only for this research but he is acknowledged and would be remembered always for his supervision, guidance, and support, which has greatly assisted in my professional development.

I am thankful to Shri. Qamar Qureshi, for his support in many ways from past six years for academic and otherwise, but I could never ever understand the person who is equally good with statistics, literature, science, entertainment and so many other qualities. Sir, I wish I would really be intelligent enough to understand all your ideas and could incorporate all in my research. I apologize for much confusion we had during our discussions.

I sincerely thank Mr. P. R. Sinha, Director of Wildlife Institute of India, Dr. V.B. Mathur and Dr. P.K. Mathur, Dean, Dr. K Sankar, Research Coordinator, WII for their support in completion of the research work. I thank Dr. V.P Uniyal, Dr. Ruchi Badola, Dr. S. Sathyakumar, Dr. Asha Rajvanshi, Dr. B.S. Adhikari, Dr. S. Chowdhury, Dr. Dhananjai Mohan, Smt. Bitapi Sinha, Dr. G.H. Talukdar, Dr. Karthikeyan Vasudevan, and Dr. Parag Nigam. for their continuous encouragement, advice, suggestions during the entire period of my stay in WII. I thank Dr. S.A Hussain for providing my staying in WII hostel. Dr. K. Ramesh is acknowledged for always being helpful with suggestions on study design, progress of field work to completion of the thesis writing. Dr. B. K. Mishra and Dr. S. P. Goyal are thanked being always caring and concerned of my study progress.

Partial funding for this project was obtained from the Gujarat Lion Conservation Society, Gujarat Forest Department. I am grateful to Shri P. Khanna IFS, Principal Conservator of Forests, Shri Ravi Asari, IFS, Chief Wildlife Warden (retd.) and Shri S.K. Goyal, IFS, Chief Wildlife Warden for their support. Shri Amit Kumar, IFS, the

then Deputy Conservator of Forests (Gir West) and subsequently, the inputs and assistance provided by Shri Mayank Sharma, IFS, Shri S.K. Chaturvedi, IFS and Shri R.L. Meena, IFS, Chief Conservator of Forests (Wildlife) Junagadh during their respective tenures are acknowledged. I acknowledge the assistance provided by Shri A.M. Parmar, IAS, ex-District Collector, Junagadh; Shri A.B. Khunti, Technical Director at the Collector Office and the Mamlatdars of Mendarda and Visavadar Talukas for providing important land and revenue records. I acknowledge the personal involvement and enthusiasm of Smt. Anita Karn, IFS, Deputy Conservator of Forests (Junagadh), without the support from her, field work in Girnar could not have been possible. Shri V.J. Rana, Director, Sakkarbaug Zoo, Shri P.S. Babaria, Assistant Conservator of Forests (Junagadh), Drs. C.S. Bhuva and R. Kadivar; veterinaries at the Sakkarbaug zoo are acknowledged for their timely assistance. I acknowledge the personal involvement and logistic support provided by Dr. S. Kumar, IFS, Deputy Conservator of Forests (Sasan) to the research team during the field work in Gir and Girnar. Assistance and logistic support provided by Shri R. Katara, Deputy Conservator of Forests (Gir West), Shri S.K. Jadeja (ex-RFOGirnar), Shri Gojia (RFO-Girnar), Shri Pandya (ex-RFO Girnar) and their staff is greatly appreciated.

I express my gratitude to “Nilanjana” (Nilanjana Roy), my teacher, loving elder sister, who taught me GIS and remote sensing in such an enjoyable way, so this subject was never fear to me as to others and I could enjoy experimenting and that’s the way I learnt. Spending time with her, the whole day in GIS lab are few of those golden moments in life, I will cherish throughout my life.

I pay my respect to those “unknown heroes” (my field assistants Taju Aata, Bikhu bhai, Ismail Bhai, Hammal bhai, Bhola bhai, Bhupat bhai, Rasul bhai and trackers of Gir protected area) whose fearlessness, trustworthiness, protection and last but not the least “knowledge” of the area and animals which was not less than any scientist studying them for years. I salute them, and without their dedication and effort; this study could not have been possible. I express my gratitude to Yogendra Shah and family, Bhusan Pandya and family, Manisha Rajput and family, and my fellow researchers, Kausik, Shamshad for making my stay in Gujarat memorable.

I am also greatly indebted to all the people of Saurashtra. While doing my field work in vast agro-pastoral landscape, I came across many things which were unknown before, “the taste of tea”, “village”, “pastoralism”, “tribal life”, the well conserved

“caste system” etc. In a nutshell, that landscape was very unique to me with its people. Not only for positive attitude towards wildlife conservation, but they will be remembered for their uniqueness. I specially thank my landlord and people from Sasan, Krankach, Dholikui, Satadhar, Dhari, Junagadh, Girnar etc. When I think of the hospitality of Saurashtra, I don't get words to express their each and every support, strength of humanity. Big thanks to you all and would love to visit the landscape again and again.

Back in WII, the natural environment of the campus rejuvenates and friendliness amongst scholars, M.Sc. students, and interns leaves you in a comfort zone where you enjoy your work in encouragement, support, and appreciation from your friends. I was lucky to have seniors like Upamanyu, Jimmy, Niladri, Santanu, Kausik, Tapajit, Gajendra, Vinay, Bopanna, Bibek Close buddies like Sabuj, Sutirtha, Krishnendu, Subhadeep, Sheena, Abesh, Angshuman, Sudip, Sitendu, Indranil, Ninad Raut, Merwyn, Samina, Shazia, Monica, Chongpi, Ridhhika, Iswari, Kunzes, Pragateesh, Selvan, Daya, Madhura, Pranita, Ninad, Preeti, Upma, Monideepa, Shruti, Sarbani, Pritha, Ankita Sinha, Ankita Bhattacharya, Amit, Tawqir, Tanushree, Samhita, Neha, Neeraj, Manjari Malviya, Madhumita, Megha, Malem, Ayan, Geetanjali, Nilofer, Kolloli, Biba, Bipin and many more interns, researchers from foreign universities I met over the time. Over the years sharing each and every moment with you all, gives a feeling of bonding, family and togetherness. Thank you all for making this phase of life memorable.

Mrs. Nalini Choudhury and Sri. B. C. Choudhury deserve to be mentioned specially not only in this thesis but in my life. Their affections, bonding, homemade food, asking about my whereabouts almost in every alternative days provided me the much needed strength to fight against all odds, giving a feeling of home away from home.

Anupya needs a special mention here, for her selfless, endless support whenever it was needed. She made my jobless, homeless life, interesting with her colorful lifestyle, loud music, lovely food and COFFEE and the most needed moral support which was unexpected from her and it was the most expensive gift of the moment.

I acknowledge the assistance and support provided by Mr. Gyanesh Chibber, Mr. Vinay Sharma for helping me with the official work, Mr. P. L. Saklani for plant identification and their botanical names, Mr. A. K. Dubey and Mr. Kharak Singh from

account section. Mr. Verma, Kishanji, Umedji, Shashi Madam, Uniyalji and Mahesh da from Library, Sambhu and Devendra bhaiya.

I thank Sh. Rajesh Thapa, system manager, WII, for permitting my access to GIS lab, I thank Dr. Pannalal, Smt. Alka Agarwal, Sh. Lekh Nath Sharma, Sukumar Sir, Sh. Muthu Veerappan, from IT and GIS cell for always being cordial and being supportive in successful execution of the project. I thank Swati from Science for her friendship and jointly playing the role with me of trouble shooter in GIS. Mr. Virendra Sharma is acknowledged for the last moment formatting of this thesis. I thank Babita di for her sisterly bonding during my initial days in WII.

From any small to bigger problems in GIS lab, I would have troubled Dr. Manoj Agarwal and he patiently listened to all which got solved soon, which would have taken long time to do on my own way. Interaction with saumya and Mrs. Agarwal was always cordial and homely. Thanks for gifting some wonderful moments.

The journey to WII and so far was not abrupt and could have been possible only with the support from many people, who had faith on my ability and that could have helped me in formation of what I am today. I sincerely thank Mr. S.K. Chakraborty, former PCCF, Karnataka, Dr. Animesh Bal, Former Additional Director, ZSI, Kolkata, Dr. Kamal Banerjee, Reader, Presidency College, and last but the most important person Late. Dr. Subrata Sinha the former Deputy Director General of the Geological Survey of India and family, I wish, you could have seen your “Jungle” acquiring the next educational degree, I will miss you in every step ahead.

Kausik, Anupya, Sabuj have read various parts of this thesis or provided constructive criticism in other ways. I would like to thank all of them who have been involved with the review of chapters in this thesis.

Facebook, I might be the first person acknowledging you. Well, you were such a great stress buster for last few months of thesis writing, when I actually confined myself to the four walls of my room, you were the only vent out, the only medium to be connected with my friends and the outer world. Thanks!

Last but not least, I would like to thank my family and parents for supporting me throughout my life, to reach many of these thanks giving moments especially at some of the most turbulent times. I deeply appreciate their love, faith and encouragement which is difficult to put in words.

Executive Summary

Due to timely protection by the Junagadh Nawabs and subsequently management by the Gujarat state forest department, the numbers of the Asiatic lions (*Panthera leo persica*) have increased from below 50 individuals to more than 400 and their range increased from few hundred square kilometer of the Gir Protected Area (PA) to over 9,000 km² of the Gir landscape. The Gir landscape is comprised of the Gir PA, Girnar sanctuary, coastal forests and human dominated areas of Amreli, Bhavnagar and Junagadh districts of Saurashtra peninsula, Gujarat. Lions in this landscape survive in a meta-population framework with the Gir PA being the 'source'. Conservation of 'source' alone may not be a sufficient strategy to ensure long term persistence of lions; rather it is enhanced by maintaining connectivity among different populations so as to permit lion movement across human-dominated areas. However, the traditional land-use patterns within Gir landscape is changing at an alarming rate mainly as a result of agricultural expansion, urbanization and industrialization. In this scenario, I attempt to (1) Examine the distribution of suitable lion habitat in Saurashtra, especially in areas outside of protected area to assess the potential of the landscape for supporting an increasing lion population, (2) Identify functional corridors joining Gir and Girnar protected area with suggested management interventions, necessary for safeguarding the corridor, (3) Evaluate the current landscape architecture of Gir PA and its surroundings (4) Assess the rate of change in land use and vegetation patterns within Gir PA, (5) Develop habitat suitability models at different spatial scales for lions and its principal prey species.

To address the above objectives, I conducted field research between November 2008 – January 2011, covering (i) the entire Greater Gir landscape of Junagadh, Amreli, Bhavnagar districts, where lions have already spread, and prevalent land use/cover pattern in this landscape includes very dense forest, moderate dense forest, open forest, nonforest, scrub and waterbodies interspersed with human land uses *viz.* settlements, cities, townships, industry, mines, orchards and agriculture (ii) the human dominated landscape between Gir and Girnar wildlife sanctuaries, covering more than 100 villages, consists of broken topography, small drainage systems, Government and privately owned barren lands, fallow lands and is predominantly agricultural with

seasonal crops and mango orchards. This area is exhibiting rapid growth, still there are many natural areas interspersed among human development, (iii) the Gir PA (1412 km²) (about 258 km² considered as the national park and 1153 km² for the Sanctuary), the Girnar wildlife sanctuary (180 km²). The entire Gir PA was divided into 2 km X 2 km grids and alternative grids were sampled. In each of these sampling grids, one 2.5 km transect was laid diagonally covering most of the area of the grid. In each transect, at 250 meter intervals, plots (n > 900) were laid of 10 meter and 5 meter radius to collect data on vegetation and ungulate pellets to estimate animal abundance. The Field work was followed by lab analysis of LANDSAT TM satellite imageries (1998, 2002 and 2009) and few vector layers digitized, to generate different layers required to fulfill study objectives.

Evaluating the potential for Asiatic lion (*Panthera leo persica*) range expansion within Saurashtra, Gujarat.

I identified priority conservation areas in Gujarat using habitat suitability models to identify sites with the most suitable habitat for lions. The relative contribution of certain environmental, anthropogenic and topographic variables to the geographic distribution of Asiatic lions was analyzed using Environmental Niche Factor Analysis (ENFA). Location of 19 radio collared lions across the Gir landscape along with lion sightings and lion scat locations (n > 10,000) were used as inputs in those models. All variables were extracted to 1kmX1km grids and attributed to respective GIS raster coverages to be used for habitat suitability analysis. The output of the ENFA modeling (Figure 1) indicated four geographically distinct prime lion habitat clusters in Gujarat. 1. One cluster is located at the coastal belt formed with Gir as a source patch with limited, fragile connectivity with other sink patches viz. Girnar, coastal patches at Kodinar, Rajula and *Prosopis* patches at the bank of Shetrunji river. 2. Second cluster covers Barda and adjoining landscape. 3. Third cluster is located at the north eastern side of the state formed of protected areas viz. Jessor wildlife sanctuary (WLS), Balaram-Ambaji WLS. 4. The Fourth cluster covers Jambugoda WLS, Ratanmahal WLS and the adjoining landscape. There are small scrub patches distributed in this landscape, sustainability rate of which is very low and these patches need to be prioritized for conservation investments, since they act as stepping stones,

providing day *refugia*. These small patches are mainly village range land or *Prosopis* patches characteristic of this semi-arid landscape.

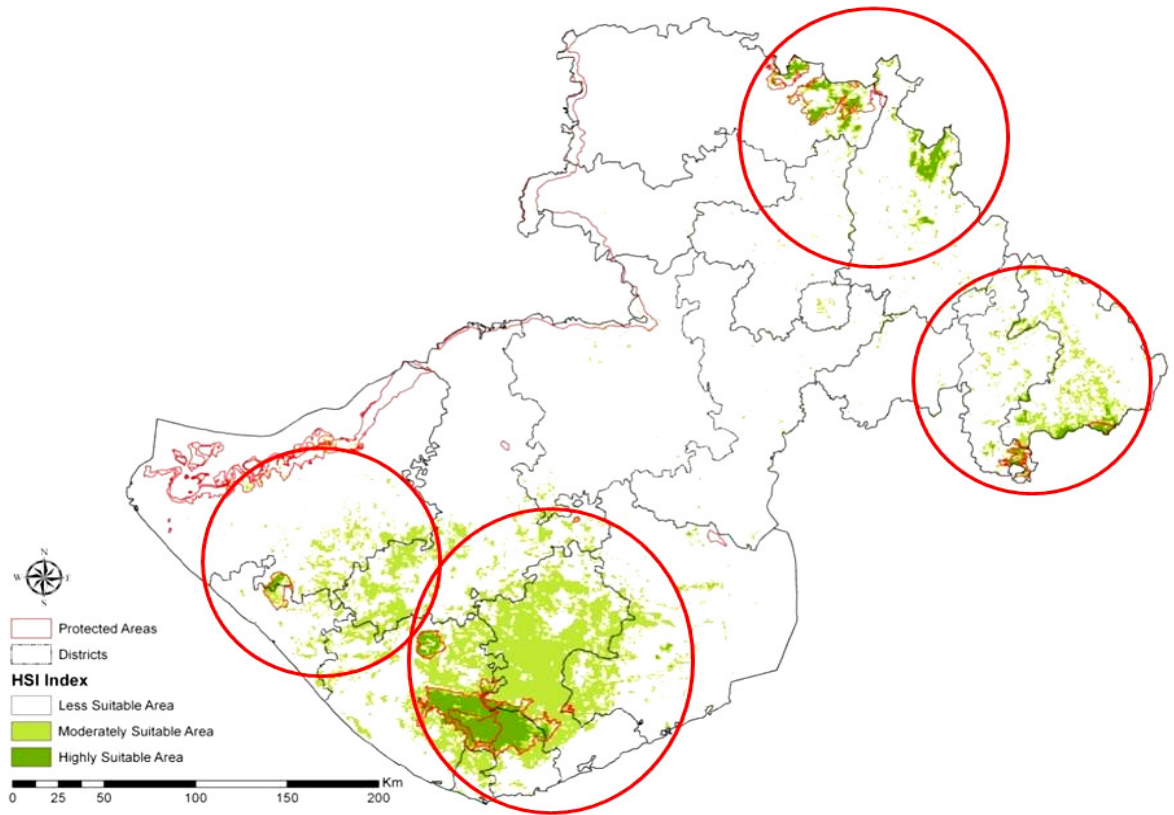


Figure 1: Areas identified having potential to support range expansion of Asiatic lions (*Panthera leo persica*) in Gujarat.

Managing the agro-pastoral landscape for Asiatic Lions (*Panthera leo persica*): Habitat suitability modeling and corridor identification between Gir and Girnar wildlife sanctuary

Policy and management that result in effective conservation of habitat permeability between source populations across human dominated landscapes are of paramount importance to wide-ranging carnivores. The Gir protected area and Girnar wildlife sanctuary are the two major protected areas separated by a mosaic of human land use matrix. To identify specific habitat characteristics associated with lion use in this matrix, I used radio-telemetry locations ($n > 2800$) to model habitat suitability using presence only models in Biomapper. Lion's use of the human dominated matrix was primarily determined by presence of rugged drainage channels, avoidance of higher

elevated areas and distance to major road networks. PATHMATRIX, a tool based on least cost path algorithm, was used in identifying the optimal pathways used by lions for movement through the human dominated landscape connecting Gir PA with Girnar WLS. A least cost path of 36.8 km with a short arm of 11.83 km was identified, which was buffered on either side by 1.5 km to delineate a corridor of 200.25 km² (Figure 2). Major obstacles in the corridor in the form of state and national highways were identified for targeted mitigation measures. On the basis of this research State Govt. of Gujarat declared the corridor area as ecosensitive under the Environment (Protection) Act 1986.

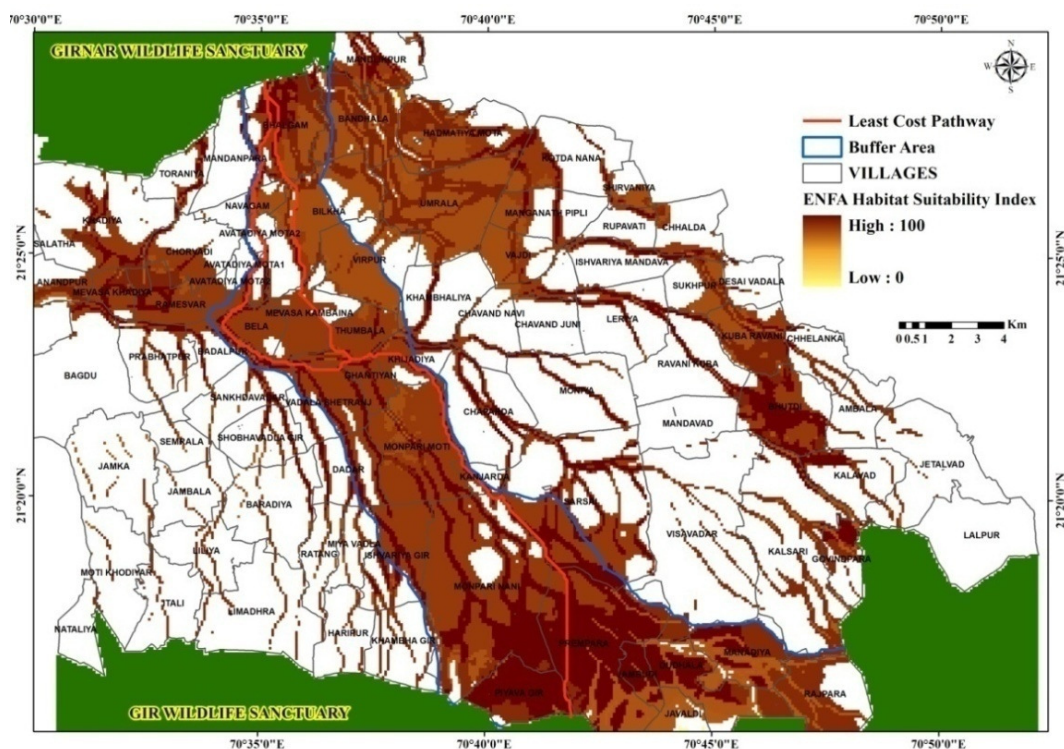


Figure 2: Lions habitat suitability, least cost pathway with buffer around showing the functional corridor in this landscape

Landscape architecture of Gir Protected Area and its surroundings, impact of spatial scale in understanding landscape configuration

The quality of a habitat is directly related to the rates of survival and reproduction of the individuals that live there, to the vitality of their offspring, and to the length of time the site remains suitable for occupancy. The current landscape architecture of Gir

PA and its surroundings (5 km buffer around Gir Protected Area) was assessed to map the land use/cover classes of different vegetation types, water bodies, barren areas and to investigate how commonly used landscape metrics respond to changing grain size (spatial resolution). A total of ten communities were identified from TWINSpan analysis which were incorporated with remote sensing approach, and ten different land use/cover classes (Figure 3) were finalized viz. (1) Moist Teak Forests. (2) *Wrightia*-Teak-*Acacia nilotica* Forests (3) Teak-*Wrightia* Forests (4) *Acacia-Boswellia* Forests (5) *Anogeissus-Boswellia-Acacia* Forests (6) Bare land representing soil that has almost no vegetation. (7) Water, where this class mainly represents water stored behind dams. (8) Agriculture land including crop as well as rainfed and irrigated land. (9) Grassland. (10) Fallow land, in producing land use/cover map of Gir PA.

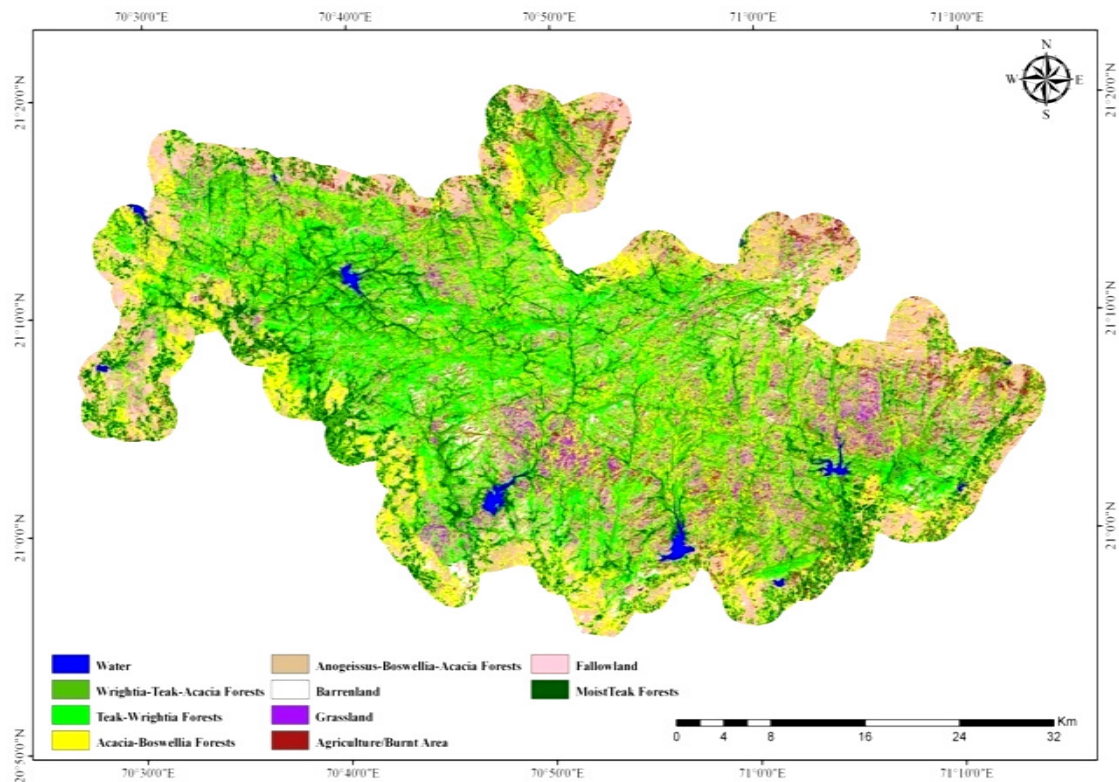


Figure 3: Land use/cover map of Gir protected area based on Landsat TM image of 2009

With the change in grain size through spatial aggregation, the responses of the 17 class-level metrics fell into two general groups: metrics showing consistent scaling relations (Type I) and metrics showing unpredictable scaling behavior (Type II). The first group was further divided into those showing both consistent and robust scaling

relations (Type IA) and those showing consistent but less robust scaling relations (Type IB).

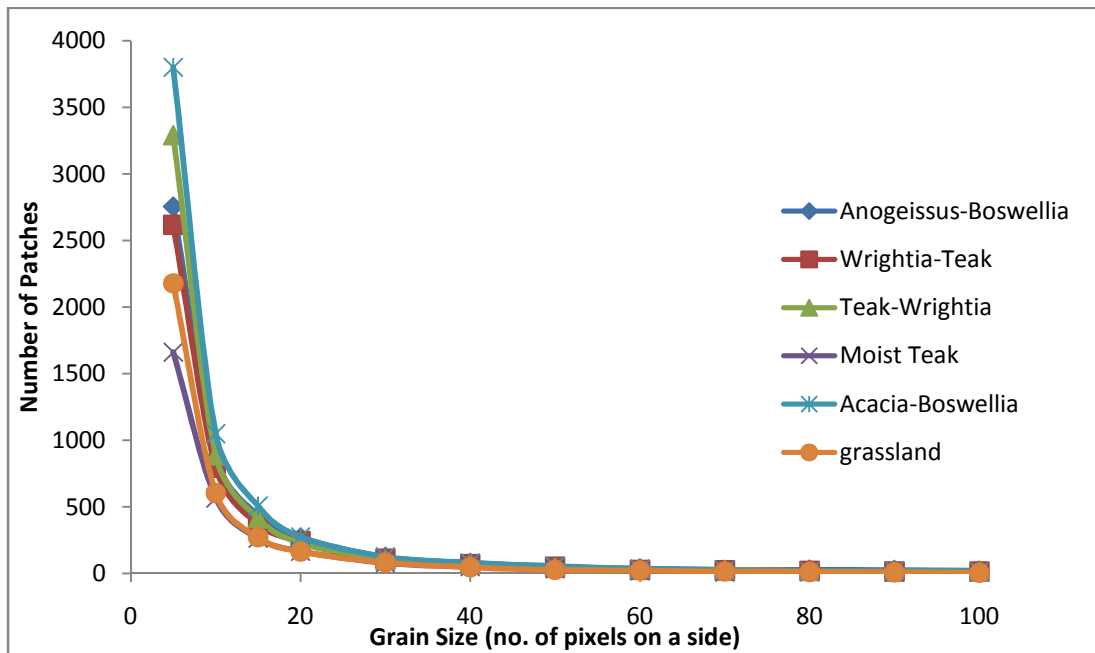


Figure 4: Scalograms showing effects of changing grain size on landscape metrics (Type IA scaling behavior), Relation of number of patches with the change in grain size

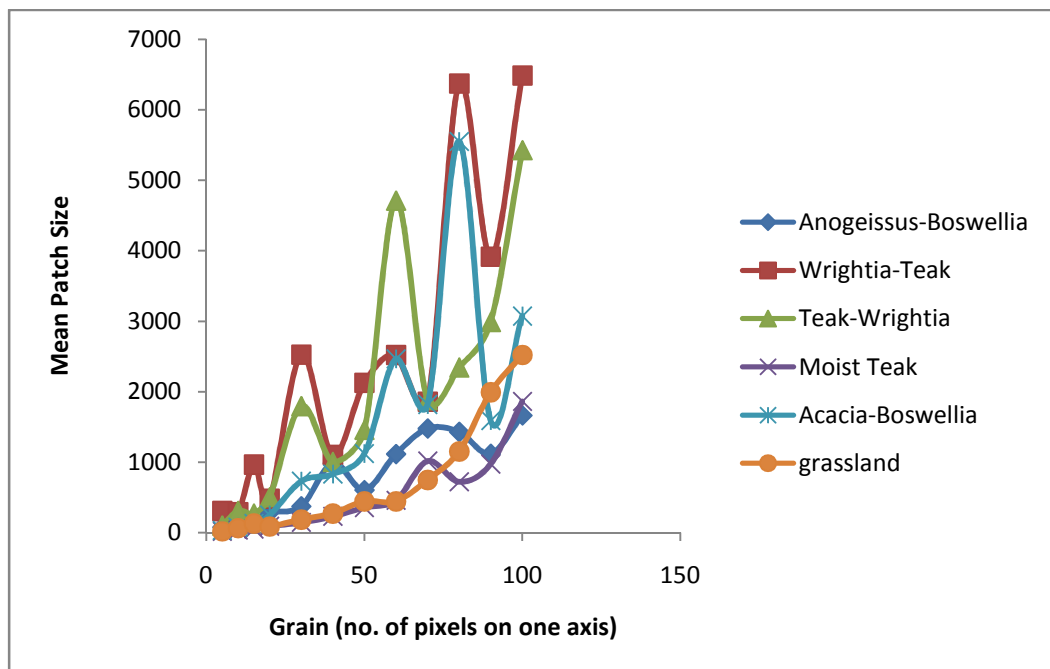


Figure 5: Scalograms showing effects of changing grain size on landscape metrics (Type IB scaling behavior), Relation of mean patch size with the change in grain size

Of the 17 metrics, 5 belonged to Type IA: Number of Patches (NP) (Figure 4), Patch Density (PD), Total Edge (TE), Edge Density (ED), and Landscape Shape Index (LSI); and 7 to Type IB: Largest Patch Index (LPI), Square Pixel Index (SqP), Mean Patch Size (MPS) (Figure 5), Patch Size Standard Deviation (PSSD), Patch Size Coefficient of Variation (PSCV), Area-Weighted Mean Shape Index (AWMSI), and Area-Weighted Mean Fractal Dimension (AWMFD). Type II metrics included: Class Area (CA), Percent of Landscape (CA %), Mean Patch Shape Index (MSI), Mean Patch Fractal Dimension (MPFD), and Double-Log Fractal Dimension (DLFD). This study corroborates the gradually more accepted notions: there is no single “correct” or “optimal” scale for characterizing spatial heterogeneity, and comparison between landscapes using pattern indices must be based on the same spatial resolution and extent. The results demonstrated that the spatial scale at which these patterns are quantified influences the result and measurements made at different scales may not be comparable. The identification of properties that do not change or change predictably across scales would simplify the extrapolation of measurements from fine scales to broad scales. However, although it may be possible to identify simple relationships between landscape parameters measured at different scales, the exact relationship varies across landscapes and does not permit extrapolation from one region to another. Information is lost in coarser grained spatial data. In general, information about the less frequent land cover types is most easily lost, but the rate of loss depends upon their spatial arrangement.

Variations in the Landscape Patterns and Vegetation Cover between 1998 and 2009 in the Semi arid Gir Wildlife Sanctuary

Increasing human activity across the globe has rapidly accelerated the pace of land cover and land use change, resulting in widespread changes in the spatial structure of native habitats and biodiversity loss. I studied the rate of change in land use and vegetation patterns within Gir PA over a period of 10 years. Viewed as a time series, the quality and quantity of different forest types have substantially varied during the time period from 1998 to 2009 (Figure 6, 7, 8). Teak-*Wrightia* forests increased four times in 2002 from their initial coverage of area in 1998, and did not expand much in 2009. The increase in the area resulted from the transition of *Wrightia*-Teak forests into both Teak-*Wrightia* and moist teak forests.

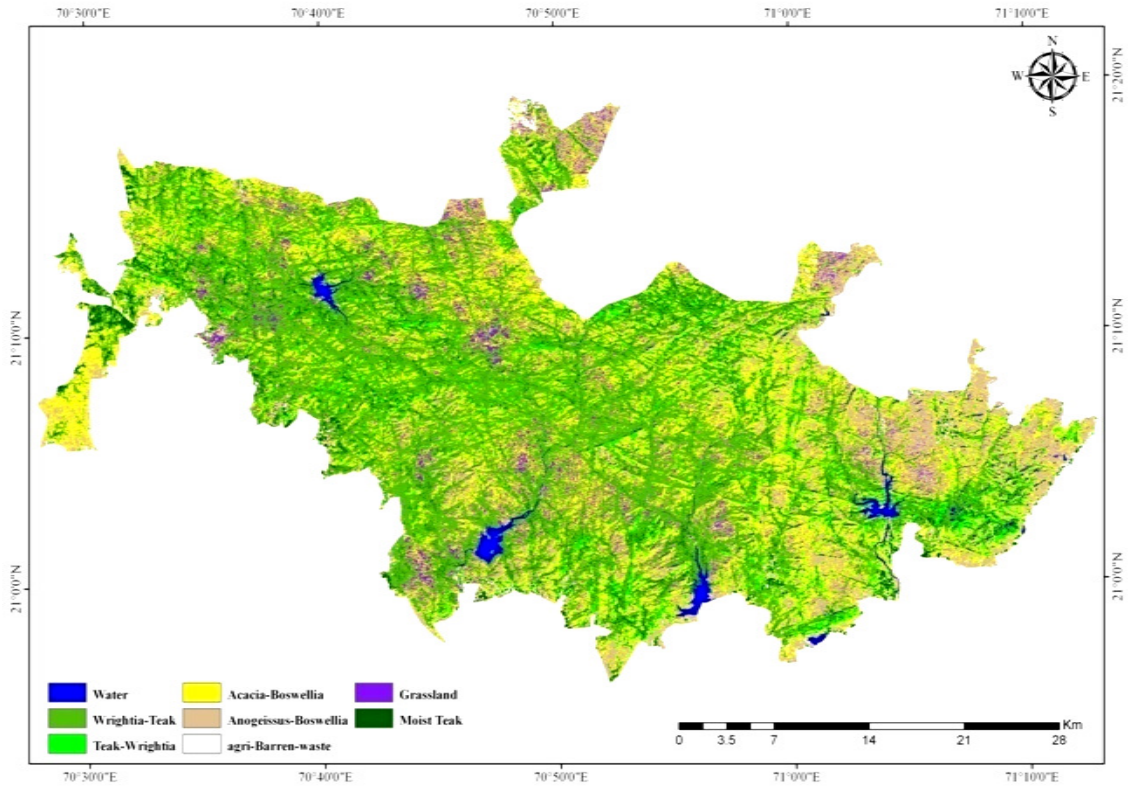


Figure 6: Land use/cover map of Gir PA in1998

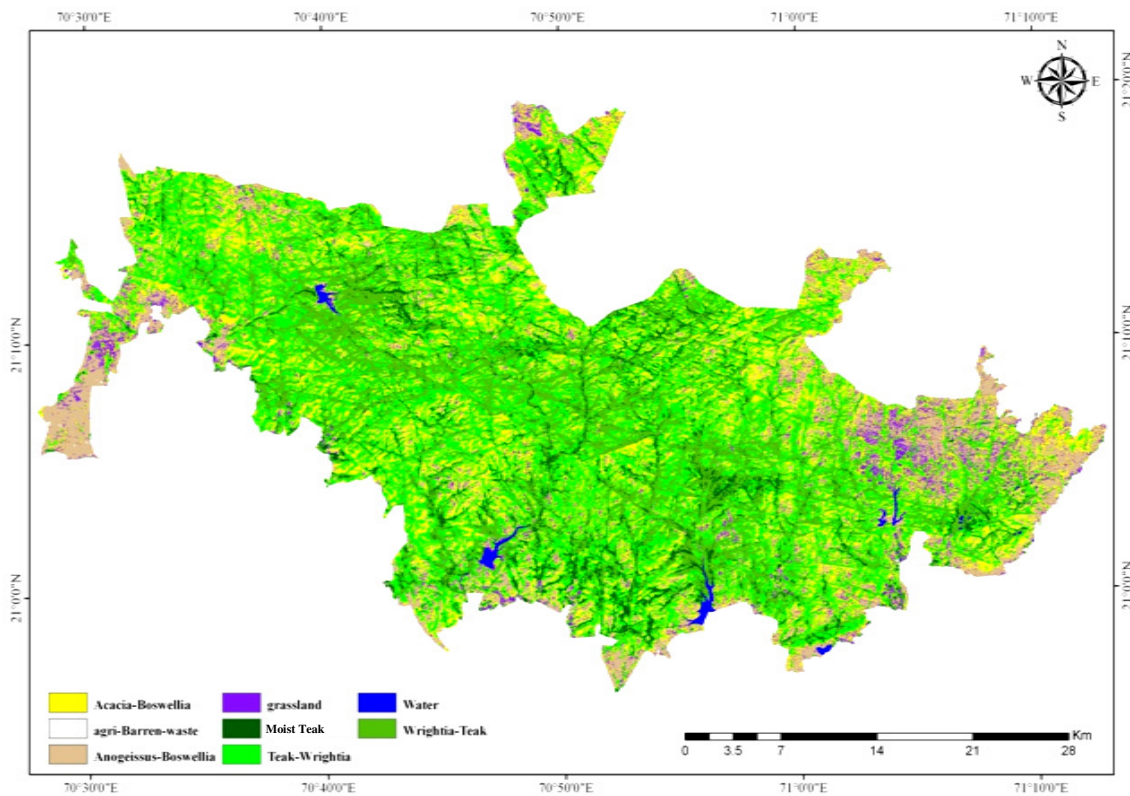


Figure 7: Land use/cover map of Gir PA in 2002

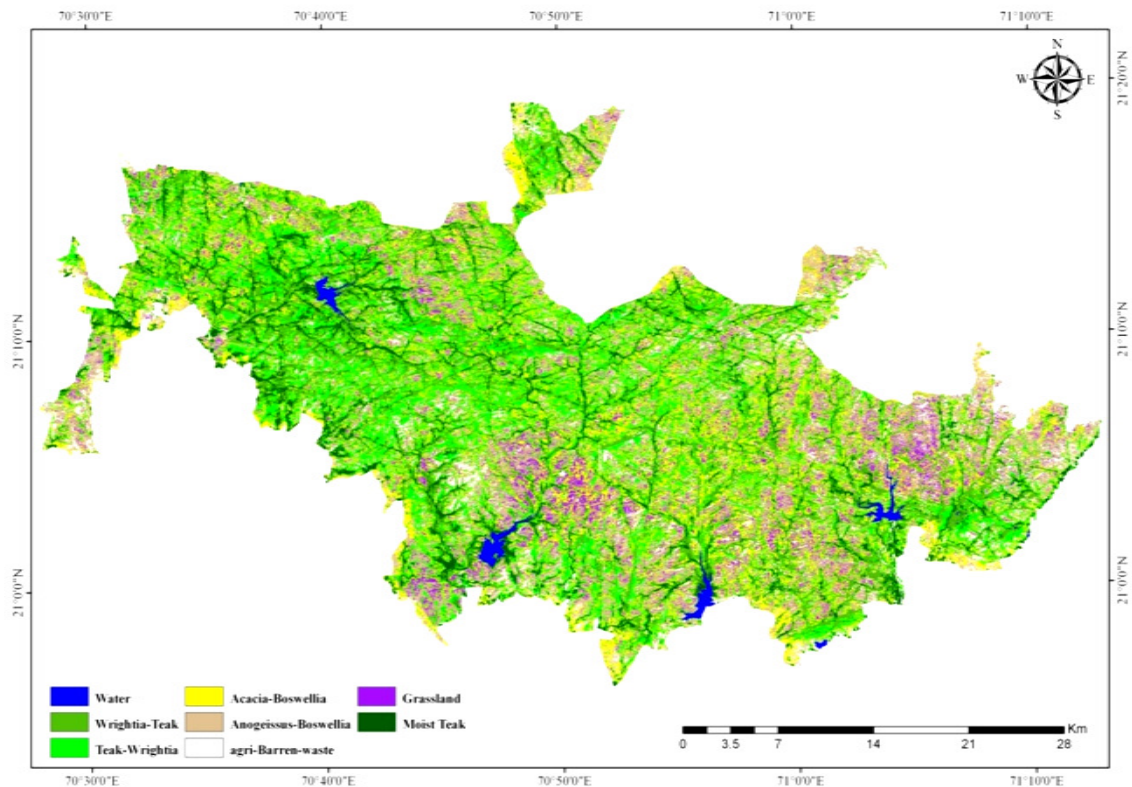


Figure 8: Land use/cover map of Gir PA in 2009

The direction of this change was mainly in central and western part of the study area where teak dominates rather than the eastern part of the sanctuary. The trend in land use/cover changes showed thickening of teak dominated forests type till 2002, after which it was stabilized. The trend of thickening might be considered as a response to the canopy opening and uprooting of million trees after 1982 cyclone following succession of the respective vegetation communities. Other notable land cover changes included the increase in the areas under *Anogeissus-Boswellia-Acacia* forests type. The gradual changes in vegetation pattern in eastern Gir, where *Anogeissus-Boswellia-Acacia* forests increased and *Acacia-Boswellia* forests decreased over time were also visible from the magnitude of changes image obtained from change vector analysis. Number of patches increased for both the forest types during the study period but the mean patch size decreased for *Acacia-Boswellia* whereas it almost remained same for *Anogeissus-Boswellia* Forest type. Average patch size increased for the moist teak forest type as small patches begin to coalesce into larger ones and decreased for the *Wrightia-Teak* forest types as the area under this forest type reduced following succession to teak dominated forest types. Visual interpretation of the land cover modeling results showed that the simulated map for the year 2009 was

reasonably similar to the classified LANDSAT TM image of the same year. A more detailed analysis was accomplished using the Kappa variations. The K_{no} , which also gives the overall accuracy of simulation, was calculated to be 70%. The model performed well in the ability to specify location correctly ($K_{location} = 71\%$), and also in the ability to specify quantity ($K_{quantity} = 71\%$). Thus the classified map of 2009 was used for the future projection of 2020. Future projections presented an acceptable output for short term forecasting.

Multi-scale species distribution modeling for Asiatic lions and its major prey species, understanding the impact of spatial scale in species distribution modeling

Accelerating habitat changes magnified the importance of understanding the role of habitat characteristics in determining species distribution. Species distribution models have used species–environment relations to predict the distribution of species across complex landscapes. However, in the changing environment, response of individual organism to its environment changes over several spatial scales and hierarchical levels; with the smallest scale corresponding to the grain of the organism and the largest scale being at least its home range. In this context, the primary concern of this study was to develop habitat suitability models at different spatial scales for lions and its principal prey species. The Gir PA was divided into grids of 2km X 2km, I estimated ungulate dung density from circular plots of 5 meter radius ($n > 900$) across the Gir PA in a systematic manner. Data from radio collared lions ($n = 13$) was used as presence points of lions. Environmental variables thought to influence utilization of habitat by animals were extracted to 500m X 500m, 1Km X 1Km, 2.5Km X 2.5Km, 5Km X 5Km grids covering the entire study area. The classified vegetation map was used as one of the layers in the model describing habitat for the animals. Distance to the nearest water resources was computed using GIS. Land use/cover data, Climatic variables, Normalized Difference Vegetation Index (NDVI), gross primary production, elevation, human disturbance, and fire occurrence points were obtained from open source GIS archives. Since the study was done at local scale, so all climatic variables were not used in all models since there was not much variation in values except for precipitation and mean temperature. The nilgai population was mostly restricted to eastern sanctuary and sanctuary border. The higher probability of sambar

suitable habitat in the higher rainfall areas implied the impact of rainfall in spatial distribution of sambar. Cheetal presence was best predicted within the forest tracts with widely scattered but assured presence of water sources. Model accuracy decreased with the coarsening of the spatial resolution as it was evident from the decrease in AUC value across the scales. At the home range scale of ungulates, presence of dense forest patches contributed more to the presence of cheetal and sambar, while in case of nilgai it contributed less. Precipitation appeared to be the next important variable at this scale in determining sambar distribution, whereas euclidean distance to artificial water sources and presence of open forest patches appeared to be second best variable in determining cheetal and nilgai distribution. At the local or foraging scale the contribution of presence of dense forest patches decreased compared to other scales in case of sambar spatial distribution, whereas the contribution of the same variable increased in this particular scale based on the mixed feeding and browsing nature of cheetal and nilgai. Euclidean distance to water resources continued to be the second best variable in determining cheetal distribution emphasizing regular water intake nature of the species. The probability of sambar presence increased with the increase in euclidean distance to nesses whereas the probability of nilgai presence decreased with the increase of euclidean distance to nesses. Lions habitat selection at coarse scale was found to be determined by the presence of specific habitat patches and the presence of water sources. At the fine scale, lions preferred areas having better drainage, with dense forest patches providing cover and preferred habitat as day refuges. Lions habitat selection and movement was found to be influenced by the presence of water points, because in semi arid ecosystem, herbivores are largely influenced by the distribution of surface water and lions have a greater chance of encountering their prey near waterholes (Figure 9). The slight average trend towards model degradation at coarser grain size, can be compared to the results of others. Species distribution model for cheetal and sambar degraded with the coarsening of the grain size whereas for nilgai and lions there was not much change in model accuracy. Therefore, It seems from these results as well as from other published studies that all three effects - improvement, no change, or degradation of models can be obtained when changing grain size. The configuration of the study area and species identity (e.g. type of organism or rarity) in addition might act as important causes of discrepancy between models, possibly yielding differential responses to a change in grain size of similar amplitude.

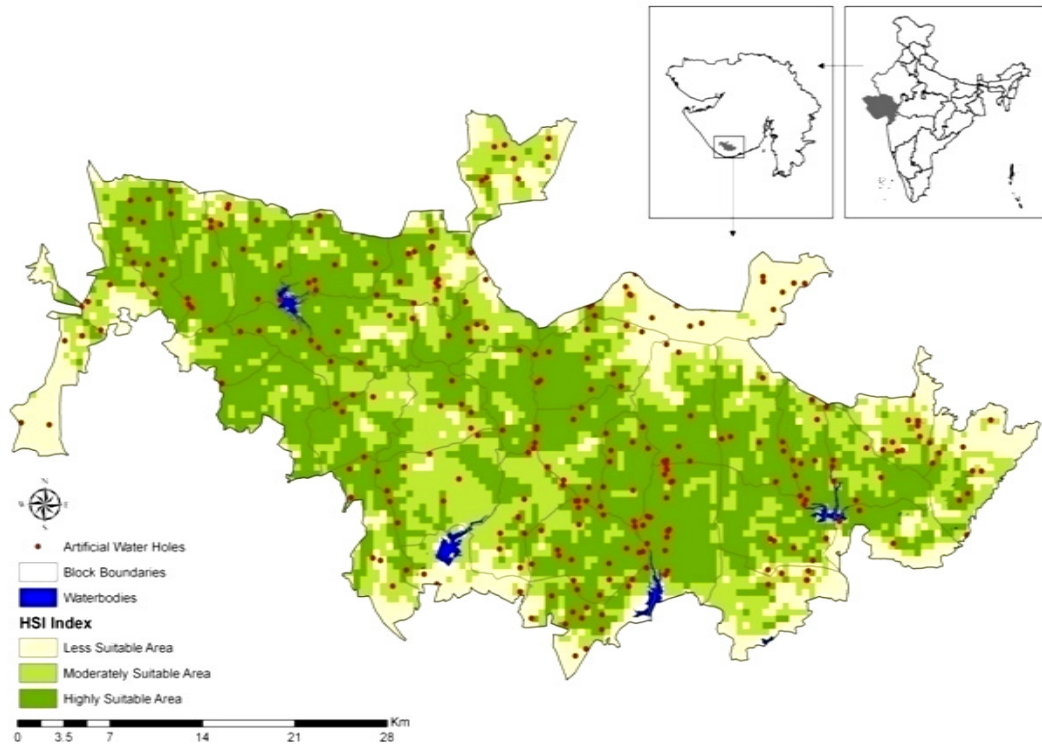


Figure 9: Relationship between lion's habitat selection and waterholes distribution

The strongest relationship with environmental variables was observed at smaller spatial scale (500m X 500m) coinciding with the foraging scale and average distance travelled by ungulates in search of water. Fine scale habitat suitability maps therefore reflected the decisions made by ungulates and lions on where to feed, rest within their home range. Gradually coarsening the spatial scale and habitat relationship at higher scales are more likely reflected the "carrying capacity" of the PA and how many groups the PA could support. Because of the different processes operating at different scales, the relative importance of different variables might often have changed with the scale. Therefore multiscale species distribution models performed better than single scale models with more practical value and Managers could make use of these maps in many ways in understanding how animals make choice of the habitat for foraging, dispersal, and other important activities. The identification of this "organism-centered approach" though was a complex task but has immense importance to wildlife managers in understanding the habitat selection pattern of species, thus implementing proper management practices.

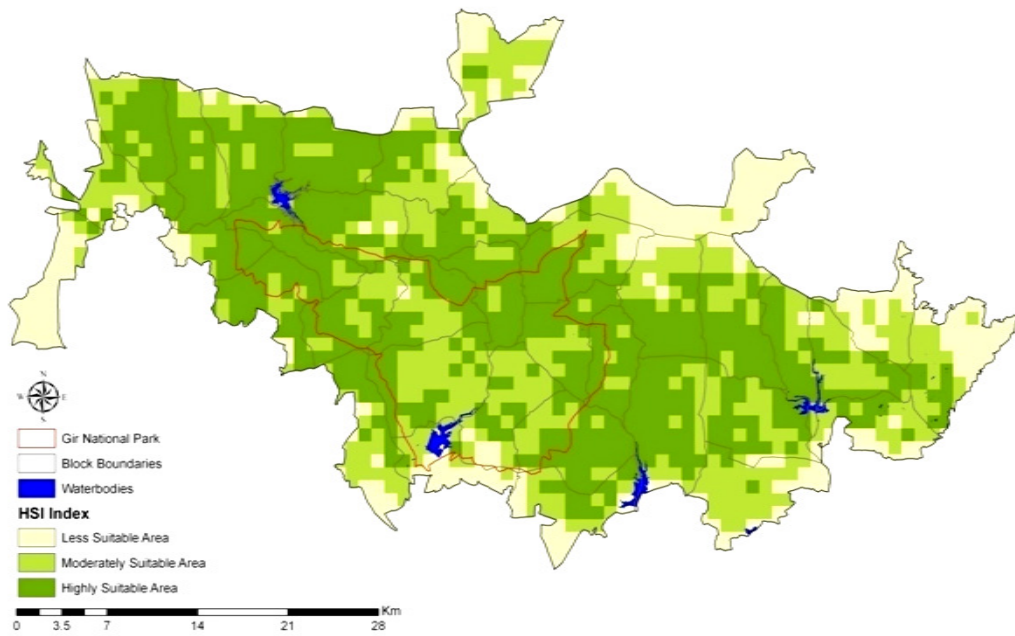
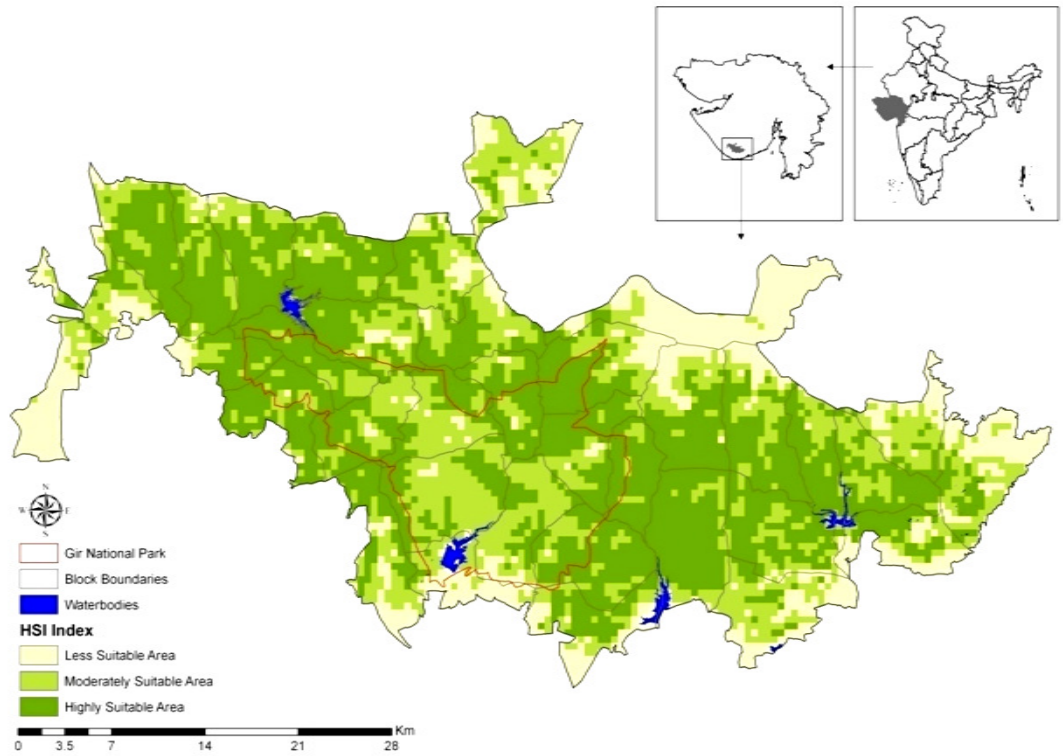


Figure 10: Species distribution modeling of Asiatic lions at 500mX500m, 1KmX1Km spatial resolution

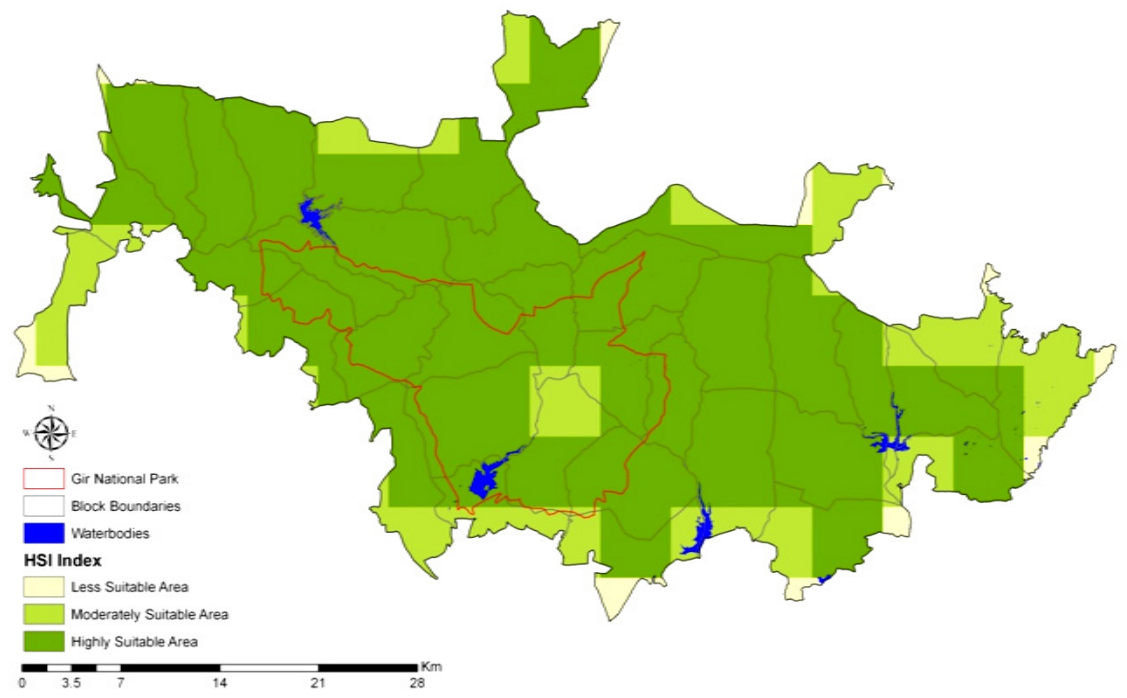
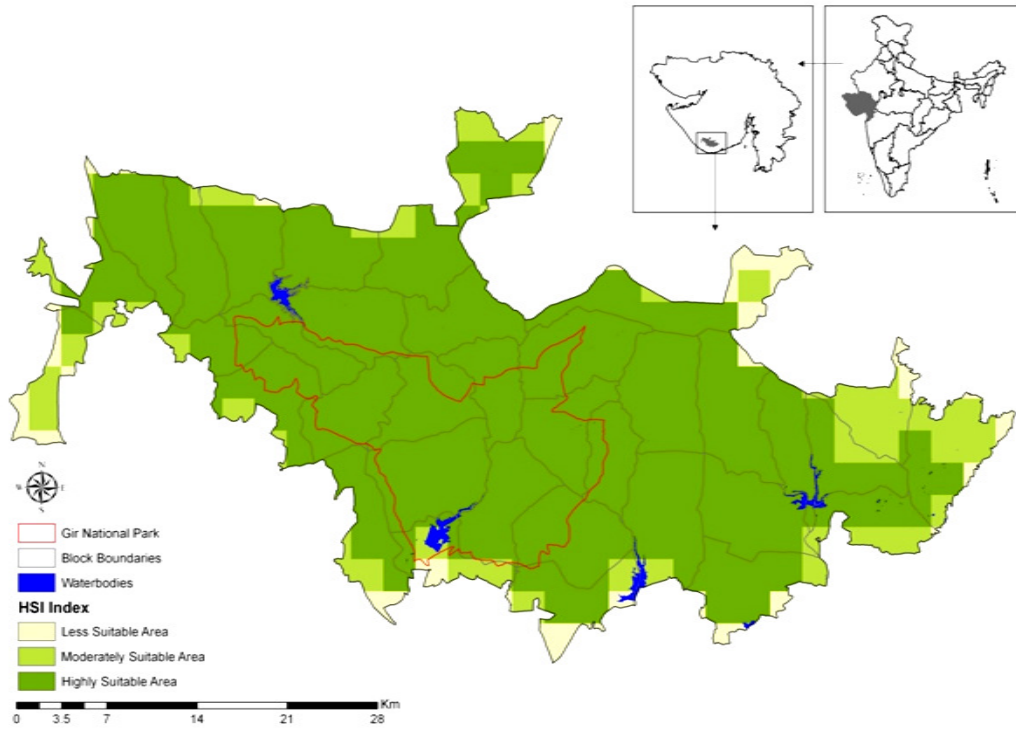


Figure 11: Species distribution modeling of Asiatic lions at 2.5 kmX2.5Km, 5KmX5Km spatial resolution

This scale sensitive research increased our ability to evaluate habitat use pattern by lions (Figure 10 and 11) and different ungulate species across the protected area. Only with the multiscale approach these patterns were ecologically interpretable. Habitat evaluations conducted at fine scale emphasized the importance of digital elevation model, euclidean distance to nesses, water resources etc. while those conducted at coarser scales have focused on mainly availability of forest cover.

This study was essential to plan development in consonance with conservation objectives for the general good for the society. The models developed through this study will assist in site specific management for restorative inputs, mitigation strategies and minimizing the potential for human–lion conflicts as well as habitat improvement of the Gir PA with proper management interventions It was a timely study to undertake, so as to plan for the conservation of a most important flagship species. In long term sustenance of Asiatic lions (*Panthera leo persica*) in the agro-pastoral landscape of Saurashtra, the contribution of this study can be summarized in following points:

1. This study has contributed to the understanding of spatial and temporal dynamics of vegetation growth of the Gir protected area and will form a basis for better planning.
2. This study demonstrated that Gir WLS is effective in preserving dry deciduous forests and lion's habitat against the alarming rate of transformation of forests into human conducive land use classes outside protected area.
3. This study supported the long time debate of thickening of Gir protected area following the cyclone of 1982 which might have happened in response to canopy opening within the protected area due to the uprooting of more than 2 million trees following cyclone and succession of the existing vegetation communities. This trend of increase in thickness was visible till 2002, after which it stabilized.
4. In understanding animal-habitat relationships using multiscale species distribution modeling, the strongest relationship with environmental variables was observed at smaller spatial scale (500m X 500m) coinciding with foraging scale of ungulates. Small-scale ungulate–habitat relationships therefore reflected the decisions made by ungulates about where to feed, shelter, drink

and avoid predation, anthropogenic disturbances within their home range. This results further illustrated of how key habitat features (e.g. waterholes in this study), determined the dispersion of prey and might have a determining influence on predator spatial ecology and movement patterns. Relationships at larger scales reflected the occupancy, carrying capacity of the locality. Because different processes might be operating at different scales, the relative importance of variables changed with scale and thus multiscale species distribution modeling (SDMs) performed better than single scale models and would therefore have greater practical value.

5. This habitat suitability modeling study covering the entire Saurashtra landscape and major portions of Gujarat provided the first detailed map of distribution of suitable habitat patches of Asiatic lions across the landscape. The output of the model identified habitat areas those are highly fragmented, potential movement corridors; though are constrained by roads or other factors at many points, and areas where particular management actions may benefit multiple species. They can therefore help in prioritizing management interventions, as well as identifying areas where certain management actions should be avoided entirely (e.g., road-building, mining in a critical connectivity area).
6. There is relentless and increasing human pressure in this landscape making it lion hostile in near future. Therefore, our modeling approach can assist in identifying conservation priorities in a spatial context integrating it into a decision-theory framework. Successfully communicating our understanding of ecological impacts to policy and decision makers is crucial in this process.
7. The human dominated landscape between Gir and Girnar is used by lions primarily for movement purpose. Lions use streams, barren land, *gauchar* (pasture land), agricultural field and scrubland for movement and also for shelter when it becomes necessary. As a sensible attempt, the identified corridor area is declared as “ECOSENSITIVE” under The Environment (Protection) Act 1986, Government of India by Gujarat Govt. with restrictions on high impact industry and mining activities.
8. Wooded and shrubby vegetation together with the corridor functioned as day *refugia* as well as passage ways for lions, signifying their importance to lion's movement in this landscape. Regular use of mango orchards and *gauchar*

indicated that such patches are important for movement in fragmented landscape and serve as a link to other forested patches. Agricultural intensification, and the associated loss due to explosion of urbanization, has resulted in riverine forest patches acting as an important contributor to biodiversity conservation in this agro-pastoral landscape.

9. Analyses of land use and cover change processes within the Gir-Girnar corridors suggested that riverine patches are vanishing at an alarming rate and the remaining patches are likely to be converted to agriculture. Sustainable management of these forest fragments huddled along the riparian corridors of tributaries of the main river Ozat and other small rivers is urgent with a focus on biodiversity.
10. Though, no study is yet conceptualized on assessing the effects of highways on the movement of lions, however, few nodes (n=12) have been identified, serving as bottlenecks (passageways across highways) and incorporation of crossing structures for wildlife is recommended in road-construction and -improvement projects.

Introduction and Review of Literature

1.1. Background and Significance

1.1.1. Target species: Asiatic lions (*Panthera leo persica*): Origin, History and Present Distribution

Widely distributed historically, population of Asiatic lions (*Panthera leo persica*), presently, survive in a metapopulation framework in and around Gir forest, in the southwest part of Saurashtra region of Gujarat, India. The historical range (Figure 1.1) of Asiatic lions once extended from Syria across the Middle East to Eastern India (Joslin 1973, Kinnear 1920). The lion entered India from Persia at least 6000 years ago (Divyabhanusinh 2005) through the north western passes like Bolan (Sinha 1975). The northernmost limit of lions in Asia was in Caucasia, lying between the Black Sea and the Caspian Sea. Owing to the cause of indiscriminate habitat destruction and game hunting, their population in Asia started declining rapidly from nineteenth century onwards. The lion population in Europe, Asia and parts of Africa, fragmented and even disappeared in certain cases, due to the successive human control of the environment (Singh 2007). Though greatly destroyed, they were yet to be found in small numbers in many countries of Asia towards the end of the eighteenth century.

The evolution and migration of lion along with all other mammals was associated with climatic changes, since a large number of them evolved in the Pleistocene era, when the climate fluctuated considerably (Barnett et al. 2006, Antunes et al. 2008). The lion being a resident of open-country-savannah, sparse sub-tropical or dry, deciduous scrub forests, and dry thorny forests, became marginalized as a consequence of the progress of the dense forest, which was unsuitable habitat, and moved to Africa through Palestine, Israel, Egypt, and to India through Syria, Mesopotamia, and Persia in the Azilian period of the Neolithic age (Singh 2007, Schnitzler 2011).

Widespread areas in North and Central India were found inhabited by this large carnivore, in the states of Punjab, Haryana, Rajasthan, Uttar Pradesh, Madhya Pradesh, Gujarat and Western Bihar (Dalvi 1969, Fenton 1909). The southernmost limit appeared to be the Narmada river (Joslin 1973). By 1888 the last lions in India outside the Gir forest had been shot (Lydekker 1895) and the last report of Asiatic lions outside India was in Iran (Heaney 1944, Champion-Jones 1945). While indiscriminate hunting wiped out lions from the rest of India by 1880 (Pocock 1930), within the Saurashtra region, they were found in Dhrangadhra, parts of Jasdan, Chotila, Alech hills, Barda hills, Girnar and Gir. By the turn of the 19th century, growth in areas under cultivation separated Gir from Girnar, Alech and Barda hills and as a result, the lion population got confined to the Gir forest (Dalvi 1969).

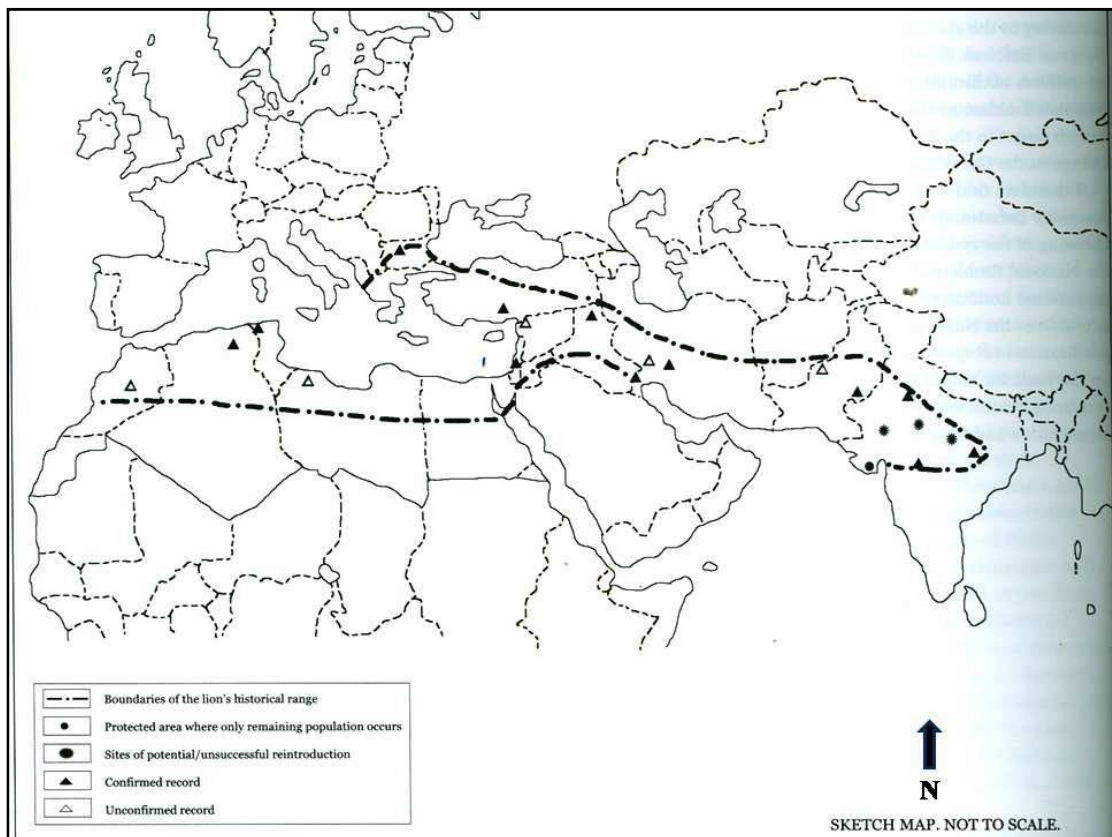


Figure1.1: Past and present distribution of lion in North Africa and Asia

(Source: Nowell and Jackson, 1996)

Over the time, the distributional range of lions has consistently shrunk, but the population numbers within the area has increased (Singh 2007). Believed to be less than 50 animals in 1880 and very close to extinction (Wynter-Blyth and

Dharmakumarsinhji 1950) but owing to the timely protection measures taken by the Nawab of Junagadh, the lion population improved to a number of 70 (Singh 2007). However, a famine, between the years of 1901 and 1904 brought the lions in unavoidable conflict with people, declining the lion population to a mere 20 at the end of the famine in 1911. In 1936, the population census indicated a population of 287 (Dalvi 1969) and lion counts in the following years, in 1950, 1955 and in 1963 estimated the population to be 227, 290, and 285 respectively (Wynter-Blyth and Dharmakumarsinhji 1950, Wynter-Blyth 1956, Singh 1997). Dispersal of lions started after the draught year of 1987 (Divyabhanusinh 2005). Lions were known to move out of Gir in post monsoon and monsoon seasons (Singh 1997). These isolated populations continued migrating in and out of Gir and keep interacting with the parent population in Gir. The emigrant lions used riverine habitat patches between Gir and the other satellite populations as their passageways.

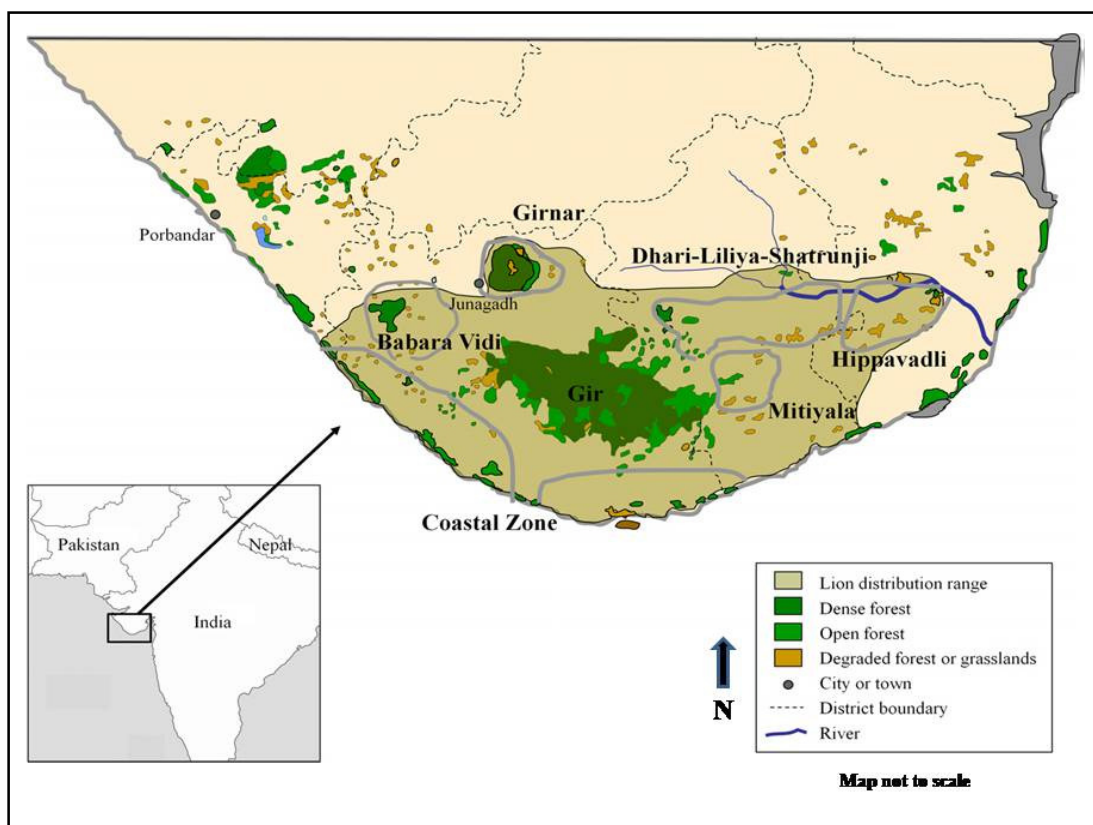


Figure 1.2: Present distribution of Asiatic lions (*Panthera leo persica*)

(Source: Singh and Gibson 2011)

With the good management strategies practiced by the Gujarat forest department, the population of lions has since shown a steady increase and presently the Gir landscape harbour (Figure 1.2) the only free ranging population of the species. The recent

population estimate of Asiatic lions (Gujarat Forest Department 2010) confirmed the presence of 411 individual lions, distributed in Gir National Park and Sanctuary and adjoining areas (297), Girnar Sanctuary (24), Mitiyala Sanctuary (7), Paniya Sanctuary (9), Coastal areas of Una, Kodinar, Sutrapada and Chhara (21), Savarkundla, liliya, and its adjoining areas of Amreli and Bhavnagar districts (53).

The sparse vegetal cover near the riverine area with predominance of thorny species and few orchards on the river bank provides shelter to the migrating animals and acts as their passageways (Jhala et al. 2012). Coastal area being water deficient in nature, and in absence of any undergrowth but *Prosopis* thickets, lions survive in an unusual habitat. With the advancement in the industrial and agricultural development there is all possibilities that the existing routes of the migratory population will be lost to the heavy limestone mining industries leaving the area unfavorable for lions movement and it is thus essential that the present sanctuaries and the associated habitats be properly conserved and managed for the survival of this rare gene pool.

1.1.2. Importance of studying landscape pattern and the impact of landscape dynamics on species conservation

The vast wild areas that are often needed to safeguard large carnivores are becoming increasingly scarce. Therefore, a firm understanding of how geography interacts with species to shape evolution, ecological relationships and landscape processes is required, establishing biological priorities for conservation. The relationship between landscape configuration and animal distribution could markedly differ between landscape contexts – for example, between natural and anthropogenic forest landscapes. In addition, response to landscape structure has been found to be species specific (Wegner and Merriam 1979, Middleton and Merriam 1983) and a number of mechanisms like behavioral differences, degree of habitat specialization, and levels of adaptive flexibility drive species response to landscape structure (Andr n 1994, Andr n et al. 1997).

Resources like food, cover, water and environmental conditions (temperature, precipitation, presence or absence of predators and competitors) altogether form habitat of a species, an area that supports occupancy by individuals of a given species (or population) and permits those individuals to survive and reproduce (Johnson and

O'Neil 2001). Thus survival rates and reproduction of the individuals that live there, the vitality of their offspring, are linked to the quality of the habitat (Van Horne 1983), and to the span of time the site remains appropriate for occupancy. Considering all these, the importance of the spatial pattern of habitat network and landscape matrix to biodiversity was acknowledged by population ecologists in the last decade (Henderson and Merrium 1985, Opdam 1987, Merrium 1988, Opdam 1988, 1991, Fahrig and Merrium 1994, Opdam et al. 1995, Fahrig 1999, Opdam 2002).

Climatic factors, through its influence on abiotic factors, influence habitat of a species, hence, influence species distribution. Rainfall influences vegetation, mediated through edaphic and topographic gradients (Bell 1982, McNaughton 1983, 1985) and can induce changes in habitat suitability capable of significantly modifying predator-prey relations (Smuts 1978, Whyte and Joubert 1988, Mills et al. 1995). Rainfall and soil nutrient availability were found to be decisive factors of the primary productivity of vegetation and hence the regional aggregate and species specific biomass densities of large herbivores (Coe et al. 1976, Bell 1982, East 1984, Fritz and Duncan 1994) and carnivores (East 1984, Van Orsdol et al. 1985) in African savannas. In addition to food and vegetation cover, rainfall affects the distribution of drinking water, thereby modulating the spatio temporal distributions of water reliant herbivores and carnivores (Western 1975, Hanby et al. 1995, Owen-smith 1996).

Rainfall found positively linearly related to the annual primary productivity of the grasslands in the Serengeti-Mara ecosystem of Kenya and Tanzania (McNaughton 1985, Sinclair 1975), including a parallel gradient of grass stature (McNaughton 1983) and probably in resident ungulate density (Campbell and Hofer 1995). However, lion density in the Masai Mara National Reserve was found lowest in the western section where rainfall was highest (Ogutu and Dublin 2002), contrary to the expected trend that resident ungulate biomass increased with rainfall. The Gir ecosystem is sustained by a complex interaction of the physical, biological and socio-economic sub-environments, which in turn are subjective to various forms of management interventions. In this semi arid and arid landscapes the physical environment (which itself is synchronized by the complex interaction of precipitation-evapotranspiration) plays an overwhelmingly significant role and influences the

biological diversity, productivity, and socio-economic profiles in the area (Mathur and Mathur 2004). Based on the conservation values, the management focus for Gir PA has been set to protect and conserve the Gir forest in a way that is consistent with the crucial and uninterrupted conservation of the Asiatic lion and all other life forms, systems that together constitute this unique ecosystem (Jhala 2004).

In spite of the appreciation for heterogeneity at some scales, most of the ecological theories remain unaffected assuming system having a uniform structure and the mechanisms dealt with that also being consistent in space (Kingsland 1985). Therefore for decades, the fact that system distinctiveness change naturally over time was overlooked. Progression took place later with the appreciation of the idea that even if the system is uniform but it exists within a matrix of other systems that vary in structure and function (Wiens 1995). Thus, the monitoring of landscape spatial pattern at a range of temporal scales has several applications (Turner 1989) and remotely sensed data have been used successfully to keep an eye on changes in forest cover (Hall et al. 1991, Skole and Tucker 1993), to enumerate changes in fragmentation specifically (Skole and Tucker 1993, Luque et al. 1994, Vogelmann 1994, Pindar et al. 1999, Staus et al. 2002, Turner et al. 2003). Patch based analysis is considered as the scale of analysis and consequently changes in the number of patches, average patch size, inter-patch distance or other measures are used to evaluate forest loss or gain. The long term monitoring of landscape spatial pattern, in addition to other biophysical variables, might lead to the detection of greater range of processes of landscape modification (Lambin and Strahler 1994).

The present lion population is scattered in various pockets with Gir having the largest concentration, others being Girnar, Coastal zone and Mitiyala. Within the coastal zone also there are three major pockets where the population resides normally – Sutrapada, MulDwarka and Nava Bandar (Singh 2007). With the increase in human activity in Gir and the onset of industrialization especially the cement factories, the landscape of the area has changed (Srivastava et al. 1998). At the same time each satellite population is affected by environmental dissimilarities, habitat quality, diseases, human impact, inbreeding depressions etc. Progressively more human and lion ranges are overlapping, with adverse consequences of both species (Chellam and Johnsingh 1993).

The intensified agricultural activities delimited the Gir forest from all around in a slow but gradual manner. The highly fertile black cotton soil around Gir made it distinct from other areas for growing several cash crops like “Kesar” mango, immensely altering the land-use pattern in the area. The adequate demand of firewood and timber of 97 villages within a radius of 5 km neighboring Gir protected area made the forest exposed to the illicit cutting (Divyabhanusinh 2005). The forested area of Gir shrunk from 3070 km² in 1880 to 1884 km² in 1900s due to spreading out of agriculture and damage of habitat (Singh 1995). Green revolution replaced the grazing land by cultivation meant the shrink of range of Asiatic lions, cover and more importantly the wild and domestic animals upon which the lion depended for food. Bulk of the lion population disappeared from the outside of the sanctuary, remaining only in few patches not cleared for cultivation (Joslin 1984, Divyabhanusinh 2005).

1.1.3. Emergence of habitat suitability modeling (HSM) in conservation studies and effect of spatial scale in HSM studies:

The strategies of a predator is reinforced by natural selection by maximizing nutrient intake while being tempered by a wide range of ecological constraints, such as prey density and habitat, that differ throughout its geographical distribution (Sunquist and Sunquist 1997). Natural selection favors efficient, optimally foraging predators (Krebs 1978) and habitat features play an important role in increasing the survival chances or reproductive success of the predator by influencing hunting activities. Physical environmental features are used by all felids extensively to get as close as possible to prey while hunting (Sunquist and Sunquist 1997). Based on the hypothesis that either individuals or groups of species will select areas that best satisfy their life requisites (Schamberger and O’Neill 1986), habitat suitability modeling (Scott et al. 2002) for focal species is one tool that is utilized extensively to guide forest planning (Angelstam et al. 2003a, b, 2004, Gibson et al. 2004).

From identifying national-level priority areas to local site habitat management, entail different conservation planning approaches at different scales (Cabeza et al. 2010). Thus scale has been accepted as an imperative factor in ecological studies focusing on disturbance (White 1979, Mooney and Godron 1983, Pickett and White 1985), forest dynamics including implications for habitat preservation and species conservation,

(Burgess and Sharpe 1981, Harris 1984). While change in patterns with scale was long recognized in ecology (Greig-Smith 1952), the prolonged application of the scale concept in the design and interpretation of surveys, comparative studies, and controlled experiments is new (Schneider 1994). The rising focus on scale appears to be an enduring change in the way that ecological research is pursued (Schneider 1998).

Since a few years, multi-scale approaches have been developed to study ecological systems (Bergin 1992, Gutzwiller and Anderson 1987, Saab 1999, Cushman and McGarigal 2002) and to explore the significance of a range of scales in the distribution of animals and in the formation of their communities (Herrando and Brotons 2002, Stephens et al. 2003). Testing the effects of spatial scale is done using a consistent set of variables and holding the resolution of any classification stable (O'Neill et al. 1991, Qi and Wu 1996, Steffan-Dewenter et al. 2002, Williams et al. 2002) with the objective to understand which processes act at each scale, how adjacent scales interact, or how species vary in their sensitivity to a given scale. Analyzing habitat relations in this way assumes that animals recognize different types of patterns at different spatial scales – generally that they distinguish finer aspect in habitat structure or heterogeneity at finer spatial scales (Hildén 1965, Hutto 1985).

Hierarchy theory (Allen and Starr 1982) has a significant bearing on scale, because systems are naturally nested in space and time. Variables influencing a process may or may not change with scale, but the relative importance of the variables or the apparent direction of the relationship often changes when spatial or temporal scales are changed. If we change the scale, the relevant processes or even the direction of relationships that we monitor may well change. Even, within the local environments under same climatic conditions, disturbance regimes (such as fire, timber harvesting or wind), physical environments (such as soil texture, fertility and nutrient status), biotic factors and neighbourhood effects (Frelich et al. 1998) decide the development of spatial variation in habitat quality. In addition, a species' response to landscape structure depends on its perception of grain and extent (Kotliar and Wiens 1990). Hence an organism's perception of grain can influence its response to landscape heterogeneity, and can define the extent of the functional landscape (Kotliar and Wiens 1990, Corkum 1999). The grain of perception, response to heterogeneity is also

influenced by the spatial and temporal scale of major ecological processes (such as dispersal or foraging) acting on, or exhibited by, an organism (Addicott et al. 1987). Thus habitat specificity, habitat availability, and spatial context, can all influence what might be considered a 'functional landscape', and by extension, will influence the relationship between landscape structure and species distribution.

Individual organism respond differently to its environment changes over several spatial scales and hierarchical levels; with the smallest scale corresponding to the grain of the organism and the largest scale being at least its home range (Kotliar and Wiens 1990). Relevant spatial scales for carnivores varies from microsites (e.g., specific structures used for denning, caching, or resting) to forest stands (e.g., the structure and composition of forest stands that provide suitable resting or foraging microhabitats) to home range (the mosaic of habitat conditions that provide all life requisites for an individual animal) to landscape (regional characteristics that support a population or metapopulation of interacting individuals) (Spencer and Rustigian-Romsos 2012). Different aspects of an organism's life history might decide habitat selection at each scale e.g. foraging decisions might occur at fine scale, while finding a mate occurs at a coarse scale. Habitat selection might get influenced by inherent differences in biotic and abiotic processes operating at each scale (Urban et al. 1987). Studies conducted over several scales more readily incorporate scale-dependent relationships to allow greater understanding of how organisms assimilate information (Ritchie 1997) and make choices that influence habitat selection and ultimately fitness.

While at the local scale, habitat quality of a particular species is expected to be of major importance (Duelli 1997, Shreeve et al. 2004), at the coarser scales, climatic factors (Pollard et al. 1996, Hill et al. 1999, Warren et al. 2001) along with landscape structure (Hanski 1999, Thomas and Kunin 1999) affect the persistence of natural populations. Landscape composition (i.e. amount and numbers of habitats present) is one key factor (Wagner et al. 2000), but also landscape configuration (i.e. spatial arrangement and connectivity of habitat) has a strong influence on local populations (Hanski and Gilpin 1997). Examining the relative importance of site and species specific local habitat suitability, climate, and landscape structure is thus of great interest to theoretical and applied ecology as well as conservation management.

The availability of water in Gir affects wildlife most. Scarcity of water during the pinch period i.e. March to June causes local shifting of the animal populations. Water being available only in principal rivers, the Kamleshwar reservoir and few other water holes during this period, northern Gir suffers the most from water shortage. Concentration of herbivores around the water sources during the summer months causes momentary scarcity of forage in the area. Thus during the pinch period, both food and water become the limiting factors for the wild herbivores (Khan et al. 1990, Singh 2007). A five year scheme was developed in Gir in 1972, in which, dreadful conditions of Gir ecosystem was reversed and habitat was improved in the park. Construction of water harvesting structures and artificial water holes improved the water accessibility to the wildlife and by that time food was not a major limiting factor in the Gir as population of important herbivores increased but shortage of space resulted in territorial fighting and migration.

Amongst the other influences in Gir, there was heavy pressure of grazing by domestic livestock and removal of grass through grass cutting. Extreme grazing in the sanctuary also lead to the worsening of the ecosystem, causing replacement of palatable grasses with unwanted unpalatable grasses and hardy weeds like *Cassia tora* and *Lantana camara* (Singh 2007). These overly grazed areas remained unfertile for the wildlife, especially regeneration succumbed casualty due to excessive grazing. Damage to the forest was inflicted by the resident *maldharis* and forest settlers (Kamboj et al. 1997). The accumulation of plenty of inflammable material like dry grass, dry teak leaves, fallen twigs and other dry vegetational residues on the forest made the forest vulnerable to forest fire. The surface devoid of the organic matter became subjected to more damaging soil erosion following monsoon, slowly and steadily degrading the habitat.

To understand the factors, influencing the ecology of lions and its major prey species; previous studies in Gir were referred. A study done on estimation of ungulate densities (Khan et al. 1996) revealed, all ungulate species in Gir showed considerable variation in their spatial abundance within three management units (Gir West, Gir East and National Park) of Gir and differed in terms of values of diversity of habitat use. This was attributed to the high spatial heterogeneity across Gir and showed a greater influence of habitat factors (e.g. cover, habitat structure, availability of food)

on distribution and densities of ungulates in Gir. The construction of large reservoirs in Gir (e.g. kamleshwar and Machundri) indirectly influenced the chital population by creating ideal habitat conditions. Open areas on the periphery of those reservoirs have been recolonized by *Acacia* and *Zizyphus* species as a consequence of secondary succession intermixed with *Tectona-Acacia-Zizyphus* woodland and riverine woodland. This provided a high diversity of habitats and food species with ample ecotones preferred by chital (Mishra 1982). Sambar exhibited its preference for hilly terrain, dense shrub cover and absence of biotic disturbance (Johnsingh 1983, Khan et al. 1990). Chowsingha and chinkara, though similar in size and diet, varied in terms of values of diversity of habitat use due to their contrasting cover requirements. While chowsingha occupied dense hilly areas throughout Gir, Chinkara was restricted to open wooded grasslands of South East Gir. Study on Gir ecology and behavior of the lions showed that food pattern of lion has improved in favor of wild prey as 65% of lion diet was recorded from wild animals whereas 35% was supplemented through livestock kills (Chellam and Johnsingh 1993).

1.2. Conservation Efforts and Research Activities in Gir

Gir has always been the focus of research activities, whether it's on the majestic Asiatic lion or the biodiversity. Ecological research in Gir began only in the late 1960s. Being the last abode of Asiatic lions, major research activities were always focused on conservation aspects of Asiatic lions (*Panthera leo persica*). Numerous accounts of writing by naturalists about the Asiatic lions and Gir can be found in the British Administrative records and the Junagadh State Records. Likewise, The Journal of the Bombay Natural History Society published many articles on Gir and the lions. Study of prey relationship and the vegetal ecosystems saved the lion population from the brink of extinction (Dharmakumarasinhji 1968). Introduction of lions in the Chakia forests of Uttar Pradesh, its multiplication and sudden disappearance from the sanctuary were well documented and the pre-requisites required before launching such experiment were also presented (Negi 1969). In 1972, in a paper by Dharmakumarasinhji, lion's presence was documented in Veraval, Chorwad, Mangrol in the coastal orbit (Dharmakumarasinhji 1972). Actions were taken from time to time by authorities to ban shooting and saving future of Asiatic Lions (Sinha 1975). The

sudden decline in lion population during nineteen fifties and sixties, grazing by cattle was a great problem at that time (Joslin 1984). Constant attacks by lions on humans hampered support among local people for lion conservation (Saberval et al 1994). In 1985, Joslin focused on the reasons responsible for the decline in lion's population in Gir forest and came out with the reality of dependency of Asiatic Lions upon livestock (Joslin 1985).

In 1998, it was anticipated to establish a second free ranging population of Asiatic lions in Kuno wildlife sanctuary in addition to improve the quality of wild ungulate habitat by thinning of teak (Johnsingh et al. 1998). The behavioral and chemical aspects of scent marking in Asiatic Lions were also investigated (Brahmachary et al. 2000). Use of modern technologies to estimate lion population started in 1999, applying vibrissae spot pattern for individual identification (Jhala et al. 1999). Various other researches also emphasized on other areas of Asiatic lion conservation. The techniques for determining prey items based on hair characteristics from scats of the free ranging Asiatic lion were improved in terms of accuracy and time efficiency (Mukherjee et al. 1994). Major prey species of lion were also studied to understand the relationship of them with the ecosystem (Khan et al. 1994).

Numerous problems such as temples, biotic pressures exerted by *maldharis*, a pastoral tribe, and the villages on the periphery of Gir PA hindering the prospects for long term survival of Gir forest and its lions were addressed (Johnsingh et al. 1998). Apart from lions, several studies were also conducted to look at prey base status of lions and its relationship with the ecosystem. Berwick recommended to reduce or eliminate the livestock pressure with introduction of the grass eating ungulate (Berwick 1976). The death and debarking of trees and shrubs, following a severe draught in 1987, were investigated in Gir lion sanctuary and National Park (Khan et al. 1994). The habitat occupancy and factors governing the distribution of major ungulate species were also investigated (Khan 1994). It's not only vertebrates, Gir represented very good invertebrate fauna too (Majumdar 1973) and had the largest population of marsh crocodile in India (Joseph et al. 1975).

Study was also done on vegetation, ungulate population and status of *Maldhari* grazers from 1987-1989 to understand the conservation and management status of Gir

Lion Sanctuary (Khan 1995). Contribution of the flora to the Gir forest (Santapau and Raizada 1954) and more intensely, phytosociological studies of the vegetation (Menon and Shah 1982) were also done. A study on the impacts of management practice on lion and ungulate habitat was conducted in Gir (Sharma 1995). The issues of poaching and illicit cutting of vegetation were also focused for the conservation of ecosystem in Gir (Dharmakumarasinghji 1972). Appropriate management suggestions to augment soil moisture were proposed (Patel et al. 1993). Impact of *maldhari* tribes on the ecosystem (Pati 2000) and collaborative efforts of the people and forest department for the sustainability of the ecosystem (Srivastava 1997) was discussed at the same time. The impact of *maldhari* pastoralists on the Gir ecosystem and the socio-economic conditions under which they existed in the 1970s was reported in studies (Berwick 1990). She explored the various management options to reduce the inherent conflicts between the *maldhari* lifestyle and the conservation of Gir habitat.

1.3. Justification of the study

Studies on African lions have discovered that lion populations differ ecologically, even within an ecosystem (Hopcraft et al. 2005). Hence, habitat affects the social biology of lions (Funston et al. 1998). A number of studies have been done on numerous aspects of wildlife in Gir like; the possible reasons for the decline in the population of Asiatic lions (Joslin 1973), the conflict between wild ungulates and livestock population and the role of predators in Gir (Berwick 1974, 1976), ungulate habitat ecology in Gir (Khan et al. 1990), vegetational and wildlife studies in Gir forests (Chavan 1993), ecology of Asiatic lions (Ravi Chellam 1993), the predation pattern of Asiatic Lions on livestock in Gir. (Srivastava 1997). The feeding niche of Asiatic lion and leopard in Gir (Singh et al. 1999) alongwith the seasonal changes in the food habits of the Asiatic lion and leopard in Gir PA were studied in detail (Dharaiya et al. 2005).

The use and applicability of the vibrissae spot method for individual identification and population estimation of Asiatic lions (Jhala et al. 1999), preparation of a broad level vegetation map of Gir (Sharma et al. 1999, Qureshi and Shah 2004), studies on the vegetation composition and the impact of *maldharis* on vegetation and the habitat

utilization by ungulates (Sharma and Johnsingh 1995) revealed the importance of the sustainability of the landscape for better regional conservation of most of the endangered flora and fauna.

The ongoing long term study on Asiatic Lions, started in late 1980s, is an indication of the prioritization and necessity of the conservation of the big cat in a complex landscape of Gir (Johnsingh et al. 1998, Jhala et al. 1999). Studies revealed that the present lion population in Gir is at its carrying capacity (Singh 1995) and therefore search for a potential site for lion re-introduction has also been done (Chellam 1994). A monitoring protocol has been prepared giving emphasis on different aspects of Gir (Jhala et al. 2004). Lion diet was assessed both by intensive search across the study area and the consumption of wild and domestic prey also varied between seasons ($\chi^2=22.3$, $df = 2$, $p < 0.0001$) showing a greater proportion of wild prey during summer months (Meena 2008). The reproductive strategy of Asiatic lion was summarized from the study of the social structure, ranging patterns, land tenure system and activity patterns (Meena 2008). A systematic effort was made to evaluate the direct as well as indirect negative impact of sympatric livestock on chital ecology (Dave 2008). Results suggested that the magnitude of livestock activity matters rather than just presence. Therefore, regulation of livestock number and spatial distribution of livestock herds could be the best management strategy for the conservation of wild prey population of rare and endangered Asiatic lion in Gir without affecting the livelihood of *Maldharis*.

Adult Lion populations in the Gir landscape were estimated in a mark-recapture framework based on individual lion identification, Gir lions increased from about 177 in 1968 to about 411 by 2010 with an $r = 0.022$ (SE 0.001) translated into an annual population growth of 2.2%. Male: female ratio was 0.63 (SE 0.04) while cub: adult lioness ratio was 0.37 (SE 0.02) (Banerjee 2012). To ensure long-term viability of Girnar lion population the corridor area between the Gir PA and Girnar was delineated and declared as 'ECOSENSITIVE' under the Environment Protection Act, 1986, so as to curtail possible negative impacts of further developments (Jhala et al. 2012). Findings suggested that a combination of strict protection regime for lions, *Maldharis*' traditional reverence towards lions and the livelihood economics would permit the delicate balance of lion-*Maldhari* coexistence (Banerjee 2012).

With these studies in account, the present study addresses the following objectives:

1.4. Objectives

1. To evaluate the current landscape architecture of Gir Protected Area and its surroundings.
 - (a) What is the current landscape pattern?
 - (b) What is the appropriate spatial scale for assessing landscape for lion conservation?
2. To assess the rate of change in land use and vegetation patterns.
 - (a) What is the pattern of change in the landscape?
 - (b) What habitat types are vulnerable to change and why?
3. Develop habitat suitability models at different spatial scales for Lions and principal prey species.

Main questions of interest were the following:

 - (a) What factors govern habitat suitability?
 - (b) What is the effect of scale on habitat suitability assessment?

In addition to the above objectives, two separate studies were conducted on following objectives and added to the thesis.

1. Evaluating the potential for Asiatic lion (*Panthera leo persica*) range expansion within Saurashtra, Gujarat.
2. Managing the agro-pastoral landscape between Gir and Girnar wildlife sanctuary: Habitat suitability modeling and identifying functional corridors for Asiatic lions (*Panthera leo persica*) in this landscape.

The research is descriptive, mensurative and based on model building. Attempting to seek patterns in distribution of animals so as to develop habitat suitability models. Per say it is based on hypothesis that animals exhibit a clear preference for certain habitats and resources. Alternatively if no such patterns are discernable then modeling is not possible. Therefore the research is not based on manipulative experimentation to test

hypothesis but on mensurative approach seeking patterns. There will not be formal hypothesis testing in the strictest sense. However several trivial hypotheses will obviously be tested.

Hypothesis for the first sub objective:

H₀: With the change in spatial resolution, landscape metrics at patch, class, landscape scales does not change showing any particular trend.

H_a: With the change in spatial resolution, landscape metrics at patch, class, landscape scales changes showing a particular trend which is predictable across the scales.

Hypothesis for the second objective:

H₀: Land use cover changes over years are similar across different habitat types or random.

H_a: Human driven land uses or land cover increases with time.

Hypothesis for the third objective:

For modeling habitat suitability for lions:

H₀: Lions do not exhibit habitat preference i.e. use habitats in proportion to their availability.

H_a: Lions exhibit habitat preferences i.e. use some habitats more in comparison to their availability.

For modeling habitat suitability for principal prey species:

H₀: Ungulates use habitats in proportion to their availability i.e. they do not exhibit habitat preferences.

H_a: Ungulates use some habitats more in comparison to their availability and they exhibit habitat preferences.

1.5. Organization of the Thesis

The thesis is organized in six chapters. Chapter 1, 2 deal with the general introduction, study area, following five (3, 4, 5, 6A and 6B) chapters deal with the

above five objectives. Each of these chapters includes a brief introduction with review of literature followed by methodology, results and discussion.

Chapter 1: Provides a general introduction and explains background of the study.

Chapter 2: Introduces Gir Lion Sanctuary, its history, area and management. It also reviews research carried out in Gir and its management, conservation implications.

Chapter 3: Describes the current landscape pattern of the Gir wildlife sanctuary. It provides detail of the vegetation community structure found in Gir and its spatial distribution. In addition, it deals with the effect of spatial scale in landscape pattern analysis.

Chapter 4: Contains details about the landscape dynamics in a ten year framework and summarizes the pattern of changes occurred in the landscape.

Chapter 5: Describes multiscale habitat suitability modeling for lions (*Panthera leo persica*) and principle prey species of lions i.e. Chital (*Axis axis*), Sambar (*Rusa unicolor*), Nilgai (*Boselaphus tragocamelus*). It also explores the effect of changing spatial scales on habitat suitability modeling.

Chapter 6A: Assesses the potential of the landscape of Saurashtra, Gujarat in supporting the range expansion of Asiatic lions (*Panthera leo persica*) within Saurashtra, Gujarat.

Chapter 6B: Describes the Management of the agro-pastoral landscape between Gir and Girnar wildlife sanctuary: Habitat suitability modeling and identifying functional corridors for Asiatic lions (*Panthera leo persica*) in this landscape

2.1. Introduction

The present study examined the effect of changing spatial scale and extent on the spatial distribution modeling of Asiatic lions (*Panthera leo persica*) and its major prey species. The maximum area covered under this study included (1) A major portion of the landscape of Gujarat (Figure 2.1), where lions are spread at present or the landscape might have the potential to incorporate the spreading lion population, description of which is given in the chapter Six A, (2) The human dominated landscape between Gir and Girnar wildlife sanctuaries (Figure 2.2), different land use classes of which landscape are used by lions as *Refugia* and passageways to move between two otherwise isolated protected areas, Gir and Girnar, description of which is given in the chapter Six B, and (3) The Gir wildlife sanctuary and national park (Figure 2.3), which is described here in detail.

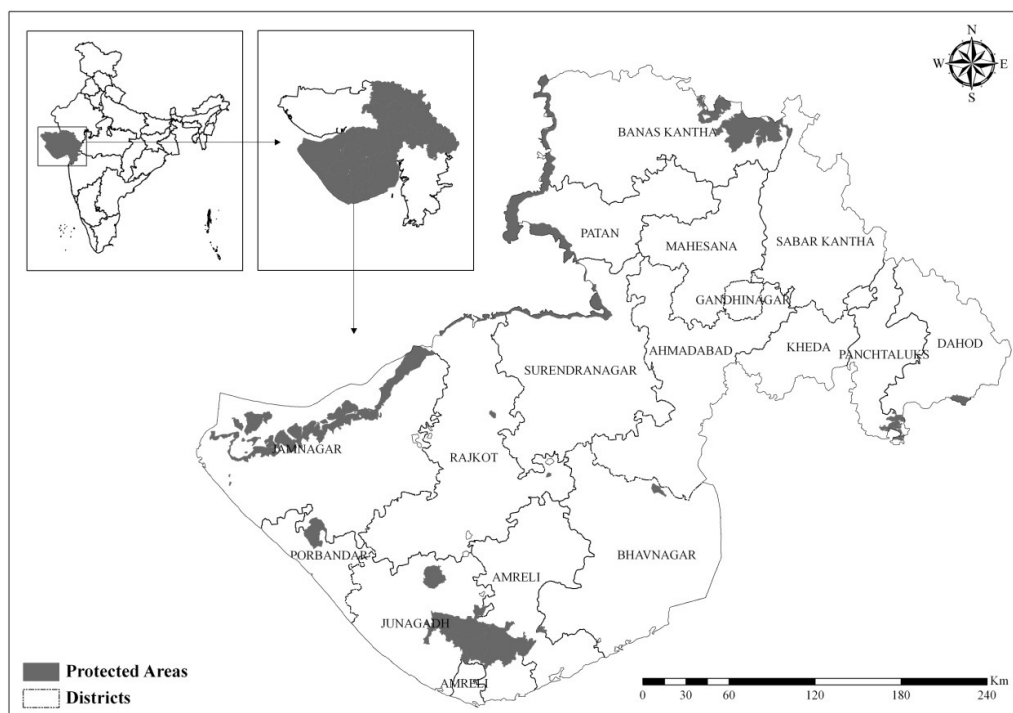


Figure 2.1: The landscape of Saurashtra. The map insets show the location of Saurashtra within India and western state of Gujarat

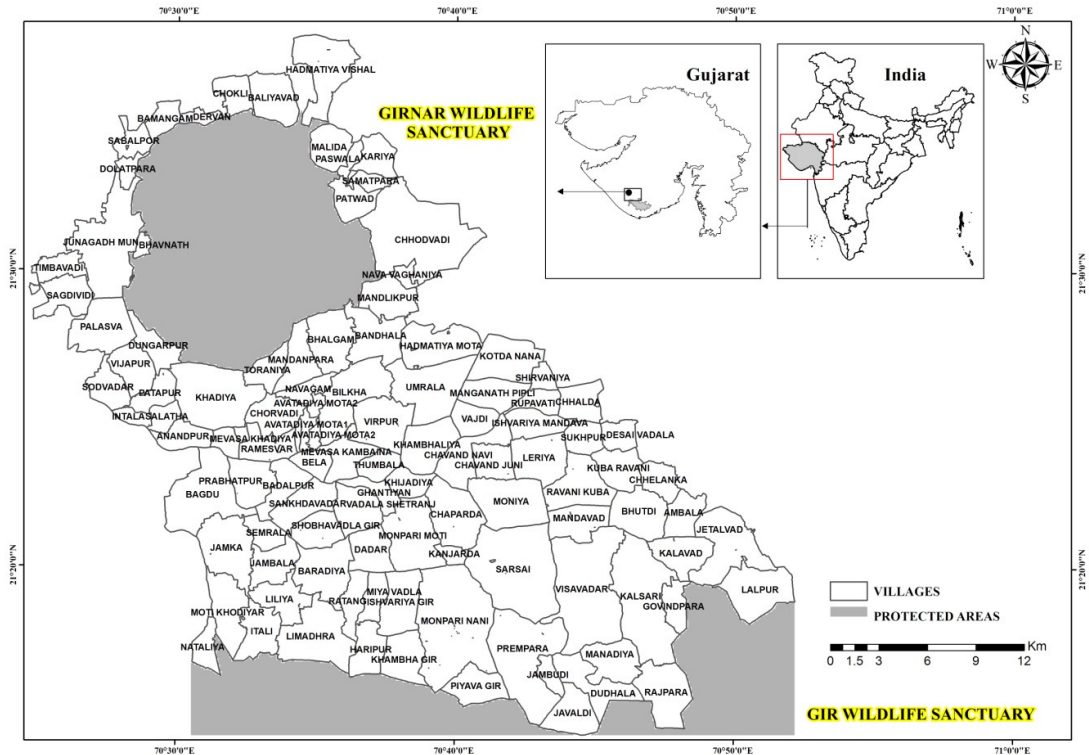


Figure 2.2: The landscape between Gir and Girnar wildlife sanctuaries. The map insets show the location of the study area within India and western state of Gujarat.

2.2. Gir Wildlife Sanctuary and National Park

Since its inception in the 19th century, the Gir wildlife sanctuary and national park was recognized as an important area for the Asiatic lion (*Panthera leo persica*). Lion hunting was banned and a small area, the “Devalia block”, was declared a sanctuary in 1920 (Divyabhanusinh 2005). The present sanctuary was established in 1965 to protect the entire fauna and flora, and was expanded to its present size in 1974 with the primary objective of conserving the sole remaining wild population of lion in India (Khan 1995). Gir protected area (PA), the largest biologically compact continuous tract of dry deciduous forests in the saurashtra region of Gujarat in northwest India, lies between the parallels of latitude 20°40’N and 21°50’N and meridians of longitude 70°50’E and 71°50’E having an area of 1412 km²: The National Park is comprised of 259 km² surrounded by the sanctuary spreading over an area of 1154 km² (Figure 2.3).

Boundaries of Gir Forests are as under:

North: Revenue areas of Junagadh and Amreli districts.

East: Revenue areas of Amreli and Bhavnagar districts.

South: Revenue areas of Junagadh and Amreli districts.

West: Revenue areas of Junagadh district.

Gir ecosystem saved the rare and threatened lion from the brink of extinction (Singh and Kamboj 1996) with an estimated population of about 50 individuals at the beginning of the 20th century to a population of over 400 individuals today (Gujarat Forest Department 2010). This ecosystem forms a part of south central highlands in Saurashtra region and falls in Afro tropical (palaeotropical) realm (Singh and Kamboj 1996). There are many forests adjoining the Gir PA which are almost unbroken with the present sanctuary. These are of varying sizes and most of these areas are managed as protected, unclassed forests and *vidis* and *gauchar* (pasture lands) from which grass is extracted every year. Reduction in existing habitat coupled with increase in lion population due to better enforcement of conservation strategies have resulted in dispersal of lions outside Gir (Singh and Gibson 2011, Banerjee and Jhala 2012) recolonizing their lost territories in Girnar and Mitiyala and also moved to coastal forests but corridors which previously existed, have been vanished to cultivation and other commercial activities. Gir has become a terrestrial island surrounded by cultivation from all around.

2.2.1. Zonation

The Gir ecological Unit has been divided into following zones (Joshi and Karamchandani 1976).

Core Zone: 510.94 sq.km

Core Extension Zone: 901.19 sq. km

Eco-development Zone: 402.90 sq. km (Peripheral forests and 97 peripheral villages)

Satellitic Population Zone (Extended Gir): 308.27sq.km (Girnar, Mitiyala, Coastal forests)

Tourism Zone: Forest roads in Sasan, Dedakadi and Gir Interpretation Zone.

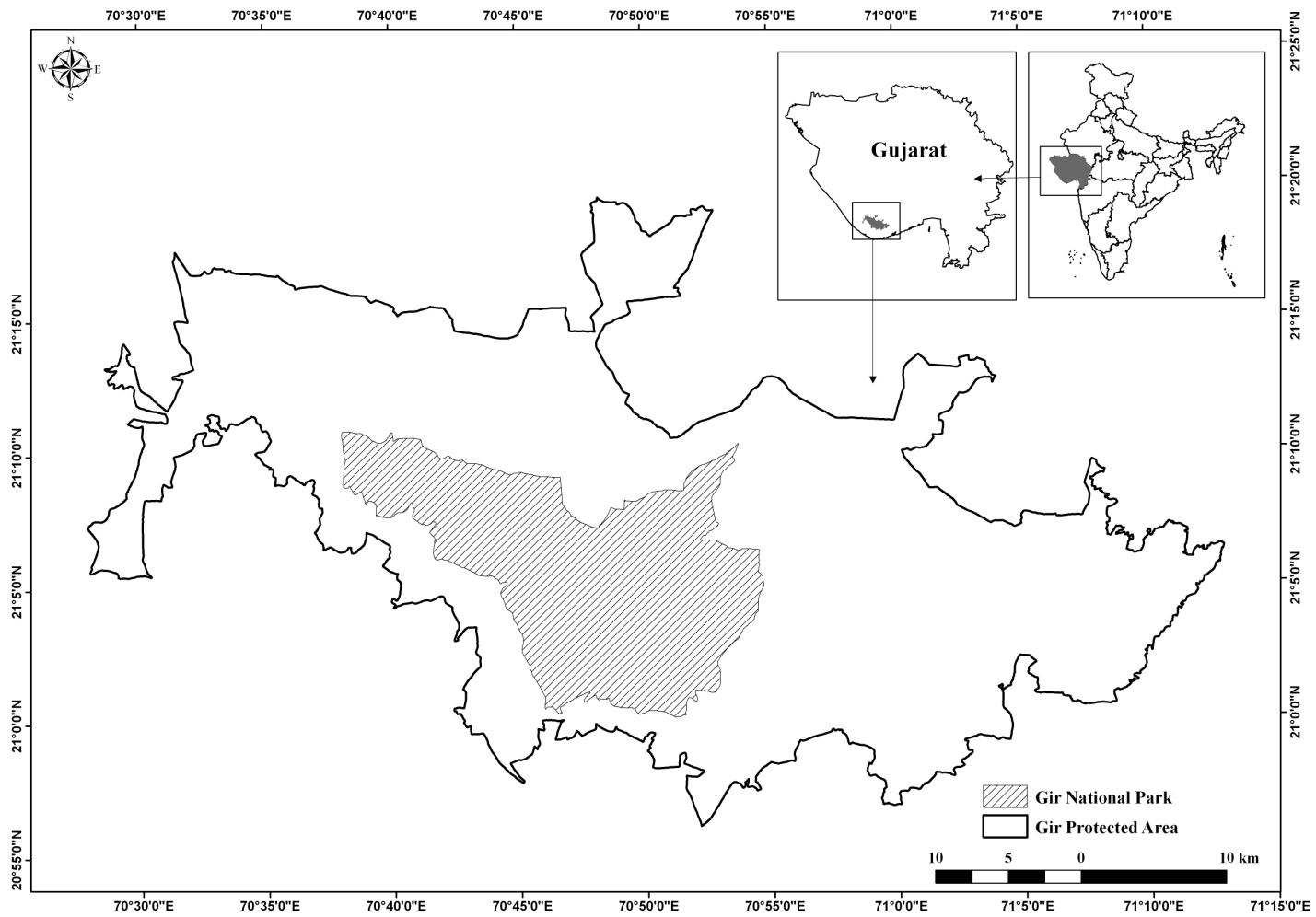


Figure 2.3: Gir Protected Area (PA). The map insets show the location of Gir PA within India and western state of Gujarat.

2.2.2. Administration

The Gir national park and sanctuary is divided into 16 ranges and 38 blocks (Figure 2.4) managed by three deputy conservator of forests (DCF), namely two territorial divisions DCF (west), DCF (east) and the DCF (wildlife) reporting under the conservator of forest (CF), Wildlife division.

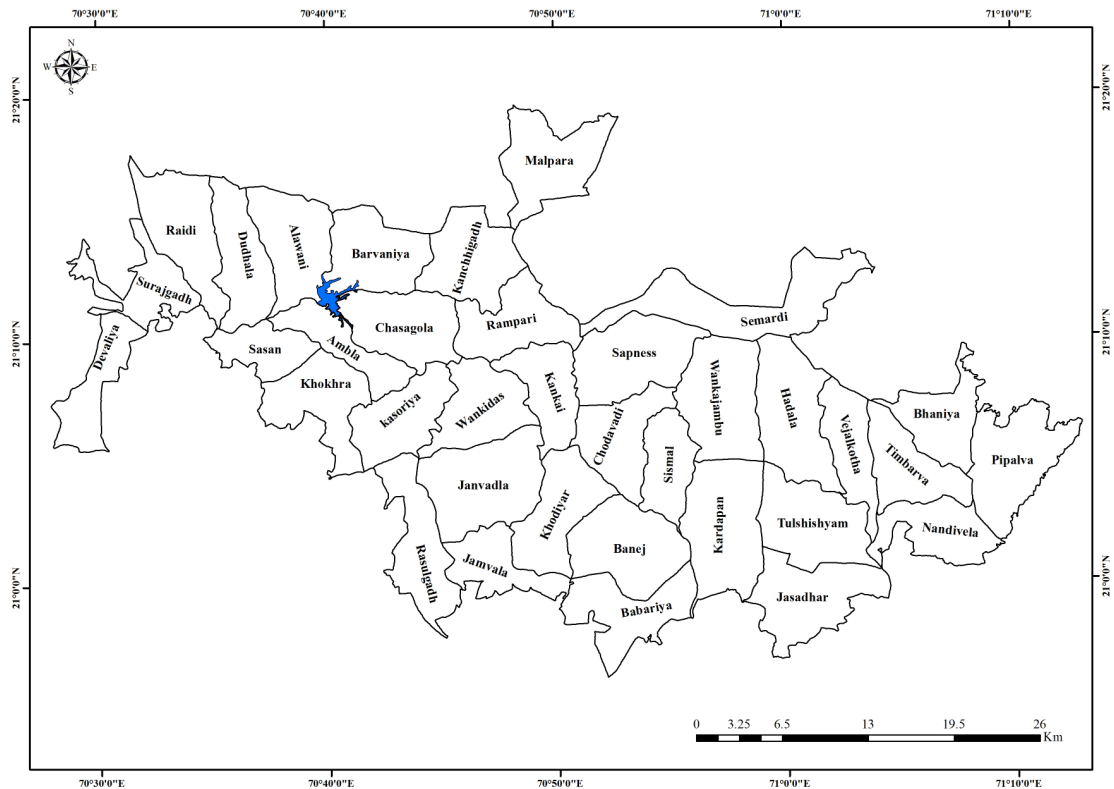


Figure 2.4: Administrative division of Gir PA into Range and Blocks
(Joshi and Karamchandani 1976)

2.2.3. History of Forest and Wildlife Management in the Area

Gir forests consist of forests of Ex. States of Junagadh and Baroda. Forests were mainly managed and functioned to get revenue and for shooting the wild animals. Additionally, the sale of timber and other forest produce, grazing fees were also charged. From period 1878-1914, revenue department managed the forests of Ex-Junagadh state and a permit holder was free to cut the timber in any manner he liked (Singh and Kamboj 1996). As a result of that, 3100 sq km of Gir Forest in 1878 (MacNabb and Enthoven 1879) shrunk to 1282 sq km by 1920 (Ratnagar 1920). Livestock grazing by local *Maldhari* grazers and surrounding villagers was also very

common. Large tracts of forest land were cleared for human settlements and brought under cultivation, resulting in the loss of approximately 1178 sq km of forest (Khan 1995).

From 1957-58 onwards the forests were controlled as per Acharya's Working Plan (Singh and Kamboj 1996), that divided the area into superior teak, inferior teak, improvement and grass working circles. Since 1920, management of forest has reduced the rate of decrease in size, but the area under forest reduced from both the forest clearances to provide agricultural land and the prevention of natural regeneration by heavy grazing (Hodd 1970). In 1965, 1251 sq km of the Gir area was declared as wildlife sanctuary under Government Notification No. GH.KH/97-WLP/660/62848-P, dated 19.09.1965 by Agriculture, Forests and Co-operation Department, Government of Gujarat and conservation efforts were accelerated and systematic studies were undertaken to understand the ecosystem. The land use pattern was studied within the lion's environment and it was found that the lions were distributed in Gir hills and avoided heavily cultivated surrounding lowlands (Joslin 1973). The causes of decline of biodiversity and lion population in Gir was revealed in 1976 (Berwick 1976). From 1976 to 1985, forests were managed as per Joshi's plan focusing on biodiversity conservation in favor of wildlife.

In 1983, Government of India determined to implement a comprehensive network of Protected Areas (PAs) as a keystone of the National Wildlife Action Plan, and in 1988 "Biogeographic Classification" of India was planned to facilitate conservation planning. Gir was specified within the "Semi-Arid Biogeographic Zone" and "4B Gujarat Rajputana" biotic province of India (Rodgers and Panwar 1988). In the report of the planning of the protected area network of India, extension of Gir National Park to 1000 sq km was strongly recommended including areas of the eastern grassland communities as well as the teak and mixed woodlands for providing adequate long term protection to the sole population of Asiatic Lion in the world (Rodgers and Panwar 1988). The 1991 Amendment to the wildlife (Protection) act strengthen wildlife conservation principally reducing the degree of large scale resource exploitation in wildlife sanctuaries. In 1996, the eco development project was conceived and monitoring scheme developed for Gir, differed slightly, being the only remaining habitat of the Asiatic Lion.

2.2.4. The Landscape Characteristics

24.14 km north of the Arabian Sea, Gir stretches 70.8 km from west to east. From north to south it is narrow ranging from 8 to 32 km wide. It lies 152 meter above sea-level, with ridges of low, steep sided hills rising to about 366 meter. There are few higher hills, the maximum, Nandivela, 530.65 meter, being near Jasadhar, at the eastern end of the Gir. The hills form the catchment area for four small perennial rivers. A majority of the area of Gir is occupied by level to gentle slopes (651.07 sq km, 46%), 315.90 sq.km (22%) area is occupied by moderate to moderately steep slopes and the remaining 445.16 sq. km. (31%) area exhibits steep to very steep slopes (Singh 2007) (Figure 2.5)

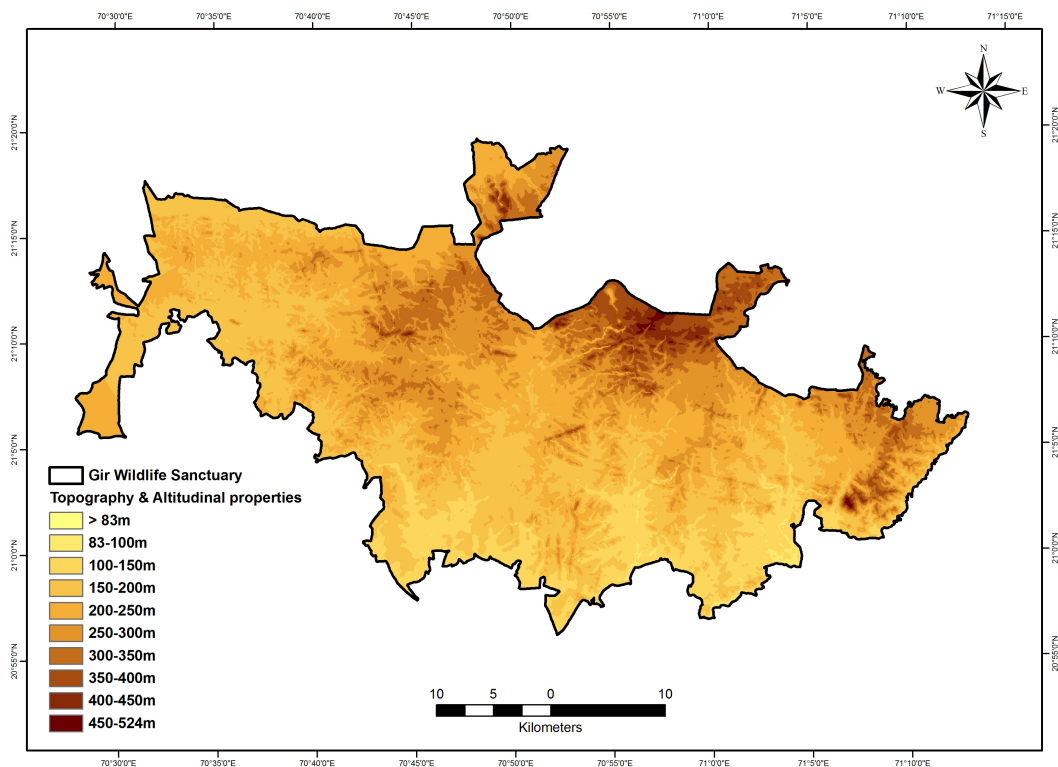


Figure 2.5: Topography and Altitudinal Profile of Gir Protected Area
(Source: Rodriguez et al. 2005)

2.2.5. Geology, Rock and Soil

Geomorphologically, the whole of the Gir PA comprising of flat topped hills, ridges, conical hills, radiating spurs, multiple scarps, pediments, pediplains and valley formed of a vast zone of dissection of lava flows (Patel 1992). Hills are of volcanic origin, of

which the chief geological formation is deccan trap. It occurs in two varieties of formation, one is acidic dyke and the other one is basic dyke. The prevailing rock is dolomite or basalt which is easily recognized by the occurrence of amygdales, with whitish or greenish tinges. Limestone is of common occurrence. Other rocks found scattered and sometimes occurring as a complex formation are gneiss, quartzite, quartz and feldspar.

Controlled by the geomorphological features, soil varies from place to place. On the hillsides, red soils prevail. Often they are very sandy and rocky with poor moisture retention properties. Where the hillside is less steep, the soil may be red or red brown clay. Black soil also known as “regur” or black cotton soil is found on gentle slopes with varying proportion of loam. Quite shallow, up to 2 metres deep and below the soil, there is often cemented, white layer, at least 30 cm thick, of “kankar”, a deposit of calcium carbonate. These black soils derived from the Deccan Trap, are very rich in montmorillonite clay, with the property of swelling up when wet and retaining moisture for long periods. They are base-rich, having pH 7 to 8. Alluvial soil is found on the plains and gentle slopes, particularly in the extreme southwest region of the Gir, deposited during the tertiary period, when the lower-lying regions of Kathiawar peninsula were under the sea. The alluvial soils are still rich in clay, probably kaolinite with a little montmorillonite. The water holding properties are not as good as those of black soil but better than those of the red soil. The soil is neutral or slightly acid, pH 6 to 7 (Pandit et al. 1992). Along the boundaries of the perennial rivers and their tributaries, silt has been deposited forming a sandy soil, rich in organic matter and grayish black in colour.

2.2.6. Climate

Situated in a semi-arid area where rainfall is erratic and irregularly distributed (Figure 2.6) ninety five percent of the rain falls during southwest monsoon. Throughout the twentieth century, droughts were frequent, however, during the last two decades; average rainfall has increased, possibly due to increased western monsoon in India (Singh and Gibson 2011). The average annual rainfall in Gir is highest in the western Gir and gradually decreases towards east. Monsoon extends from June to mid October

temperature ranging from 70°F to 90°F and relative humidity from 80% to almost 100%.

From late October to February there is cold, dry season, with cold north-east winds coming from the Himalayas and lowering the temperature. Temperature ranges from 40°F at night to 75°F in the day. From March to early June, the hot dry season prevails with the temperature ranging from 70°F to 110°F (Figure 2.7). In March-April the relative humidity may drop to 10% during the day, with the rise to 70% during May-early June. Average rainfall in Gir is estimated to be about 800mm/year with west Gir receiving higher rainfall (about 1000mm/year) than east Gir (about 600mm/year) (Sharma 1995)

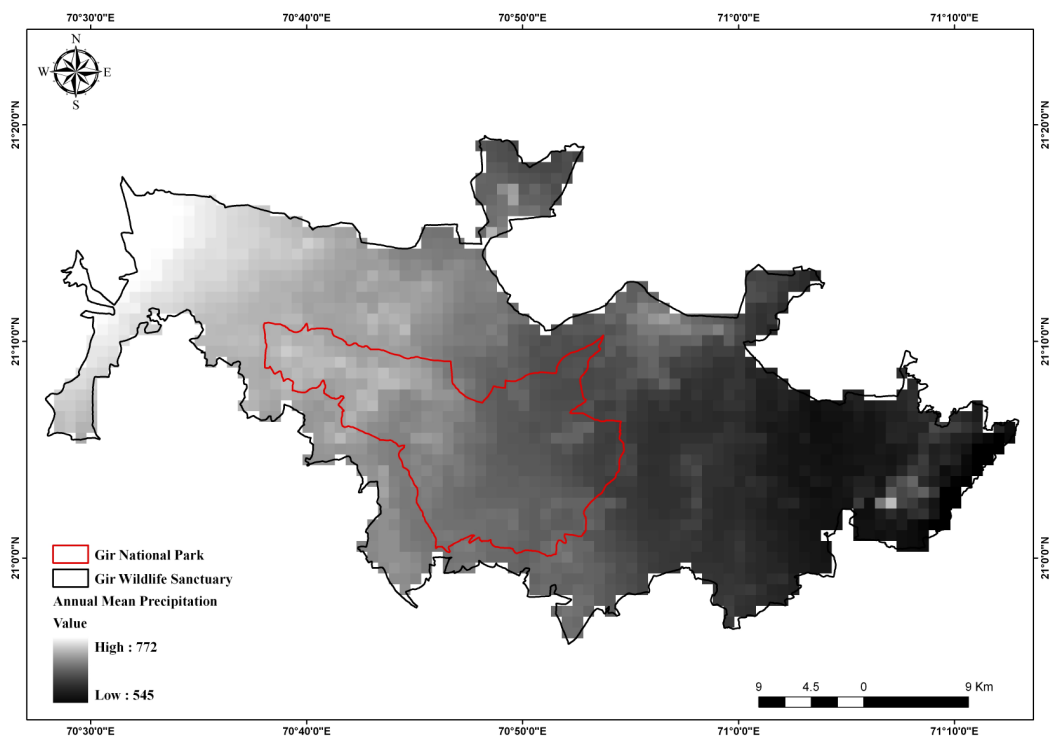


Figure 2.6: Annual Precipitation Profile of Gir Protected Area. The eastern region has lower rainfall compared to the western part of Gir
(Source:Hijmans et al. 2005)

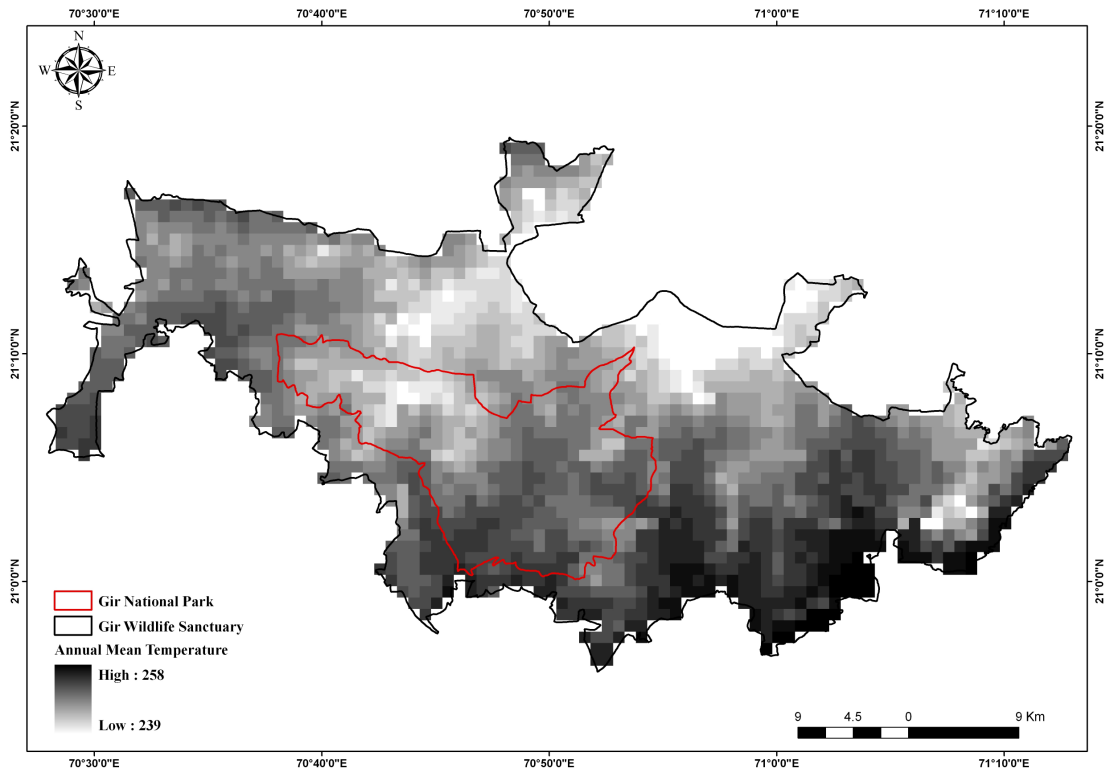


Figure 2.7: Annual Mean Temperature Profile of Gir Protected Area

(Source:Hijmans et al. 2005)

2.2.7. Rivers and other water supply points

The entire area is divided into the catchments of rivers Hiran, Shingoda, Shetrunji, Macchundari, Raval, Malan, Datardi, and Popatdi (Figure 2.8). Because of the uneven topography and structural variations, streams in the hilly terrain flow in all directions creating a radial drainage pattern. These rivers have perennial water supply, except in severe famine years. Except Shetrunji, all rivers flow from north to south tract is cut up by innumerable streams.

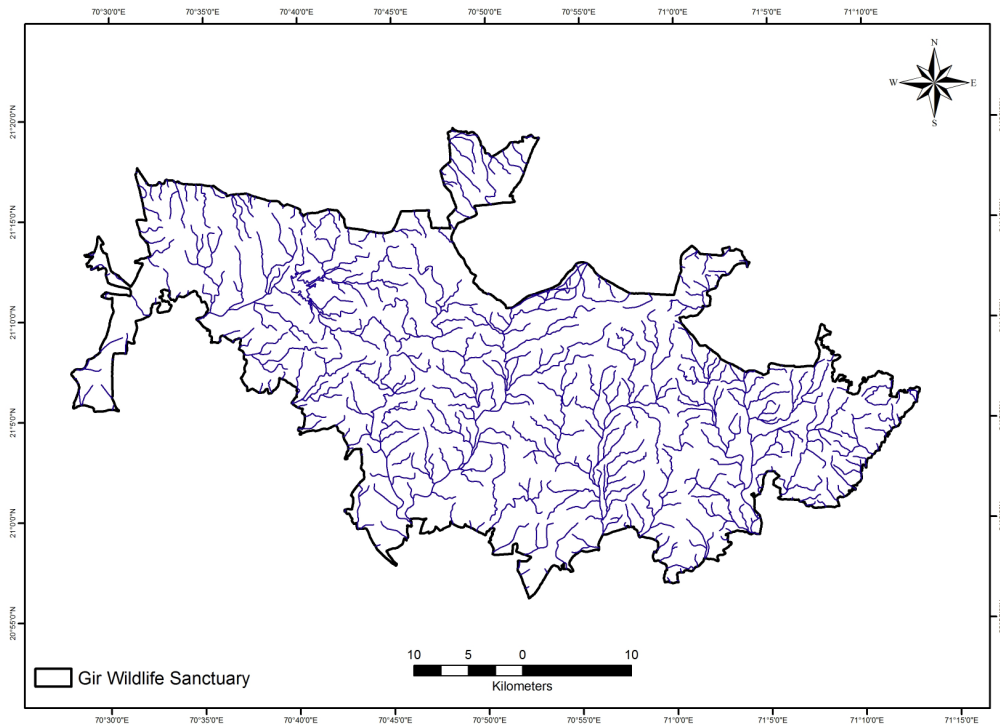


Figure 2.8: Drainage system of Gir protected area
 (Source: Digitized on Survey of India Toposheet)

Topography and terrain impact the water regime by contributing towards run off and recharge to the ground water. Water table is usually upto 6 to 10 meters and water supply has improved in many places due to construction of four dams in Gir. In addition to seven rivers and four dams, about 350 artificial water holes are also maintained by forest department from March to the onset of monsoon every year (Figure 2.9).

2.2.8. *Maldharis* and Forest Settlement Villages

The *maldharis* (agro-pastoral community) and forest settlement villages constitute the integral part of Gir. During 1971, there were 129 *maldhari* nesses with 845 families dotted all over Gir forests but with the enactment of *maldhari* resettlement scheme, 592 families have been moved and resettled. At present there are 253 families residing in 54 *nesses* in Gir Forest. The *ness* occupies about half hectare of ground, nearby a water source. About a half a dozen to a dozen huts made of mud and timber stand within the compound. Each hut is having a small wooden courtyard, a gate, and a heap of firewood. The compound fence is high and thick enough to keep the lions

away at night, but some manage to break or cross the barricades (Singh 2007). Distribution of the existing nesses is shown in figure 2.10.

There are also 14 forest settlements covering an area of 51.76 sq km. within the Gir Forest. Out of these, 18.67 sq km. has been given for cultivation and rest for other than cultivation. These settlement villages form part of the sanctuary. *Siddis*, a primitive tribe whose origins are in Africa, lives in Sirvan, one of these settlement villages.

There are 97 revenue villages within a periphery of 6 km. from sanctuary. Primary occupation is farming with major crops in the region *viz.* wheat, cotton, maize, millet etc. and orchards of Kesar mangoes (Vijayan and Pati 2001). The 917 households of the *maldharis* and settlers are solely dependent on Gir forests for their necessities, whereas the 97 adjoining villagers are partially dependent on PA (Srivastava 1997).

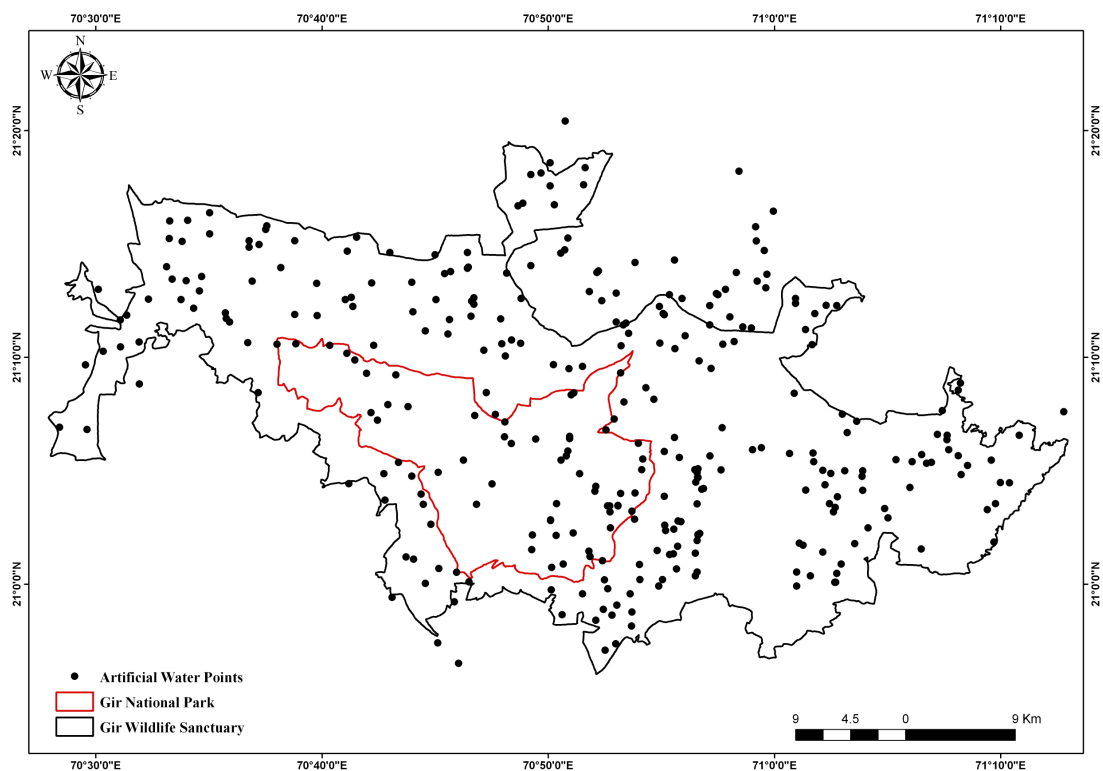


Figure 2.9: Artificial water holes distributed within the Gir protected area

(Source: GPS points from Gujarat forest Dept.)

From the forestry point of view, the growing stock may be largely classified into the following sub types:

- (i) Teak forests : (Type 5A/C1a)
- (ii) Non-teak forests : (Type 5/DS1 & 5/DS2)

Teak although poor in size and quality, accounts for over 50 percent of the tree stands. Babul (*Acacia nilotica*) is also abundant and probably accounts for about 25 percent of the total tree growth. Other species include : sadad (*Terminalia alata*), behda (*Terminalia bellirica*), tendu or timru (*Diospyros melanoxylon*), haldu (*Haldina cordifolia*), sissam (*Dalbergia sissoo*), khair (*Acacia catechu*), karanj (*Derris indica*), siris (*Albizia lebbek*), krangsa (*Albizia procera*), mahuda (*Madhuca indica*), amla (*Phyllanthus emblica*), aritha (*Sapindus emarginata*), garmala (*Cassia fistula*), jamun (*Syzigium cumini*), khakra (*Butea monosperma*), kudi (*Wrightia tinctoria*), aal (*Morinda pubescens*), salie (*Boswellia serrata*) and some patches of bamboo (*Dendrocalamus strictus*). There are few climbers, but thorny bushes or shrubs are commonly intermixed with the trees. These consist primarily of *Acacia* sp., ber (*Zizyphus mauratiana*), guggal (*Commiphora wightii*) and so forth

The trees in the Gir, with very few exceptions, lose their leaves during the dry season (Spillet 1968); stunted in nature, probably because rock is everywhere near to the surface. The undergrowth is seldom dense so that the grazing of cattle and buffaloes can take place throughout much of the forest. Open grassy glades exist near its edges, patches of cultivation (Wynter-Blyth 1949).

2.2.10 Fauna

The Gir forest has a diverse assemblage of wildlife harboring 32 species of mammals, around 300 species of birds, 26 species of reptiles and more than 2000 species of insects (Singh and Kamboj 1996).

The main carnivores of the Gir are the Asiatic lion (*Panthera leo persica*), leopard (*Panthera pardus*), jungle cat (*Felis chaus*), rusty-spotted cat (*Prionailurus rubiginosus*), hyaena (*Hyaena hyaena*), jackal (*Canis aureus*), mongoose (*Herpestes edwardsi*), civet cat (*Viverricula indica*), fox (*Vulpes bengalensis*), ratel (*Mellivora*

capensis), and desert cat (*Felis libyca*). Chital (*Axis axis*), nilgai (*Boselaphus tragocamelus*), sambar (*Rusa unicolor*), four horned antelope (*Tetracerus quadricornis*), and chinkara (*Gazella gazella*) constitute the major wild herbivores of the Gir. Wild pig (*Sus scrofa*) also depends largely upon vegetation and roots of grasses. Amongst the smaller mammals, common langur (*Presbytis entellus*), porcupine (*Hystrix indica*) and hare (*Lepus nigricollis*) are common but pangolin (*Manis crassicaudata*) is rare.

The reptilian fauna is represented by crocodile (*Crocodylus palustris*), the star tortoise (*Geochilon elegans*), the monitor lizard (*Varanus bengalensis*), python (*Python molurus*) and number of snakes.

There are more than 300 bird species listed in Gir, out of which some are migratory but most are resident. The main scavenger bird is vulture of which about six species are recorded. In addition to peafowl, other ground birds commonly found are grey quail (*Coturnix coturnix*), the jungle bushquail (*Perdica asiatica*), the grey partridge (*Francolinus pondicerianus*), and nightjar (*Caprimuggus asiaticus*). Snake bird (*Anhina rufa*), black ibis (*Pseudhis popillosa*), grey heron (*Aardea cineria*), little egret (*Egretta garzetta*), paddy bird (*Ardeola gravil*), cattle egret (*Bubulcus ibis*), the Indian moorhen (*Gullinula ehlorpos*) are amongst the water birds found in Gir. The main predator birds of Gir are the great horned owl (*Bubo bubo*), the spotted owlet (*Athene brama*), the tawny eagle (*Aquila rapax*), shikara (*Accipiter badius*), brahminy kite (*Heliastuz indus*), the black winged kite (*Elanus caeruleus*) etc.

Landscape architecture of Gir Protected Area and its surroundings, impact of spatial scale in understanding landscape configuration

3.1 Introduction

Spatial pattern of habitat governs the persistence of natural populations (Harrison et al. 1988, Sjögren 1991, Verboom et al. 1991, Dunning et al. 1995, Villard et al. 1995, Thomas and Hanski 1997, Hanski 1999, Thomas and Kunin 1999, Vos et al. 2000), affects the distribution of the dispersing individuals to patches (Dunning et al. 1995, Matthysen and Currie 1996, Schumaker 1996, Sutcliffe and Thomas 1996, Vos et al. 2002). Hence it is important in constructing ecological communities (Levins 1976), may impact the spread of disturbance (Romme and Knight 1982, Franklin and Forman 1987, Turner 1987a), the distribution and persistence of populations (Van Dorp and Opdam 1987, Fahrig and Paloheimo 1988), large herbivore foraging (Senft et al. 1987), the horizontal flow of materials such as sediments or nutrients (Peterjohn and Correll 1984, Ryszkowski and Kedziora 1987), and other ecologically important processes such as net primary production (Turner et al. 1987b, Sala et al. 1988).

Differences in climate, edaphic factors, resource distribution and physical disturbances are natural factors that regulate landscape patterns (Wiens et al. 1985).

The extensive use of remotely sensed data to monitor environmental change and to map land cover has increased in last 20 years (Booth 1989, Lillesand and Kiefer 1994) but adaptations of arid vegetation and near identical spectral responses across different vegetation types confounded conventional classification techniques (Ray 1995). Spectral response patterns alone often prove insufficient in heterogeneous, arid environments. Usage of aerial photographs for forestry applications has already been well established (Tiwari 1978, Madhavanunni 1983). But more recently both visual and digital techniques on satellite data have been tried under numerous projects to separate different vegetation types and their density classes (Kachwaha 1983, Yool et

al. 1986, Sudhakar 1991a, 1991b, 1992a, 1992b, Madhavanunni 1990, Rao 1990, Roy et al. 1990, 1991, Madhavanunni et al. 1991).

Profuse literature is now available demonstrating the value of Advanced Very High Resolution Radiometry (AVHRR), Multi Spectral Scanning (MSS) and Thematic Mapping (TM) data for mapping vegetation and land cover types (Hobbs et al. 1989, Lucas et al. 1993, García and Alvarez 1994, Fuller et al. 1998, Tuomisto 1998, Rodríguez- Yi et al. 2000). Several studies established the effectiveness of using satellite derived data in generating land use/cover maps as well as detecting landscape change over time (Sabins 1997, Abou El-Magd and Tanton 2003, Cardille and Foley 2003, Lobo et al. 2004). Certain studies used supervised and unsupervised multi-spectral classifications, as well as a combination of the two approaches, to classify vegetation types (O'Neill 1989). Other authors have used multi-temporal imagery to improve the accuracy of their classification (Richards 1984, Franklin 1991).

Patterns of plants and animals distributions often are influenced by different processes at different spatio-temporal scales resulting in variable patterns when observed across multiple scales (O'Neill et al. 1986, Carlile et al. 1989, Wiens 1989, Menge and Olson 1990). The array of spatial and temporal scales at which ecological problems are posed has expanded dramatically in recent years, and the need to study scale in ecological analyses has often been noted (Allen and Starr 1982, Delcourt et al. 1983, O'Neill et al 1986, Addicott et al. 1987, Getis and Franklin 1987, Meentemeyer and Box 1987, Morris 1987). No single scale exists to completely describe population, community, and landscape patterns (Greig-Smith 1983, Wiens 1989, Levin 1992), parameters and processes significant at one scale are often not important or predictive at another scale, and information is sometimes lost as spatial data are measured at coarser scales of resolution (Henderson-Sellers et al. 1985, Meentemeyer and Box 1987). Thus analyses of multiple scales are required to precisely describe the relationship between organisms and their environment (Sugihara and May 1990, Glenn et al. 1992, Fuhlendorf and Smeins 1996).

In explaining ecological problems, the necessity of the extrapolation of fine scale measurements for the analysis of broad scale phenomena and failure to account for scale-dependent patterns has confused and confounded ecological synthesis and led to

many inappropriate extrapolations of research results (Wiens 1989, Costanza and Maxwell 1994, Gardner 1998). Therefore the progress in methods that will preserve information across scales or compute the loss of information with altering scales has become a critical task. Such methods are essential before ecological understandings can be extrapolated between spatial and temporal scales. Both theoretical and empirical approaches have confirmed relationships between spatial scale and spatial pattern (Wiens 1989, Turner 1990, Costanza and Maxwell 1994) and to better understand, manage, and forecast the behavior of the complex systems that provide life on earth, we need an improved understanding of the scale-specific interactions responsible for landscape metabolism (Levin 1992), robust techniques for visualizing and interpreting multi-scale processes from patterns (Turner et al. 1991), and appropriate scaling strategies for linking and modeling data at multiple scales (King 1990, Ehleringer and Field 1993).

Several studies have inspected how metric values alter with data resolution and aggregation (Benson and MacKenzie 1995, Moody and Woodcock 1995, Wickham and Riitters 1995, Jelinski and Wu 1996, Saura 2002, 2004, García-Gigorro and Saura 2005) or spatial extent of the study area (Turner et al. 1989). Saura and Martinez-Millan 2001 simulated landscapes with multiple configurations and fragmentation levels over different spatial extents and established that the sensitivity of most metrics tended to increase with aggregation. Further research results have varied the pixel size and found that land cover classes were less articulated or lost altogether when pixel size was increased (Turner et al. 1989, He et al. 2002). Scaling relations for metrics measured over a range of pixel sizes and extents was inspected more recently, where some displayed simple scaling functions whilst others had random behavior (Wu et al. 2002, Wu 2004, Shen et al. 2004). A distinct sensitivity below 16-type classes was observed as the effect of spatial extent, spatial resolution, and thematic resolution on landscape patterns (Baldwin et al. 2004).

In recent years, the importance of scale effects on spatial analysis and modeling has been progressively emphasized in light of spatial heterogeneity and hierarchy theory (Allen and Starr 1982, Meentemeyer and Box 1987, Morris 1987, Turner et al. 1989, Levin 1992, Costanza and Maxwell 1994, Wu et al. 1994, Wu and Levin 1994). It has been broadly accepted that spatial pattern is scale-dependent; that is, it changes

with the scale of observation or analysis (Gardner et al. 1987, Meentemeyer and Box 1987, Woodcock and Strahler 1987, Turner et al. 1989, O'Neill et al. 1991, 1996, He and Legendre 1994, Moody and Woodcock 1995, Jelinski and Wu 1996, Qi and Wu 1996, Gardner 1998, Wu et al. 2000). Numerous current studies, reporting several scale effects, have shed light on the problem of scale in pattern examination. Apparently, variables that describe landscape pattern, such as the number, area, and spatial pattern of different patch types, will change when scale (grain size and/or extent) is changed (Wiens 1989, Wu 1999). Scale effect is caused not only by changing grain size or extent itself, but also by associated changes in landscape composition (e.g. Diversity of patch types) and configuration (e.g., spatial arrangement of different patch types).

Theoretically, scale means the “Window of perception”, the filter or the measuring tool with which a system is observed and computed. An important characteristic of scale lies in the difference between grain and extent. Grain refers to the smallest intervals in an observation set, while extent refers to the range over which observations at a particular grain are made (O'Neill and King 1998). Within a remote sensing context, grain is equal to the spatial resolutions of the pixels composing an image, while extent signifies the total area that an image covers. Spatial resolution affects the accuracy at which land covers are mapped as well (Moody and Woodcock 1995).

Scale effects have long been studied in human geography as part of the modifiable areal unit problem or MAUP - the problem in spatial analysis that occurs when area-based data are aggregated (Openshaw 1984, Arbia et al. 1996, Jelinski and Wu 1996, Wrigley et al. 1996, Marceau 1999). MAUP covers two distinct but related aspects: the result of statistical analysis is affected by both the level of data aggregation or grain size (so-called “scale problem”) and by alternative ways of aggregating pixels at a given grain size (often called the “zoning problem” or “aggregation problem”). MAUP has often been discussed together with the so-called “ecological fallacy” (Robinson 1950) which refers to inappropriate extrapolation of statistical relationships from one scale to another. A more suitable term for this kind of scale-related problems maybe “spatial transmutation” (O'Neill 1979, Wu and Levin 1994).

The modifiable areal unit problem includes both the effect of altered grain size and the mode of this alteration. Likewise, there are also different ways of changing extent: e.g., boxing out from the center of a map or starting from one corner along a diagonal direction. Overall, much more research has been done into the effects of changing grain size (particularly in the context of MAUP) than those of changing extent, and a quantitative understanding of these two types of scale properties across different systems and methods is still lacking. Though the importance of MAUP has formerly been noted in landscape ecology (Jelinski and Wu 1996), its relationship to remote sensing data remained poorly documented and understood (Wu et al. 2000). In particular, the effects of MAUP can be especially devastating during scaling, where randomly extrapolating site-specific measurements to coarser scales can result in substantial error (Gardner et al. 1982, King 1990). Thus the difficulties for inappropriately using remote sensing data to understand multiscale landscape patterns/processes are profound.

3.2 Previous Vegetation Studies in Gir

The Gir ecosystem falls in the biogeographic zone 4 (Rodgers and Panwar 1988) (semi-arid) (Figure 3.1) and biographic province 4-B (Rodgers and Panwar 1988) (Figure 3.2) Gujarat Rajwara of India. The semi-arid zone in numerous ways is a transition from the western arid zone to the moister areas of the peninsula to the east. As a transition, it is fairly heterogeneous. The Gujarat-Rajwara biotic province that covers all of Kathiawar in Gujarat is typically different being drier largely *Anogeissus pendula* forests than the moist teak forests. In many parts of the world, semi-arid regions with their dominant grass and palatable shrub layers achieved some of the highest wildlife biomasses and Gir is not an exception. It harbors diverse variety of flora and fauna which comprise numerous endangered species and delivers breeding ground for numerous migratory as well as resident birds. The semi-arid zone of India has strong biological associations with western Asia (Pakistan, Iran and Middle East) and northern Africa. Many of the plants are of African affinity: *Acacia*, *Anogeissus*, *Balanites*, *Capparis*, *Grewia* etc. The Kathiawar peninsula of Gujarat is lower and has black cotton soil. French Institute of Pondicherry classified the vegetation of this region as *Acacia-Capparis* scrubland (Gausson et al. 1968).

Dry deciduous forest of Gir has been divided into 13 sub-types ranging from riverine forest to scrub forest. The grasslands within and outside the protected area provide valuable fodder to the *panjrapoles*, *goshalas*, village *panchayats* and individuals (Srivastava 1997).

According to Champion and Seth's classification of Forest types, 1964, these areas fall under the type 5A/C1a, i.e. very dry teak forests with rainfall less than 900 mm. and dry shallow soil. Teak occurs mixed with dry deciduous species. Due to heavy grazing and annual fires, ground cover is insufficient; the degradation stages of these subtypes are also met with in this tract under:

- (i) Sub-type 5/DS1: Dry deciduous scrub forests
- (ii) Sub-type 5/DS2: Dry savannah Forests (locally known as *vidis*)

From the forestry point of view, the growing stock may be broadly classified into the following sub types:

- (i) Teak forests : (Type 5A/C1a)
- (ii) Non-teak forests : (Type 5/DS1 and 5/DS2)

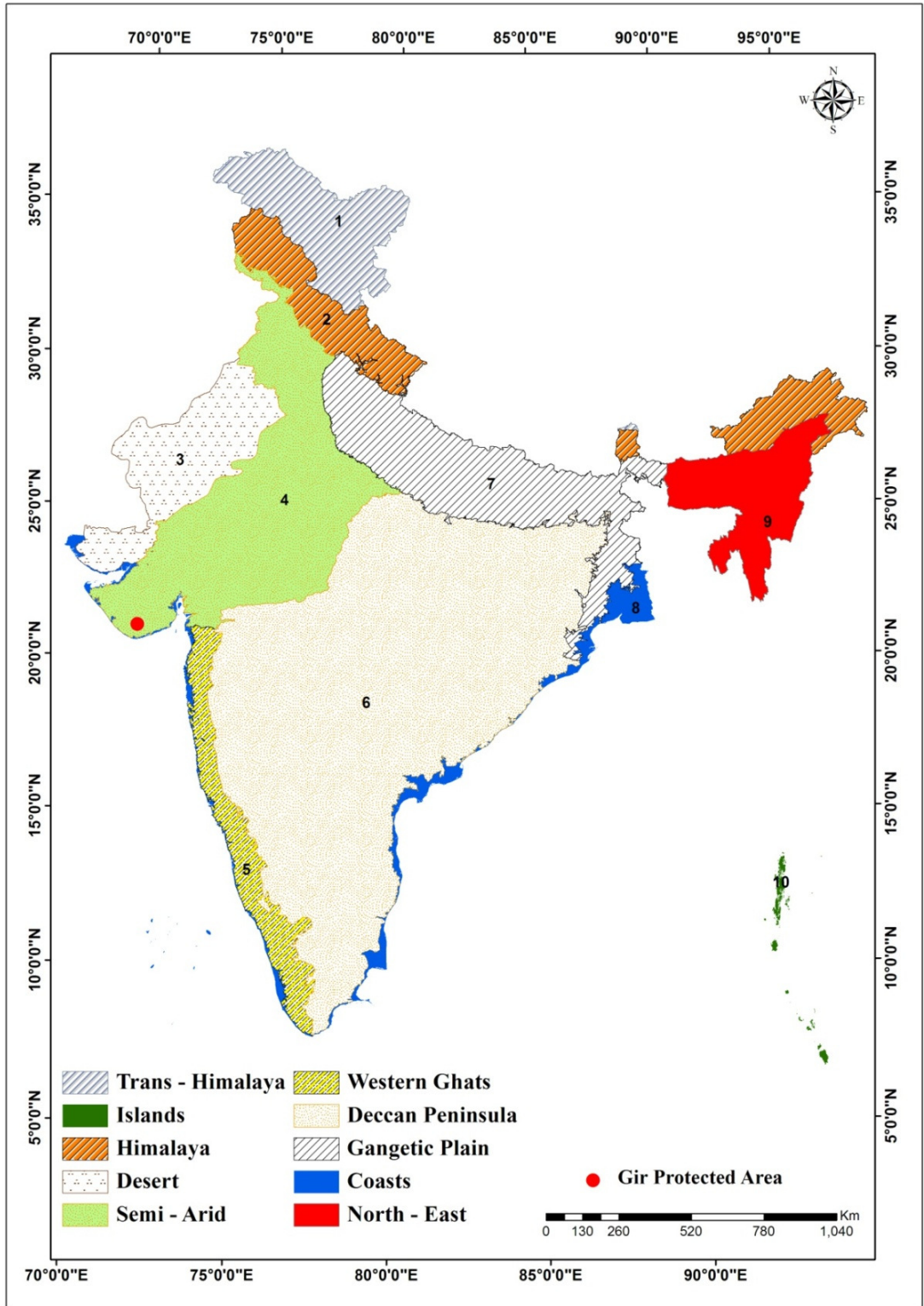


Figure 3.1: Bio geographic zone map of India

(Source: Rodgers and Panwar 1988)

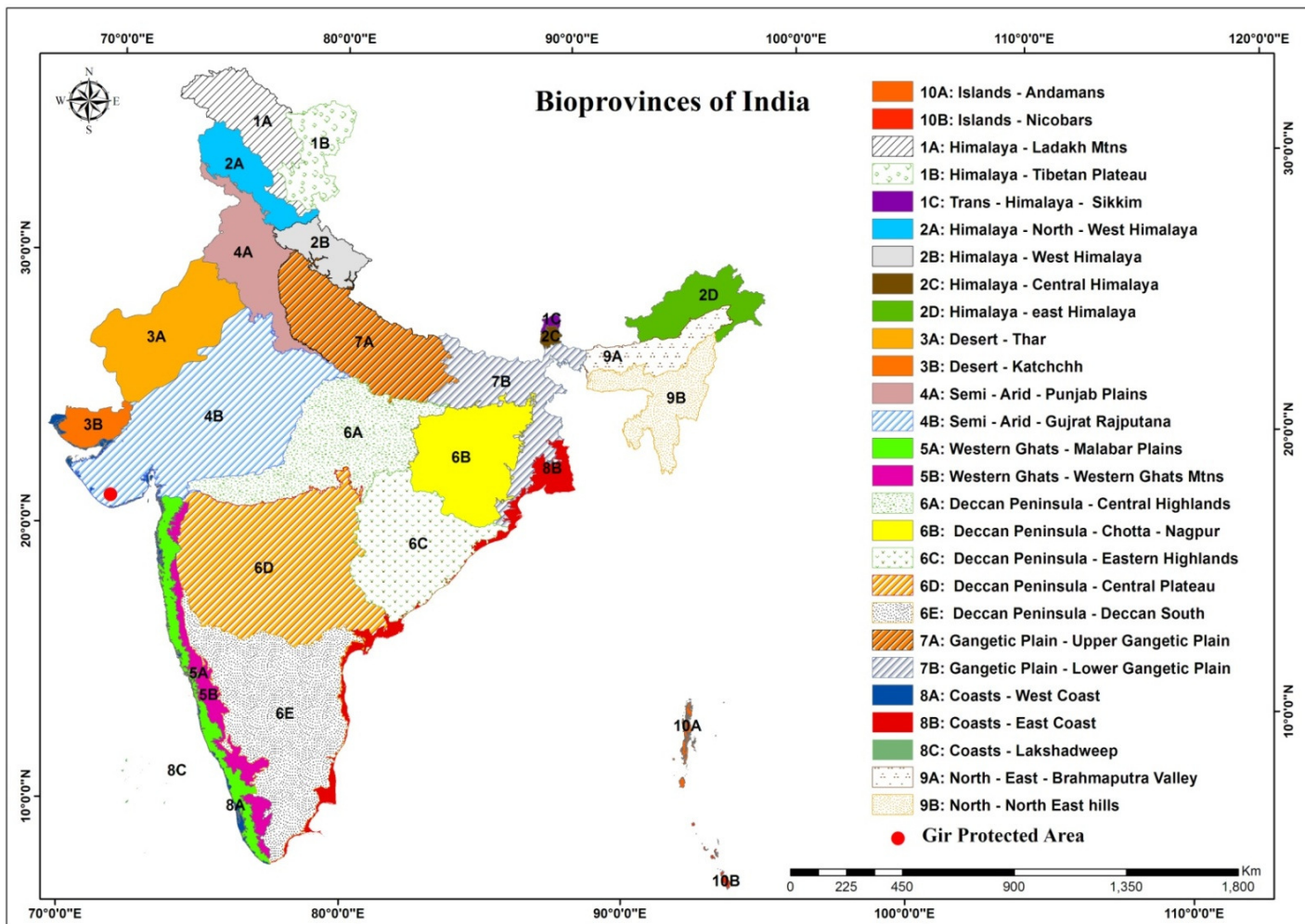


Figure 3.2: Biographic Provinces of India

(Source: Rodgers and Panwar 1988)

Habitat of Gir has been studied in past, but in diverse contexts and using different methods like studies with vegetation component (Berwick 1974, Khan et al. 1990). Berwick classified habitat types subjectively, the first effort to classify the habitat/vegetation. Earlier this, the work done was restricted to listing the flora (Santapau and Raizada 1956). The habitat of Gir was classified using quantitative analytical techniques (TWINSPAN programme) (Khan et al. 1990). They increased the number of habitat types to 11 as against 4 by Berwick. The study categorized 11 broad habitat types with distinctive tree species described in table 3.1 but identification of a tree and/or shrub was based on its height and not on its life form. Hence larger individuals of the same species were categorized as trees and smaller individuals as shrub based on height. Later the Gir forest was broadly divided into two major vegetation types (Chellam 1993). The western two-thirds of the forest were found dominated by stunted teak, in association with various other species, chiefly *Acacia*, *Zizyphus* and *Terminalia*. The remaining eastern portion consisted of a grassland savannah interspersed with patches of thorn (*Acacia* and *Zizyphus*) and dry deciduous forests (*Anogeissus latifolia* and *Boswellia serrata*), teak was absent in this region. The understorey was often sparse in the more densely wooded areas but shrubs like *Carissa opaca*, *Capparis sepriaria*, and *Helicteres isora* along with various grass species dominated the understorey in other areas. Evergreen riverine forests were found along the banks of most of the rivers and streams. *Syzygium rubicundum*, *Derris indica*, *Manilkara hexandra* and various other species were found in the riverine tracts. The eleven broad categories classified are mentioned in Table 3.1. Various forest types of Gir, were classified under 13 subtypes (Chavan 1993) and later in fifteen major vegetation associations (Sharma 1995) (Table 3.1). Most recently the forest was classified into three major classes (Qureshi and Shah 2004) which is described in table 3.1.

Table 3.1: Previous vegetation studies in Gir

Khan et al.1990		Chellam 1993	Chavan 1993	Sharma 1995	Qureshi and Shah 2004		
Classes	Characteristic species	Classes	Classes	Classes	Classes	Characteristic species	
(1) Riverine Woodland	<i>S. rubicunda</i> , <i>P.pinnata</i> , <i>Manilkara hexandra</i> , <i>Ficus bengalensis</i>	(1) Riverine forest,	(1) Dry Teak Forest (5A/C1b)	(1) <i>Acacia catechu</i> - <i>Zizyphus numularia</i> - <i>Aristida adscensionis</i> association,	Moist mixed vegetation	Moist mixed forest	Dominant species: <i>Tectona grandis</i> replaced by <i>Anogeissus latifolia</i> . In Gir (east)
(2) Thorn woodland	<i>Acacia nilotica</i> , <i>Acacia catechu</i> , <i>Zizyphus mauritiana</i> , <i>Acacia senegal</i>	(2) Teak Forest	(2) Southern Dry Mixed Deciduous Forest (5A/C3)	(2) <i>Apluda mutica</i> - <i>Themeda quadrivalvis</i> - <i>Sehima nervosum</i> association,			Associates: <i>Acacia spp.</i> , <i>Wrightia tinctoria</i> , <i>syzigium spp.</i> , <i>Mitragyna parviflora</i> , <i>Bauhinia racemosa</i> , <i>Diospyros melanoxylon</i> and <i>Emblica officinalis</i> .
(3) Teak-Acacia- <i>Zizyphus</i> woodland	<i>Tectona grandis</i> , <i>Acacia senegal</i> , <i>Zizyphus mauritiana</i> , <i>Acacia nilotica</i>	(3) <i>Acacia</i> - <i>Zizyphus</i> woodland,	(3) Northern Dry Mixed Deciduous Forest (5B/C2)	(3) <i>Anogeissus latifolia</i> - <i>Acacia spp</i> - <i>Terminalia crenulata</i> association			Understorey: <i>Acacia spp.</i> , <i>Zizyphus spp.</i> , <i>Grewia tiliaefolia</i> , <i>Helecteres isora</i> , <i>Carissa carandas</i> , <i>Manilkara hexandra</i> and <i>ixora arborea</i> .
(4) Mixed teak woodland	<i>Tectona grandis</i> , <i>Terminalia crenulata</i> , <i>Acacia catechu</i> , <i>Lannea coromandelica</i> , <i>Boswellia serrata</i>	(4) Mixed Forest	(4) Dry Deciduous Scrub (5/DS1)	(4) <i>Anogeissus latifolia</i> - <i>Acacia catechu</i> -association		Mixed forest	Dominant species: <i>Tectona grandis</i> replaced by <i>Anogeissus latifolia</i> in Gir (east)
(5) Mixed valley community	<i>Tectona grandis</i> , <i>Pterocarpus marsupium</i> , <i>Schrebera swietenoides</i>	(5) <i>Acacia</i> Woodland	(5) Dry Savannah Forest (5/DS2)	(5) <i>Acacia spp</i> - <i>Zizyphus mauritiana</i> association			Associates: <i>Diospyros melanoxylon</i> , <i>Garuga pinnata</i> , <i>Gmelina arborea</i> and <i>Mallotus phillipensis</i> .

Khan et al.1990		Chellam 1993	Chavan 1993	Sharma 1995	Qureshi and Shah 2004		
Classes	Characteristic species	Classes	Classes	Classes	Classes	Characteristic species	
(6) Teak-Boswellia-Sterculia woodland	<i>Tectona grandis</i> , <i>Boswellia serrata</i> , <i>Sterculia urens</i> , <i>Lanneacoromandelica</i> , <i>Acacia catechu</i>	(6) Scrubland	(6) Dry Grasslands	(6) <i>Zizyphus mauritiana</i> association	Thorn forest	Understorey: <i>Acacia spp.</i> , <i>Zizyphus spp.</i> , <i>Wrightia tinctora</i> , <i>Grewia tiliifolia</i> , <i>Helecteres isora</i> , <i>Carissa carandas</i> , <i>Manilkara hexandra</i> and <i>Capparis sepiaria</i> .	
(7) <i>Anogeissus-Boswellia-Lannea</i> woodland	<i>Anogeissus latifolia</i> , <i>Boswellia serrata</i> , <i>Sterculia urens</i> , <i>Lannea coromandelica</i> , <i>Acacia catechu</i>	(7) Teak <i>Acacia-Zizyphus</i> woodland	(7) <i>Boswellia serrata</i> forest (5/B2)	(7) <i>Acacia nilotica-Zizyphus mauritiana</i> association		Teak – Acacia-Zizyphus	Dominant species: <i>Tectona grandis</i> replaced by <i>Anogeissus latifolia</i> . In Gir (east), <i>Acacia catechu</i> , <i>Zizyphus mauritiana</i>
(8) <i>Anogeissus-Terminalia</i> woodland	<i>Anogeissus latifolia</i> , <i>Acacia catechu</i> , <i>Terminalia crenulata</i> , <i>Butea monosperma</i> ,	(8) Teak-Mixed Forest,	(8) Babul (<i>Acacia nilotica</i>) Forest (5/B3)	(8) <i>Tectona grandis-Acacia catechu-Zizyphus mauritiana</i> association			Associates: <i>Acacia spp.</i> , <i>Zizyphus spp.</i> , <i>Terminalia spp.</i> ,
(9) Pure Teak woodland	<i>Tectona grandis</i>	(9) Teak- <i>Acacia</i> Woodland	(9) <i>Butea (Butea monosperma)</i> Forest (5/B5)	(9) <i>Tectona grandis-Acacia catechu-Terminalia crenulata</i> association			Understorey: <i>Zizyphus spp.</i> , <i>Carissa carandas</i> and <i>Capparis sepiaria</i>
(10) Thorn savannah	<i>Acacia and Zizyphus sp</i> , <i>Dichrostachys cineria</i>	(10) Open land	(10) Dry Tropical Riverine forest (5/1S1)	(10) <i>Tectona grandis</i> association			
(11) Thorn Bushland	<i>Zizyphus sp</i> , <i>Dichrostachyscineria</i>	(11) Agricultural land.	(11) Southern Thorn Scrub (6A/C2/DS1)	(11) <i>Tectona grandis-Acacia catechu-Lannea coromandelica-Boswellia serrata</i> association,		Acacia – Zizyphus	Dominant species: <i>Acacia spp.</i> , <i>Zizyphus spp.</i> ,

Khan et al.1990		Chellam 1993	Chavan 1993	Sharma 1995	Qureshi and Shah 2004		
Classes	Characteristic species	Classes	Classes	Classes	Classes	Characteristic species	
			(12) Desert Thorn Scrub (6B/C1)	(12) <i>Tectona grandis</i> - <i>Acacia</i> spp- <i>Wrightia tinctoria</i> association			Understorey: <i>Carissa carandas</i> and <i>Capparis sepiaria</i>
			(13) Tropical Euphorbia Scrub (6B/DS2)	(13) <i>Tectona grandis</i> mixed association		Scrubland	Dominant species: <i>Acacia catechu</i> , <i>Acacia leucohloea</i> , <i>Zizuphus numularia</i>
				(14) Mixed association			Associates: <i>Zizyphus</i> spp., <i>Capparis sepiaria</i> and <i>Balanites aegyptica</i>
				(15) <i>Syzygium rubicundum</i> - <i>Derris indica</i> association		Savanna	Dominant species: <i>Acacia</i> spp., <i>Zizyphus</i> spp., <i>Terminalia crenulata</i> , <i>Bauhinia racemosa</i> , <i>Tectona grandis</i> , <i>Anogeissus</i> spp, <i>Boswelia serrata</i> and <i>Balanites aegyptica</i> .
							Understorey grasses: <i>Appluda mutica</i> , <i>Heteropogon contotus</i> , <i>Themeda quadrivalvis</i> and <i>Sehima nervosum</i>
				Hill forest	Acacia – Anogeissus	Dominant species: <i>Tectona grandis</i> replaced by <i>Anogeissus latifolia</i> . in Gir (east),	

Khan et al.1990		Chellam 1993	Chavan 1993	Sharma 1995	Qureshi and Shah 2004	
Classes	Characteristic species	Classes	Classes	Classes	Classes	Characteristic species
						Associates: <i>Acacia spp., Terminalia spp., Wrightia tinctora, Grewia tiliifolia, Boswellia serrata, Flacourtia indica, Bauhinia racemosa and Zizyphus spp.,</i>
					Acacia – Lannea – Boswellia	<i>Acacia spp., Lannea coromandelica, Boswellia serrata, Tectona grandis, Terminalia crenulata, Wrightia tinctora, Soyamida febrifuga and Sterculia urens</i>

3.3 Literature Review

3.3.1 Remote Sensing Approach in understanding Vegetation Pattern

Scientists and practitioners have developed advanced classification approaches and techniques refining classification accuracy (Gong and Howarth 1992, Kontoes et al. 1993, Foody 1996, San Miguel-Ayanz and Biging 1997, Stuckens et al. 2000, Franklin et al. 2002, Pal and Mather 2003, Gallego 2004). However, classifying remotely sensed data into a thematic map is still a challenge because many factors, such as the complexity of the landscape in a study area, selected remotely sensed data, and image processing and classification approaches, may affect the success of a classification. Multi-spectral data has been useful in providing a number of independent measurements of the emitted electromagnetic radiation of the observed surface elements. Each type of surface element with a different typical spectral signature provides a basis for classification. The spectral measurements are effective as the emitted radiation has a simple and consistent relation with the inherent properties of surface materials, and it is desired that the data within each spectral band will be uncorrelated with all the rest. The multispectral band data is correlated in various degrees and hence a performance of principal component analysis over the multispectral data is prescribed. This process enabled to extract the true dimensionality of the spectral data and its interpretation as had been carried out by Kauth and Thomas 1976 and Crist and Cicone 1984 for the Landsat MSS and TM data respectively.

3.3.2 Principal Component Analysis (PCA)

Principal Component Analysis (PCA) is a useful tool for analysis of multispectral remote sensing images (Roger 1996). It is used to simplify data processing of satellite multi-spectral imagery (Richards 1986) transforming a set of correlated spectral bands of image data into an equivalent set of uncorrelated components of data. The main advantage of PCA is that the transformed components can be ordered by decreasing variances. PCA changes pixel definition from M-channel sets of numbers (counts) to K-principal component (PC) sets (K-PC, $K < M$), without remarkable loss of (relative) information, and so allowing the saving of computing time. Visual discrimination of targets may be improved by using the first two or three PCs to build files with output presented as coded

printer output or as false-coloured image in RGB video screen (Dwivedi and Ravi Shankar 1992).

Normally, for remote sensing data, just a few of the low order components have large variance and contain useful information, and the other high order components appeared as noise. The inherent thermal properties of a surface element – its thermal inertia (Watson 1975) and its evaporation rate, do not have simple relation, or a steady one with the thermal infrared radiant properties (the contribution of the variations in surface elements emissivities to the total variance in the thermal images is usually small). Therefore, efforts to analyze simultaneously the reflective and the thermal signatures of surface elements have not been very productive (Price 1981, Altamira et al. 1986), even though the thermal data carry additional unique discriminative factors. Nevertheless, some success had been attained over vegetative landscapes, where it had a strong negative correlation between spectral vegetation indices and surface temperature (Gowerd et al. 1985, Whitehead et al. 1986, Nemani and Running 1989, Sellers et al. 1992). Thus the inclusion of the thermal infrared data affects the principal components by changing their relative importance and their corresponding eigenvectors; hence omission of thermal infrared band in PCA is practiced widely.

3.3.3 Supervised Classification or Unsupervised Classification?

Supervised classification of multispectral remote sensing imagery, a common method used for land-cover determination, depends on defining training areas which sufficiently represent the spectral characteristics of each class in the image to be classified, as the quality of the training set has a significant effect on the classification process and its accuracy (Chuvieco and Congalton 1988). The procedure of finding and verifying training areas is rather labour-intensive, and need experience in selecting representative pixels for each of the classes which is done by visual examination of the image data and by information extraction from additional sources such as field data or existing maps (Schowengerdt 1997).

Contrasting supervised classification, clustering methods (or unsupervised methods) involve no training sets at all. Instead it attempts to analyze the underlying structure automatically by organizing the data into classes sharing similar, i.e. spectrally

homogeneous characteristics. By specifying the number of clusters present and with a little *a priori* information about the data, cluster analysis provides a useful method for organizing a large set of data so that the recovery of information may be made more efficiently. In addition, one main objective of using clustering algorithms for pre-classification of multispectral remote sensing data is to obtain optimum information for the selection of training regions for subsequent supervised land-use segmentation of the imagery.

3.3.4 Accuracy Assessment

Accuracy assessment, governs the degree of “correctness” of the classified vegetation groups compared to the actual ones. A vegetation map resulting from image classification is considered correct if it provides a true representation of the region it characterizes (Foody 2002, Weber 2006). Four essential stages have been used in accuracy assessment methods (Congalton 1994). 1. By visual inspection of the resulting maps. This method was found very subjective and often not precise. 2. Comparisons of the area extents of the classes in the resultant thematic maps (e.g. the percentage of a specific vegetation group in area) were made with the matching extents on ground or other reference dataset. However, the problem with this non-site-specific approach is the correct proportions of vegetation do not always mean the precise locations at which they are found. 3. In the third stage, the accuracy metrics were prepared on a comparison of the class labels in the thematic map with the ground data for the same locations. Measures such as the percentages of cases correctly (and wrongly) classified were used evaluating the classification accuracy. 4. The accuracy assessment at the fourth stage made further refinements on the basis of the third stage and confusion or error matrix was used, which defines the fitness between the derived classes and the reference data using the measures like overall accuracy and kappa coefficient.

Although accuracy assessment is important to qualify the result of image classification, it is perhaps impossible to specify a single, all-purpose measure for measuring classification accuracy. For example, the confusion matrix and its derived measures of accuracy may seem sensible and practical but they may not be applicable under some circumstances, especially in vegetation mapping at coarse scales (Cingolani et al. 2004). One of the problems caused by the pixel based confusion matrix evaluation is that a pixel at a coarse

resolution may contain several vegetation types. As shown in Figure 3.3, suppose a pixel in imagery represents a composite of three vegetation classes (class A, B and C). Undoubtedly the eclipse located in the center of the pixel may be the sampling area. Since it is impracticable to sample the whole pixel at a large scale mapping, this pixel would most expected be labeled with class B in image classification considering its percentage of the occupied area. Therefore, the vegetation class between the derived (class B) and the referenced (class A) will not match and this mismatch will introduce classification errors. In this case the non-site specific accuracy measures may be more suitable if not for the constraint mentioned previously. Additionally, rather than using field samples to test the classification accuracy, a commonly accepted practice is to use finer resolution satellite data to evaluate coarser resolution products (Cihler et al. 2003), although the high resolution data are themselves subject to interpretation and possible errors (Defries and Townshend 1999). The result evaluating for image classification still remains a hot debating topic today (Foody 2002).

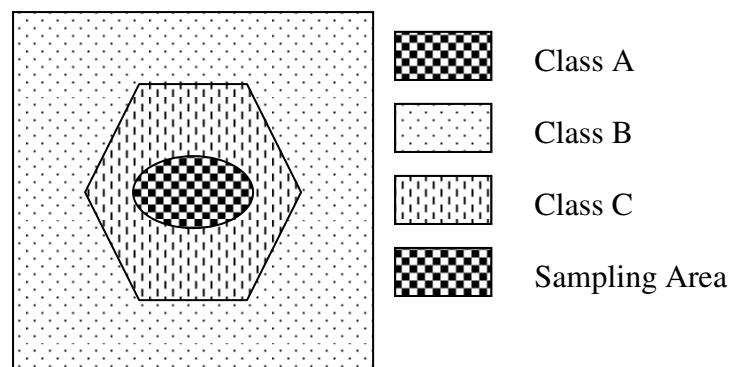


Figure 3.3: Illustration for pixel based accuracy assessment at coarse scale. The envelope square represents a pixel in imagery. Here problem occurs: ground ‘true’ vegetation class is A, but classified result for the pixel, if correctly classified, would be labeled with B. This would lead to a mismatch between ground referenced data and classified result, which is very typical in pixel-based accuracy assessment especially at large scale vegetation mapping.

KAPPA analysis is a discrete multivariate technique used in accuracy assessments (Congalton and Mead 1983, Jensen 1996). KAPPA analysis yields a K_{hat} statistic (an estimate of KAPPA) that is measure of agreement or accuracy (Congalton 1991). The K_{hat} statistic is computed as:

$$K_{\text{hat}} = \frac{N \sum_{i=1}^r x_{ii} - \sum_{i=1}^r (x_{i+} x_{+i})}{N^2 - \sum_{i=1}^r (x_{i+} x_{+i})}$$

where r is the number of rows in the matrix, x_{ii} is the number of observations in row i and column i , x_{i+} and x_{+i} are the marginal totals for row i and column i respectively and N is the total number of observations.

3.3.5 Understanding the effect of Scaling

With the change in thematic resolution of categorical maps the number of classes and their spatial pattern might often get altered, thus resulting in differences in landscape metrics. The number of patch types in contrived maps revealed to be directly related to thematic resolution, had significant effects on contagion (Li and Reynolds 1993). In addition the level of classification detail could affect numerous landscape indices (Li and Wu 2004). There are two key methods to produce categorical spatial patterns with different spatial resolutions for a given landscape. The more common and simpler one is spatial aggregation. Majority rules have been normally used for this purpose in landscape ecology (Turner et al. 1989, Benson and Mackenzie 1995, Wickham and Riitters 1995, Frohn 1998, Wu et al. 2002). These studies anticipated that majority rules could produce aggregated patterns adequately similar to those directly mapped through remote sensors with coarser spatial resolutions. The other approach is directly classifying simultaneously gathered satellite images covering the same study area but with different sensor spatial resolutions. This latter approach has been less commonly used in landscape ecological studies (Benson and MacKenzie 1995).

3.4 Objectives

The aim of this chapter was to classify and show spatial patterns of the existing vegetation of the Gir PA. The following were the broad objectives of the present study:

- To evaluate the current landscape architecture of Gir Protected Area and its surroundings (5 km buffer around Gir Protected Area).
- To map land use/cover classes of different vegetation types, water bodies, barren areas.

- To investigate how commonly used landscape metrics respond to changing grain size (spatial resolution).

3.5 Hypothesis

H₀: With the change in spatial resolution, landscape metrics at patch, class, landscape scales does not change showing any particular trend.

H_a: With the change in spatial resolution, landscape metrics at patch, class, landscape scales changes showing a particular trend which is predictable across the scales.

3.6 Methods

3.6.1. Field Methods

3.6.1.1. *Stratification and Sampling Units*

In Gir PA, precipitation determines the structure of vegetation communities. In a dry deciduous forest tract, the vegetation is almost entirely confined to watercourses. This discontinuous cover and the scattered distribution of shrubs play an important role in structuring plant communities. A vegetation survey was therefore carried out in November-January 2008-2011 to assess the variation in vegetation structure in different habitats. The study area was explored with a view to classify the habitat focusing on the species survival. Following a systematic sampling design, the entire Gir Protected Area (Gir PA) was divided into 2 km X 2 km grids and alternatively each grid was selected as sampling grid (Figure 3.4). In each of these sampling grids, one 2.5 km transect was laid diagonally covering most of the area of the grid. In this transect, at 250 meter intervals, different plots were laid of 10 meter and 5 meter radius to collect phytosociological data of plants (Figure 3.5). Other descriptive parameters were also used to differentiate habitats like: topography, precipitation, hydrology, vegetation structure, and human land use i.e. presence of *nesses*, cultivable lands, presence of roads inside the Gir PA.

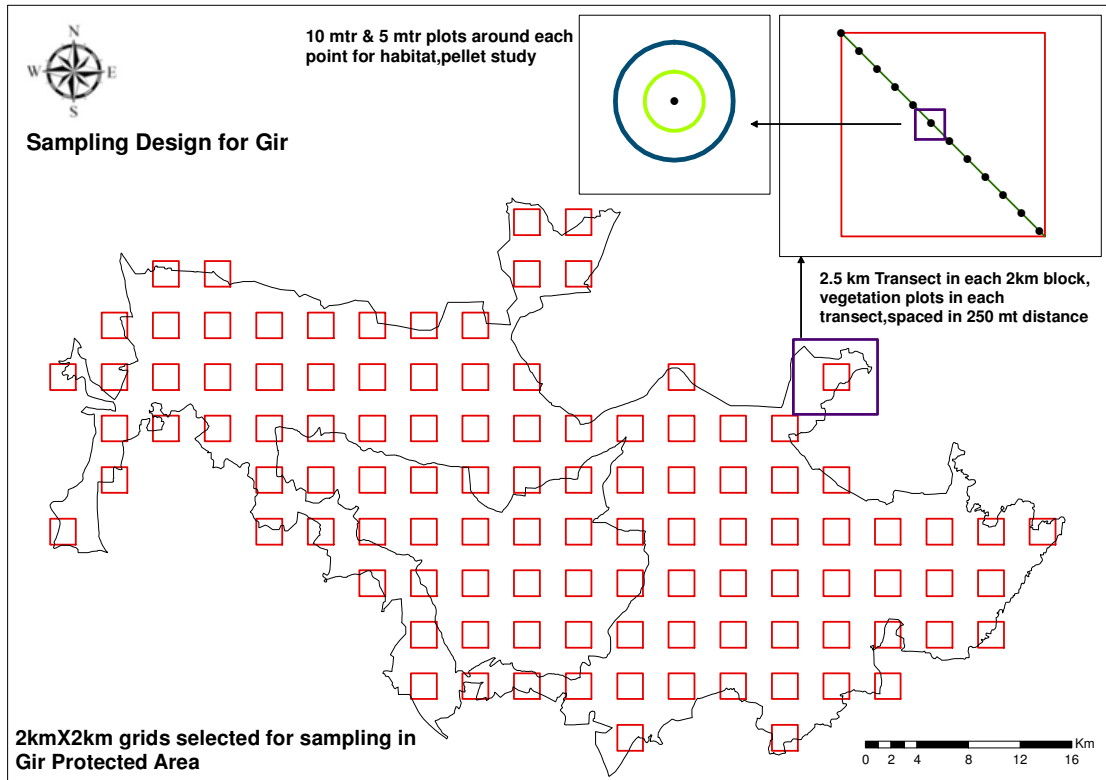


Figure 3.4: Vegetation Sampling Design

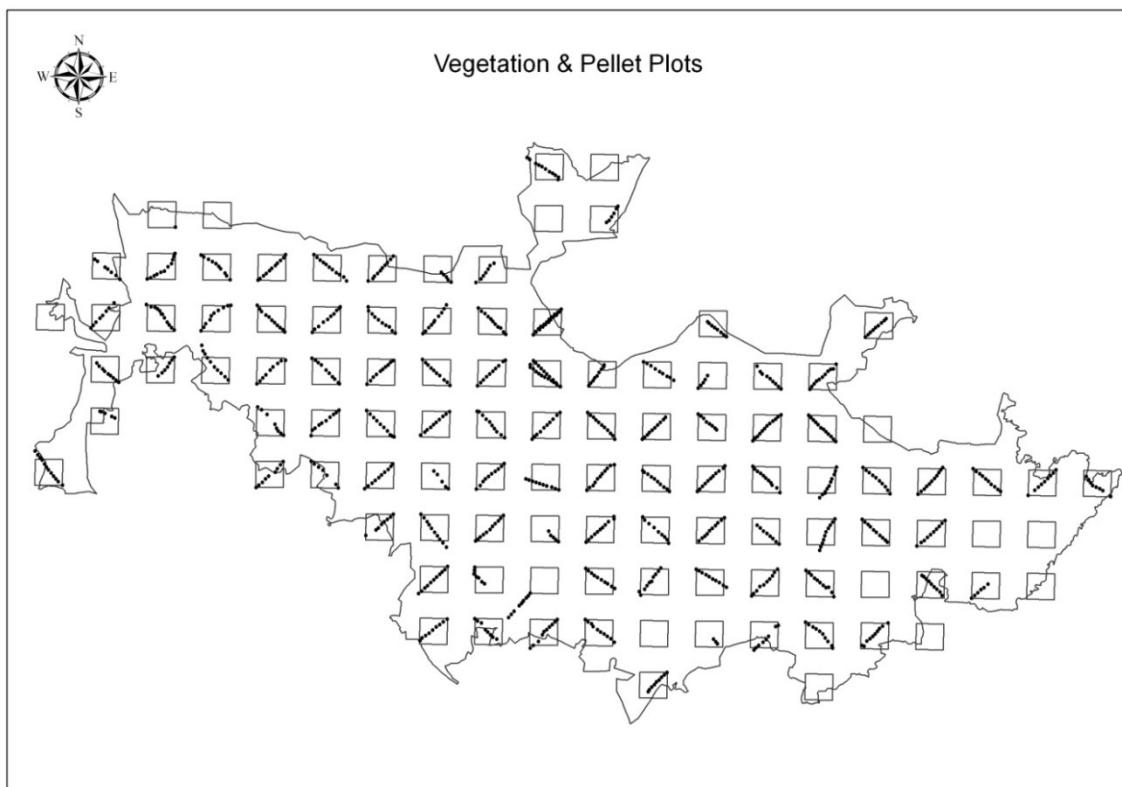


Figure 3.5: Distribution of vegetation sampling plots in Gir Wildlife Sanctuary

3.6.1.2. *Data Collection Protocols*

Data on species composition and structure were collected using circular plots method following Muller-Dombois and Ellenberg 1974. Circular plots are expeditious in allowing accurate area sampling with relatively less effort for plot layout (a single central marker for permanent location) and they reduce the number of edge decisions because they minimize perimeter to area ratio (Mc Cune and Grace 2002). The following details were collected from the plots:

Tree Species: At each sampling point, a circular plot of 10 m radius was laid for enumeration of trees. The individuals with > 20 cm girth at breast height (gbh) and height > 1.37 m with distinct bole were considered as trees (Muller-Dombois and Ellenberg 1974). In each plot, parameters like species names, number of trees, gbh and % canopy cover were recorded. Tree saplings (gbh > 10.5cm and < 20 cm; height > 30cm and < 1.37m) (Singh et al. 1995) in 5m radius nested plot were also recorded.

Shrub Species: Names and number of shrubs were enumerated from nested circular plot of 5m radius.

Herb Species: Nested 1m×1m quadrat was laid for estimation of ground vegetation. Name of species and percentage of herb cover, grass cover, litter cover, weed cover were recorded from each quadrat.

Community structure and distribution pattern of species: Density, frequency, abundance and dominance of constituent species were determined. We calculated the Importance Value Index (IVI) for all species by adding the relative values of frequency, density and dominance (basal area) (Curtis and McIntosh 1950, Brown and Curtis 1952). With the A/F ratio the distribution of trees was calculated which indicates regular distribution if the value is <0.025, random distribution, if it is in between 0.025 to 0.050 and contiguous distribution if >0.050 (Curtis and Cotton 1956).

The following formulae were used for analysis

$$\text{Density/ Quadrat} = \frac{\text{Total number of individuals in all quadrats}}{\text{Total number of quadrats studied}}$$

$$\text{Frequency} = \frac{\text{Total number of quadrats in which species occurred}}{\text{Total number of quadrats studied}}$$

$$\text{Abundance / Quadrat} = \frac{\text{Total number of individuals of a species in all quadrats}}{\text{Total number of quadrats in which species occurred}}$$

$$\text{Total Basal Area} = \text{Mean Basal Area} \times \text{Density}$$

$$\text{Mean Basal Area} = \frac{C^2}{4\pi}$$

$$\text{where } C \text{ is Mean of the circumference} = \frac{\text{Sum of all CBH of a species}}{\text{Total number of individuals of a species}}$$

$$\text{Relative Density} = \frac{\text{The density of a species}}{\text{The density of all species}} \times 100$$

$$\text{Relative Frequency} = \frac{\text{The frequency of a species}}{\text{The frequency of all species}} \times 100$$

$$\text{Relative Dominance} = \frac{\text{Total Basal Cover of a species}}{\text{Total Basal Cover of all species}} \times 100$$

Importance Value Index

$$= \text{Relative Density} + \text{Relative Frequency} + \text{Relative Dominance}$$

Diversity, Richness and Evenness:

To analyze that the individuals are randomly sampled from an infinite large population and all the species from the community are included in the sample. Species diversity was computed using **Shannon Wiener Index** (Shannon and Weaver 1949).

$$H' = \sum (n_i / N) \times \ln (n_i / N)$$

Where, H' = Shannon's information index of species diversity

n_i = Total number of individuals of one species

N = Total number of individuals of all the species in one stand

Richness was calculated by counting total number of species observed in each habitat

Evenness (Equability) was calculated using the Pielou's (1966) equation

$$\text{Evenness (E)} = H' / H' \text{ max,} = H' \log S \text{ or } H' / \ln S$$

Where, S = Number of species

$$H' = \text{Diversity}$$

Evenness ranges between 0 and 1. If the evenness value is higher, the variation in communities between the species would be less.

3.6.2. Remote sensing approach

This study used a cloud free LANDSAT TM scene, which was located in Path 149, Row 45 and acquired in November 2009. Bands 1-5 and 7 were utilized. The image was checked for radiometric distortions such as line striping. Using ERDAS software, the TM image was geometrically rectified using known ground control points.

To assess the potential to map forest classes, natural groupings in both the remotely-sensed and ground-collected ecological data sets were sought. The ground data (n > 900 vegetation sampling plots) were grouped into ecological classes with a conventional ecological ordination technique. These data were classified according to species composition using a TWINSpan analysis (Hill 1979), which can be used to define a number of possible groupings by focusing on different levels in the classification hierarchy. The remotely sensed multispectral data were compressed via a principal component analysis and was grouped with an unsupervised classification approach using Erdas 10.

All bands of the TM image except the thermal band were used in PCA to extract information and transforming a set of correlated spectral bands of image data into an equivalent set of uncorrelated components of the data. Three principal components were derived based on eigen values and eigen vectors. The eigen values (variances of the principal components) are shown in Table 3.2. The unsupervised classification was performed on the first three principal components, which accounted for more than 99% of the spectral variance in the original six LANDSAT TM5 channels.

Table 3.2: Principal Component Analysis

Principal Component	1	2	3	4	5	6
Normalized Variance	0.845	0.121	0.022	0.007	0.004	0.001

These three principal components were stacked together to use for unsupervised classification. Scene classification was undertaken using a decision rule which first separated water bodies and forest-non forest areas and then classifying the forest areas

into different forest types. The stacked PC layers were initially classified using the Isodata algorithm (Unsupervised classification) into 100 classes and then regrouped into the ten major classes under investigation based on the field study.

Initially classified images were evaluated visually, and the effects of topographic normalization were noted. The image was evaluated quantitatively with ground truth data points. Several other types of ground truth maps, such as drainage network, road network, and settlement areas were included in the ground truth map. Between November 2008 to January 2011, extensive field reconnaissance trips were made throughout the study site. Vegetation composition and structure were noted and many zones of anthropogenic disturbance were mapped in the field. We assessed the accuracy of the classification using a classification error matrix. Cross-tabulations were made of sites visited in the field and the image classifications. Errors were assessed using commission error (a measure of the number of sites of other vegetation types incorrectly assigned to a particular vegetation type), omission error (a measure of the number of sites of a particular vegetation type incorrectly assigned to other vegetation types).

I delineated an one hectare quadrat plot around each of our circular vegetation plots, used in the accuracy assessment. One hectare was used as the unit because one hectare area of any forest type was assumed as minimum requirement to be specified as a patch of that forest type. As the satellite imagery used for generating vegetation map is of 30 meter resolution, I quantified the number of pixels of different vegetation types falling within the one hectare quadrat and denoted the vegetation type of the quadrat according to dominant vegetation type found in that quadrat. While doing accuracy assessment, I checked whether the vegetation type affirmed by our sampling area is same with the major vegetation type of the quadrat or not. KAPPA analysis was used to perform a classification accuracy assessment based on error matrix analysis. The K-index measures the actual agreement between the two maps minus chance agreement. The overall K-index is based on all entries in the table and is a single index expressing both omission and commission error (Franklin 1991). For the classified image, accuracy was determined based on a stratified random sample of 200 points distributed throughout the area. Kappa statistics, which assess overall accuracy by incorporating individual errors of omission and commission were also calculated (Congalton 1991) (Table 3.4)

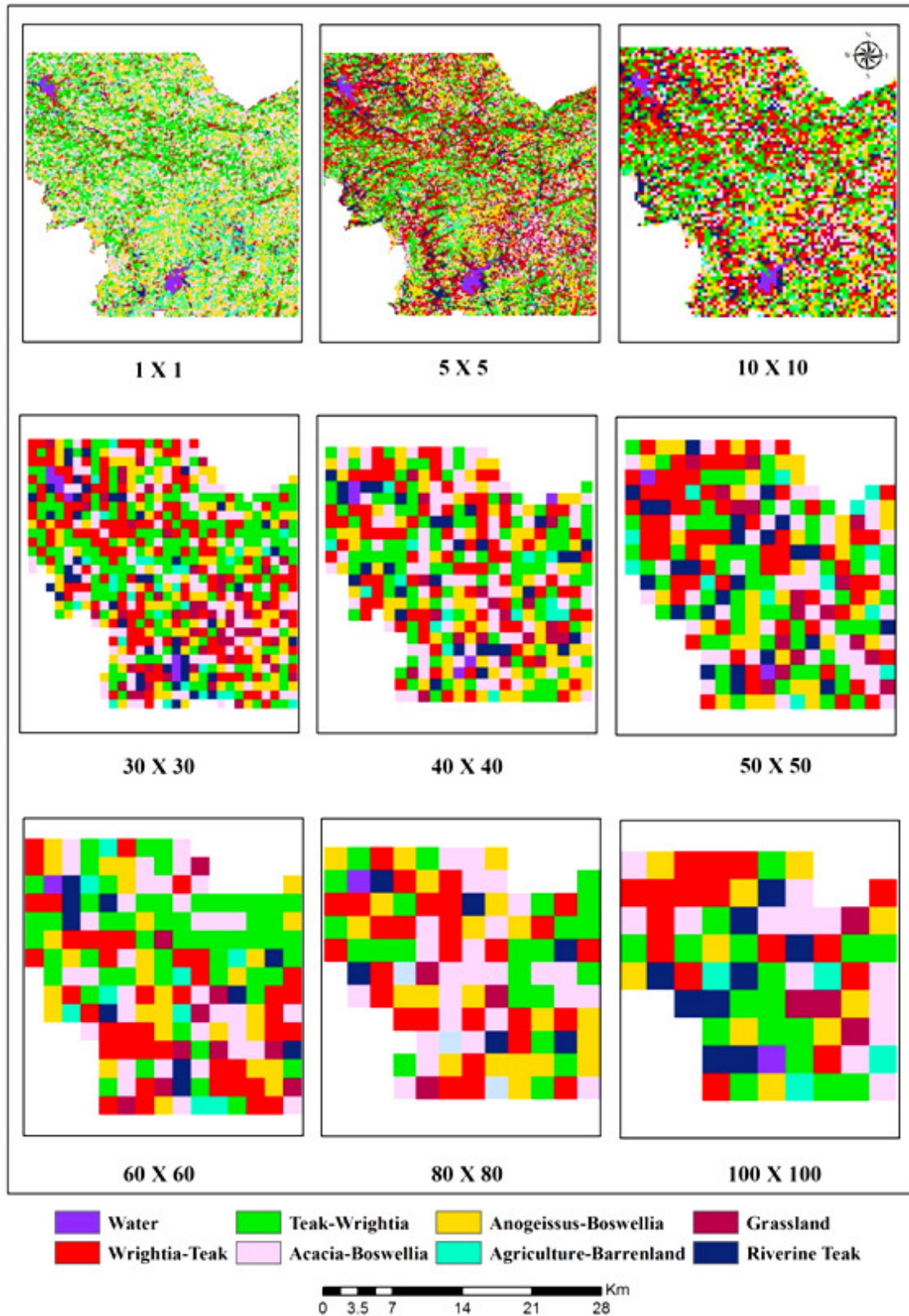


Figure 3.6: A portion of the land use/cover map of Gir PA with different grain sizes or spatial resolutions, ranging from 1X1 to 100X100 original Landsat TM pixels

To investigate the effects of changing grain size, the grain size was systematically changed from 1 by 1 (30 m X 30m) to 100 by 100 pixels (3000m X 3000m) following the majority rule with the extent kept constant. The grain of the landscape data was changed by aggregating groups of n adjacent pixels into a single data unit. The land cover type of the majority of pixels in an aggregate was assigned to the new data unit (Shown schematically in Figure 3.6). If two land covers in an aggregate occurred in equal proportions (e.g., two pixels each of different forest types), the assignment of a landscape type was done at random. The finest aggregation was 150 meter and the coarsest aggregation was 3000 meter. In this study, the aggregation at each successive grain size always started with original (30 m X 30 m) data. This may be called the ‘independent’ aggregation scheme as opposed to the iterative aggregation scheme in which the aggregation at the next grain size is based on the already aggregated data of the initial grain size (Turner et al. 1989). Each new map, with progressively larger grain size (e.g., 1X1, 2X2, ..., 100X100), was created by directly aggregating the original data set, instead of using a cumulative procedure that would introduce more errors.

I calculated all patch, class, and landscape level metrics in this study. Few indices, having ecological importance, have been mentioned here (Table 3.3): The software package, FRAGSTATS 2.0 (McGarigal and Marks 1995) was used to compute these indices.

Table 3.3: List of landscape metrics showed considerable response according to the change in grain size

Landscape Metric	Abbreviation	Description
Number of Patches	NP	The total Number of Patches in the Landscape
Patch Density	PD	The number of patches per unit area, e.g., per km ²
Total Edge	TE	The sum of the lengths of all edge segments (unit:m)
Edge Density	ED	The total length of all edge segments per ha for the class or landscape of consideration (unit:m/ha)
Patch Richness	PR	The number of different patch types in the landscape

Landscape Metric	Abbreviation	Description
Patch Richness Density	PRD	The number of patch types per unit area
Shannon's Diversity Index	SHDI	<p>A measure of patch diversity in a landscape that is determined by both the number of different patch types and the proportional distribution of area among patch types</p> $H = - \sum_{i=1}^m p_i \ln(p_i)$ <p>Where m is the total number of patch types and P_i is the proportion of the landscape area occupied by patch type i (unitless)</p>
Largest Patch Index	LPI	The ratio of the area of the largest patch to the total area of the landscape (%)
Mean Patch Size	MPS	The average area of all patches in the landscape (unit: ha)
Patch Size Standard Deviation	PSSD	The standard deviation of patch size in the entire landscape (unit: ha)
Patch Size Coefficient of Variation	PSCV	The standard deviation of patch size divided by mean patch size for the entire landscape (unit %)
Landscape Shape Index	LSI	<p>A modified perimeter-area ratio of the form:</p> $LSI = \frac{0.25 E}{\sqrt{A}}$ <p>Where E is the total length of patch edges and A is the total area of the landscape (unitless)</p>
Mean Patch Shape Index	MSI	<p>A patch level shape index averaged over all patches in the landscape:</p> $MSI = \frac{\sum_{i=1}^m \sum_{j=1}^n \left[\frac{0.25 P_{ij}}{\sqrt{a_{ij}}} \right]}{N}$ <p>Where P_{ij} and a_{ij} are the perimeter and area of</p>

Landscape Metric	Abbreviation	Description
		patch ij , respectively, and N is the total number of patches in the landscape (unitless)
Area-Weighted Mean Patch Shape Index	AWMSI	<p>Mean patch shape index weighted by relative patch size :</p> $AWMSI = \sum_{i=1}^m \sum_{j=1}^n \left[\left(\frac{0.25 P_{ij}}{\sqrt{a_{ij}}} \right) \left(\frac{a_{ij}}{A} \right) \right]$ <p>Where P_{ij} and a_{ij} are the perimeter and area of patch ij, respectively. A is the total area of the landscape, m is the number of patch types, and n is the total number of patches of type I (unitless)</p>
Double-Log Fractal Dimension	DLFD	<p>The fractal dimension for the entire landscape which is equal to 2 divided by the slope of the regression line between the logarithm of patch perimeter:</p> $DLFD = \frac{2}{\frac{[N \sum_{i=1}^m \sum_{j=1}^n (\ln(P_{ij}) \ln(a_{ij}))] - [(\sum_{i=1}^m \sum_{j=1}^n \ln(a_{ij}))]}{(N \sum_{i=1}^m \sum_{j=1}^n (\ln(P_{ij}^2))) - (\sum_{i=1}^m \sum_{j=1}^n \ln(P_{ij}))^2}}$ <p>where P_{ij} and a_{ij} are the perimeter and area of patch ij, respectively, m is the number of patch types, n is the total number of patches of type i, and N is the total number of patches in the landscape (unitless).</p>
Mean Patch Fractal Dimension	MPFD	<p>The average fractal dimension of individual patches in the landscape, which is the summation of fractal dimension for all patches divided by the total number of patches in the landscape:</p> $FD = \frac{\sum_{i=1}^m \sum_{j=1}^n \left(\frac{2 \ln(0.25 P_{ij})}{\ln(a_{ij})} \right)}{N}$ <p>Where P_{ij} and a_{ij} are the perimeter and area of patch ij, respectively m is the number of patch types, n is the total number of patches of type i,</p>

Landscape Metric	Abbreviation	Description
		and N is the total number of patches in the landscape (unitless)
Area-Weighted Mean Patch Fractal Dimension	AWMFD	<p>The patch fractal dimension weighted by relative patch area:</p> $AWMFD = \sum_{i=1}^m \sum_{j=1}^n \left(\frac{2 \ln(0.25 P_{ij})}{\ln(a_{ij})} \left(\frac{a_{ij}}{A} \right) \right)$ <p>Where P_{ij} and a_{ij} are the perimeter and area of patch ij, respectively, m is the number of patch types, n is the total number of patches of type i, and A is the total area of the landscape (unitless)</p>

3.7 Results

A. TWINSPAN Analysis

Initial analysis centered on the definition of ecological classes in the ground data. The TWINSPAN classification hierarchy separated the Gir vegetation into teak associated and non-teak forests (Table 3.4). A total of ten communities were identified as a result of the analysis (Figure 3.7) i.e. *Ziziphus mauritiana*,-*Tectona grandis* community, pure *Tectona grandis* community, mixed teak communities like *Wrightia tinctoria*,-*Tectona grandis* community and *Tectona grandis*-*Wrightia tinctoria* community, *Terminalia alata*-*Tectona grandis*-*Lannea coromandelica*-*Acacia catechu* community, *Boswellia serrata*-*Acacia catechu* community, *Acacia catechu*-*Ziziphus mauritiana* community, *Anogeissus latifolia*-*Boswellia serrata* community, *Terminalia alata*-*Acacia leucophloea*-*Anogeissus latifolia*-*Butea monosperma* community, *Acacia nilotica*-*Butea monosperma*-*Ziziphus mauritiana* community.

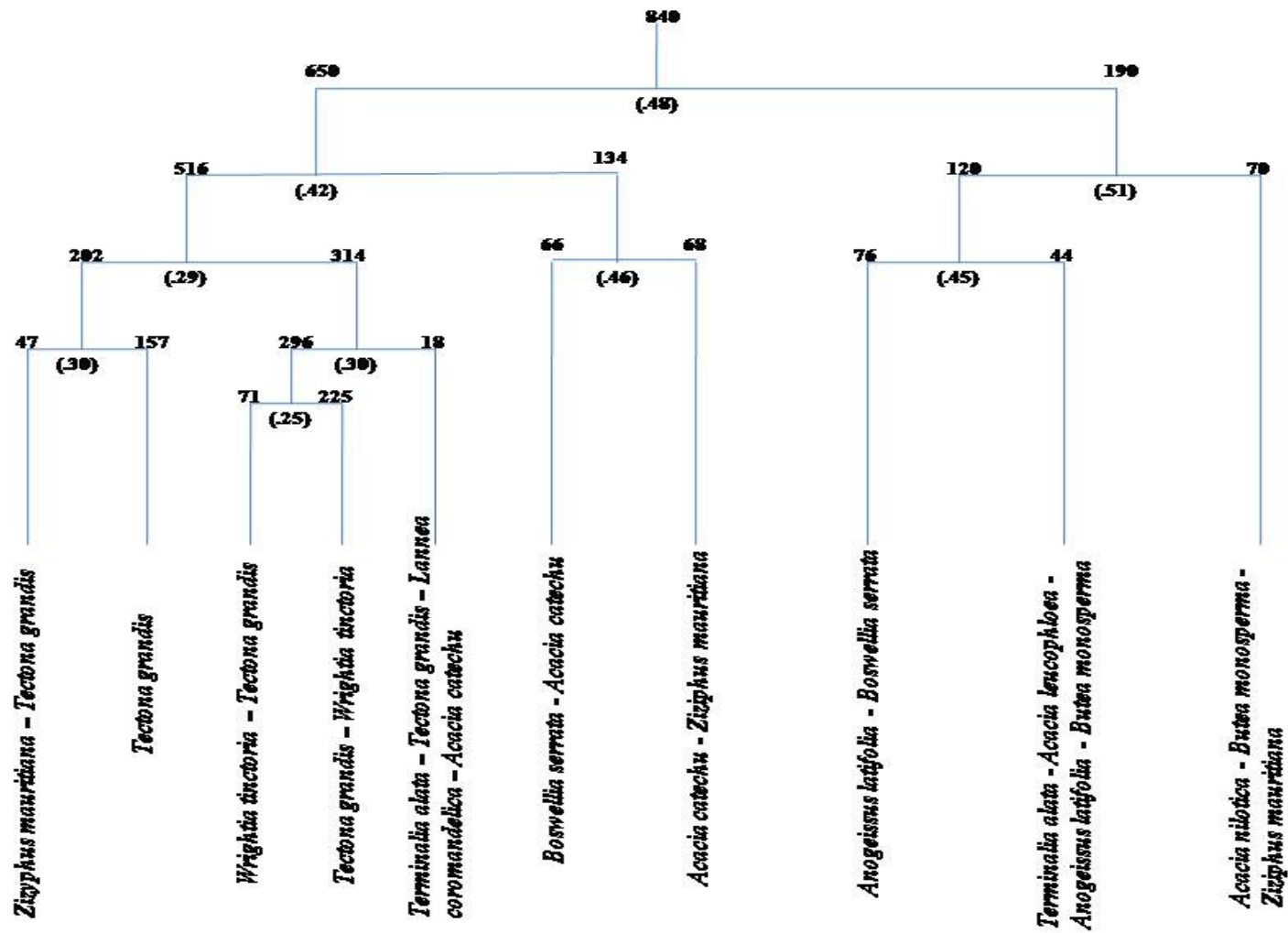


Figure 3.7: Dendrogram showing vegetation communities identified through TWINSpan analysis

Table 3.4: Community wise distribution of vegetation sampling plots, dominant species, constant species, diversity, richness and Evenness

Community	No of Plots	Name of the Community	Dominant Species (Based on IVI Value)	Constant Species (Frequency>25%)	Diversity	Richness	Evenness
1	45	<i>Ziziphus mauritiana</i> , <i>Tectona grandis</i>	<i>Ziziphus mauritiana</i> (85.00), <i>Tectona grandis</i> (81.02)	<i>Ziziphus mauritiana</i> (100.00), <i>Tectona grandis</i> (71.11), <i>Acacia catechu</i> (26.67), <i>Acacia auriculiformes</i> (26.67)	2.13	25	0.66
2	157	<i>Tectona grandis</i>	<i>Tectona grandis</i> (143.83)	<i>Tectona grandis</i> (94.90), <i>Ziziphus mauritiana</i> (37.58), <i>Acacia nilotica</i> (29.94)	2.09	51	0.53
3	71	<i>Wrightia tinctoria</i> , <i>Tectona grandis</i>	<i>Wrightia tinctoria</i> (68.11), <i>Tectona grandis</i> (46.86)	<i>Wrightia tinctoria</i> (92.96), <i>Tectona grandis</i> (59.15), <i>Acacia catechu</i> (35.21), <i>Ziziphus mauritiana</i> (38.03), <i>Bauhinia purpurea</i> (30.99)	2.84	48	0.73
4	223	<i>Tectona grandis</i> , <i>Wrightia tinctoria</i>	<i>Tectona grandis</i> (124.43), <i>Wrightia tinctoria</i> (29.12)	<i>Tectona grandis</i> (98.21), <i>Wrightia tinctoria</i> (44.39), <i>Acacia catechu</i> (27.80)	2.30	55	0.57
5	51	<i>Terminalia alata</i> , <i>Tectona grandis</i> , <i>Lannea coromandelica</i> , <i>Acacia catechu</i>	<i>Terminalia alata</i> (39.52), <i>Tectona grandis</i> (38.13), <i>Lannea coromandelica</i> (30.64), <i>Acacia catechu</i> (31.00)	<i>Terminalia alata</i> (47.06), <i>Tectona grandis</i> (45.10), <i>Lannea coromandelica</i> (45.10), <i>Acacia catechu</i> (43.14), <i>Boswellia</i>	2.92	40	0.79

Community	No of Plots	Name of the Community	Dominant Species (Based on IVI Value)	Constant Species (Frequency>25%)	Diversity	Richness	Evenness
				<i>serrata</i> (27.45), <i>Emblica officinalis</i> (25.49)			
6	31	<i>Boswellia serrata</i> , <i>Acacia catechu</i>	<i>Boswellia serrata</i> (106.51), <i>Acacia catechu</i> (41.21)	<i>Boswellia serrata</i> (90.32), <i>Acacia catechu</i> (54.84), <i>Manilkara hexandra</i> (29.03), <i>Acacia spp</i> (29.03)	2.20	20	0.73
7	66	<i>Acacia catechu</i> , <i>Ziziphus mauritiana</i>	<i>Acacia catechu</i> (131.10), <i>Ziziphus mauritiana</i> (25.10)	<i>Acacia catechu</i> (100.00)	2.16	40	0.58
8	75	<i>Anogeissus latifolia</i> , <i>Boswellia serrata</i>	<i>Anogeissus latifolia</i> (89.37), <i>Boswellia serrata</i> (26.54)	<i>Anogeissus latifolia</i> (86.67), <i>Acacia catechu</i> (30.67), <i>Acacia nilotica</i> (32.00)	2.43	34	0.69
9	44	<i>Terminalia alata</i> , <i>Acacia leucophloea</i> , <i>Anogeissus latifolia</i> , <i>Butea monosperma</i>	<i>Terminalia alata</i> (80.08), <i>Acacia leucophloea</i> (52.75), <i>Anogeissus latifolia</i> (37.46), <i>Butea monosperma</i> (27.55)	<i>Terminalia alata</i> (59.09), <i>Acacia leucophloea</i> (56.82), <i>Anogeissus latifolia</i> (25.00), <i>Butea monosperma</i> (25.00)	2.40	27	0.73
10	69	<i>Acacia nilotica</i> , <i>Butea monosperma</i> , <i>Ziziphus mauritiana</i>	<i>Acacia nilotica</i> (103.49), <i>Butea monosperma</i> (34.55), <i>Ziziphus mauritiana</i> (26.03)	<i>Acacia nilotica</i> (88.41), <i>Butea monosperma</i> (39.13), <i>Ziziphus mauritiana</i> (30.43)	2.29	30	0.67

A.1. *Ziziphus mauritiana*- *Tectona grandis* community

This vegetation community occurred in the south-western and middle part of the Gir Wildlife Sanctuary (Gir WLS). Approximately 25 species of trees were found in this community. According to IVI value, the dominant trees were *Ziziphus mauritiana* (85.00), *Tectona grandis* (81.02). The constant species with the frequency of occurrence more than 25 % were *Ziziphus mauritiana* (100.00), *Tectona grandis* (71.11), *Acacia catechu* (26.67), and *Acacia auriculiformes* (26.67). The total tree density of this vegetation type was 189.67/ha. The diversity value was 2.13 and the richness, evenness values were 25, 0.66. Details of distribution of vegetation sampling plots within this communities and species composition, status are given in Figure 3.8 and Table 3.5.

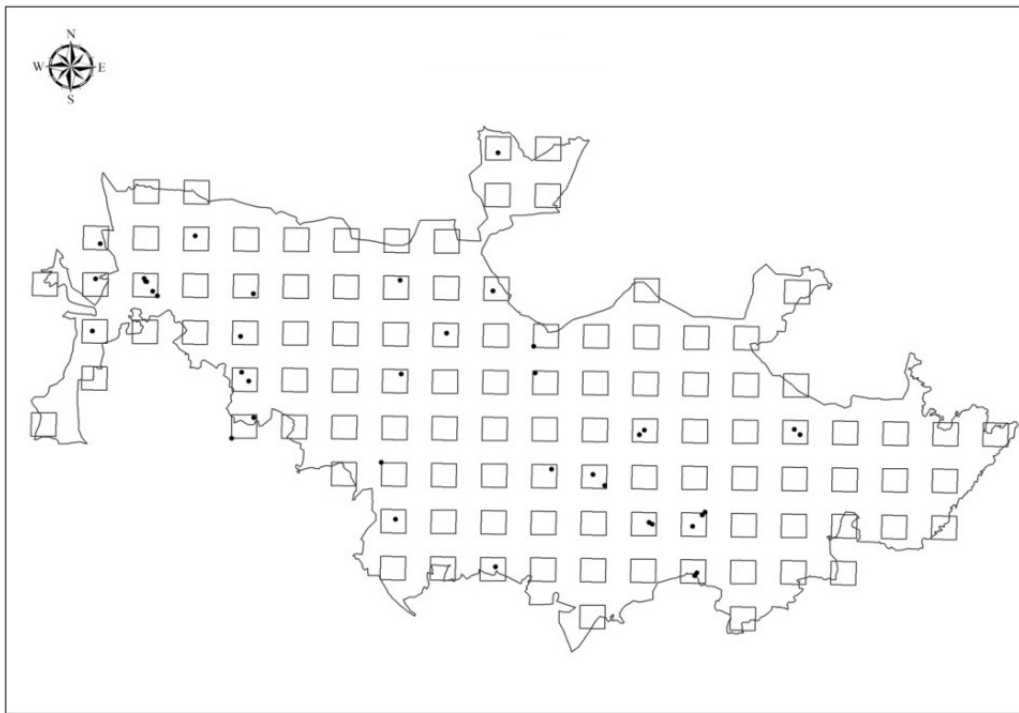


Figure 3.8: Distribution of Vegetation sampling plots classified in *Ziziphus mauritiana*- *Tectona grandis* community

Table 3.5: Species composition and status in *Ziziphus mauritiana*- *Tectona grandis* community

Species	Density/ha	Frequency	Basal Area	IVI	A/F Ratio
<i>Bauhinia purpurea</i>	4.95	15.56	0.03	8.44	0.06
<i>Acacia sp</i>	3.54	11.11	0.07	8.52	0.09
<i>Acacia nilotica</i>	15.57	26.67	0.13	22.36	0.07
<i>Aegle marmelos</i>	3.54	6.67	0.07	7.44	0.25
<i>Ziziphus mauritiana</i>	58.03	100.00	0.51	85.00	0.02
<i>Grewia tiliiaefolia</i>	2.83	6.67	0.03	4.87	0.20
<i>Wrightia tinctoria</i>	10.62	13.33	0.12	15.40	0.19
<i>Acacia senegal</i>	0.71	2.22	0.00	1.09	0.45
<i>Zizyphus xylopyra</i>	0.71	2.22	0.01	1.31	0.45
<i>Acacia leucophloea</i>	1.42	4.44	0.00	2.30	0.23
<i>Balanites aegyptiaca</i>	2.12	4.44	0.04	4.24	0.34
<i>Holarrhena pubescens</i>	0.71	2.22	0.01	1.33	0.45
<i>Acacia catechu</i>	12.03	26.67	0.21	24.74	0.05
<i>Phoenix sylvestris</i>	0.71	2.22	0.00	1.04	0.45
<i>Azadirachta indica</i>	1.42	4.44	0.02	3.18	0.23
<i>Flacourtia indica</i>	0.71	2.22	0.02	2.06	0.45
<i>Dichrostachys cinerea</i>	2.83	8.89	0.00	4.33	0.11
<i>Lannea coromandelica</i>	1.42	4.44	0.05	4.62	0.23
<i>Grewia asiatica</i>	0.71	2.22	0.00	1.19	0.45
<i>Morinda pubescens</i>	0.71	2.22	0.03	2.63	0.45
<i>Tectona grandis</i>	55.91	71.11	0.62	81.02	0.03
<i>Terminalia alata</i>	1.42	4.44	0.00	2.06	0.23
<i>Boswellia serrata</i>	4.25	6.67	0.01	4.82	0.30
<i>Schleichera oleosa</i>	1.42	4.44	0.01	2.64	0.23
<i>Ficus glomerata</i>	1.42	4.44	0.02	3.18	0.23

A.2. *Tectona grandis* Community

This vegetation community occurred in the western and central part of the Gir WLS. Approximately 51 species of trees were found in this community. *Tectona grandis* (143.83) was found as the dominant species. Other constant species with the occurrence value more than 25% were *Tectona grandis* (94.90), *Ziziphus mauritiana* (37.58), and *Acacia nilotica* (29.94). The total tree density of this vegetation type was 266.75/ha. Diversity value was 2.09, richness and evenness values were 51 and 0.53. Details of distribution of vegetation sampling plots within this communities and species composition, status are given in Figure 3.9 and Table 3.6.

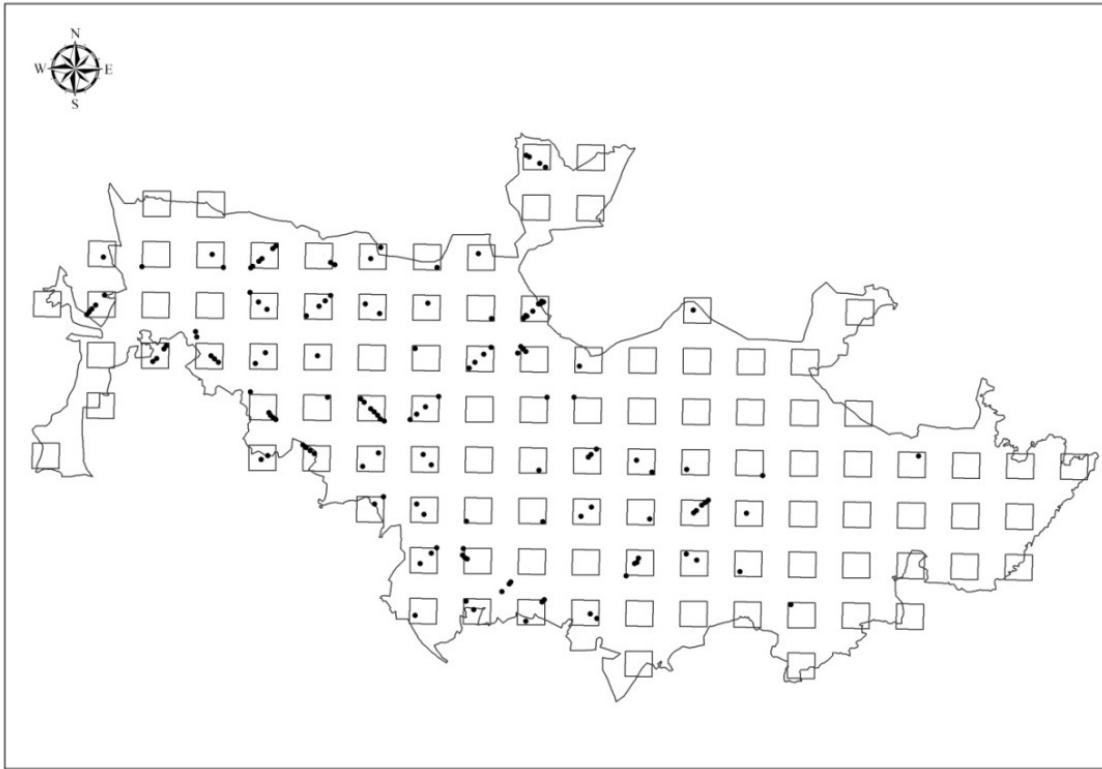


Figure 3.9: Distribution of Vegetation sampling plots classified in *Tectona grandis* Community

Table 3.6: Species composition and status in Community *Tectona grandis* Community

Species	Density/ha	Frequency %	TBA	IVI	A/F Ratio
<i>Bauhinia racemosa</i>	0.20	0.64	0.00	0.28	1.57
<i>Bauhinia purpurea</i>	4.67	13.38	0.05	6.26	0.08
<i>Emblica officinalis</i>	1.83	5.10	0.05	2.83	0.22
<i>Phyllanthus emblica</i>	1.22	3.82	0.06	2.44	0.26
<i>Tamarindus indica</i>	2.03	3.18	0.10	3.14	0.63
<i>Asparagus dumosus</i>	0.20	0.64	0.00	0.31	1.57
<i>Acacia sp</i>	3.45	8.28	0.07	4.72	0.16
<i>Terminalia bellirica</i>	0.20	0.64	0.00	0.27	1.57
<i>Acacia nilotica</i>	12.98	29.94	0.17	15.73	0.05
<i>Aegle marmelos</i>	2.03	3.82	0.05	2.54	0.44
<i>Pterocarpus marsupium</i>	0.20	0.64	0.00	0.27	1.57
<i>Ziziphus mauritiana</i>	14.81	37.58	0.28	20.17	0.03
<i>Grewia tiliiaefolia</i>	6.69	11.46	0.08	6.86	0.16
<i>Anogeissus latifolia</i>	0.81	0.64	0.02	0.79	6.28
<i>Wrightia tinctoria</i>	3.85	8.28	0.08	4.97	0.18

Species	Density/ha	Frequency %	TBA	IVI	A/F Ratio
<i>Grewia tenax</i>	1.42	4.46	0.01	1.89	0.22
<i>Cassia fistula</i>	2.03	5.73	0.02	2.66	0.19
<i>Acacia senegal</i>	2.84	5.10	0.06	3.33	0.34
<i>Zizyphus xylopyra</i>	3.25	7.01	0.04	3.71	0.21
<i>Haldinia cordifolia</i>	0.20	0.64	0.01	0.40	1.57
<i>Acacia leucophloea</i>	5.88	14.01	0.09	7.45	0.09
<i>Balanites aegyptiaca</i>	1.22	3.18	0.02	1.62	0.38
<i>Syzygium cumini</i>	0.81	0.64	0.17	3.11	6.28
<i>Holarrhena pubescens</i>	4.06	8.28	0.03	4.26	0.19
<i>Sterculia urens</i>	0.41	1.27	0.02	0.84	0.79
<i>Mallotus philippinensis</i>	0.81	2.55	0.02	1.26	0.39
<i>Mitragyna parvifolia</i>	0.20	0.64	0.01	0.34	1.57
<i>Derris indica</i>	0.20	0.64	0.02	0.54	1.57
<i>Acacia catechu</i>	7.91	15.29	0.13	9.17	0.11
<i>Butea monosperma</i>	9.53	24.84	0.21	13.67	0.05
<i>Azadirachta indica</i>	0.61	1.91	0.01	0.98	0.52
<i>Flacourtia indica</i>	0.81	1.91	0.01	0.95	0.70
<i>Dichrostachys cinerea</i>	1.22	1.91	0.01	1.21	1.05
<i>Catunaregam spinosa</i>	2.64	6.37	0.02	2.98	0.20
<i>Lannea coromandelica</i>	1.22	3.82	0.04	2.14	0.26
<i>Ficus arnottiana</i>	0.20	0.64	0.25	4.15	1.57
<i>Soymida febrifuga</i>	0.20	0.64	0.01	0.38	1.57
<i>Parkinsonia aculeata</i>	0.41	1.27	0.01	0.58	0.79
<i>Manilkara hexandra</i>	1.01	2.55	0.03	1.54	0.49
<i>Morinda pubescens</i>	2.84	5.73	0.05	3.46	0.27
<i>Tectona grandis</i>	151.73	94.90	3.84	143.83	0.05
<i>Terminalia alata</i>	2.03	5.73	0.03	2.77	0.19
<i>Boswellia serrata</i>	0.41	1.27	0.02	0.74	0.79
<i>Delonix elata</i>	0.41	1.27	0.01	0.59	0.79
<i>Holoptelia integrifolia</i>	0.20	0.64	0.01	0.35	1.57
<i>Dalbergia latifolia</i>	0.20	0.64	0.00	0.28	1.57
<i>Bombax ceiba</i>	0.41	1.27	0.01	0.67	0.79
<i>Diospyros melanoxylon</i>	2.03	5.73	0.07	3.45	0.19
<i>Schleichera oleosa</i>	0.61	1.27	0.02	0.89	1.18
<i>Ficus glomerata</i>	1.01	3.18	0.01	1.39	0.31
<i>Ehretia laevis</i>	0.61	1.91	0.00	0.82	0.52

A.3. *Wrightia tinctoria*-*Tectona grandis* Community

This type of vegetation occurred in the south-western and middle part of the Gir WLS. Approximately 48 tree species were found in this community. The dominant species were *Wrightia tinctoria* (68.11), *Tectona grandis* (46.86) and the constant species were with the occurrence value more than 25 % *Wrightia tinctoria* (92.96), *Tectona grandis* (59.15),

Acacia catechu (35.21), *Ziziphus mauritiana* (38.03), *Bauhinia purpurea* (30.99). The total tree density of this vegetation type was 252.98/ha. The diversity value of this vegetation type was 2.84/ha. The richness and evenness values of this vegetation type were 48 and 0.73. Details of distribution of vegetation sampling plots within this communities and species composition, status are given in Figure 3.10 and Table 3.7.

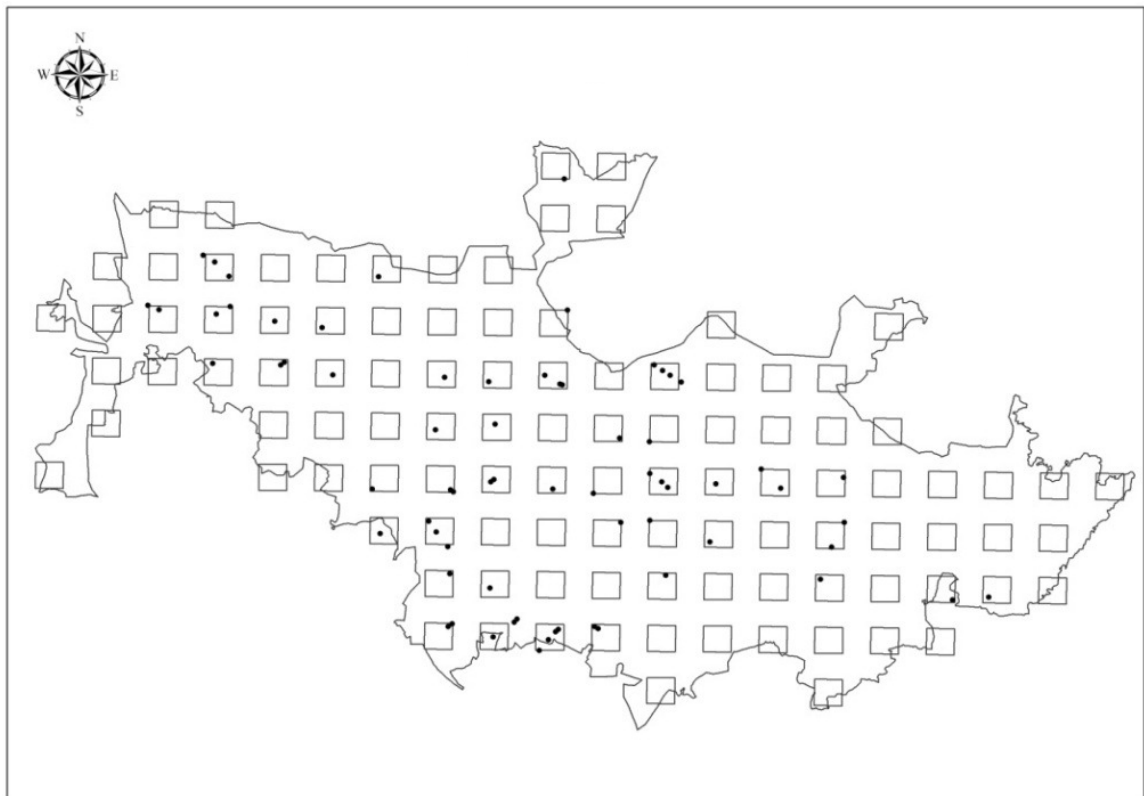


Figure 3.10: Distribution of Vegetation sampling plots classified in *Wrightia tinctoria-Tectona grandis* Community

Table 3.7: Species composition and status in *Wrightia tinctoria*-*Tectona grandis* Community

Species	Density/ha	Frequency %	TBA	IVI	A/F Ratio
<i>Bauhinia purpurea</i>	10.77	30.99	0.22	14.37	0.04
<i>Embllica officinalis</i>	0.90	2.82	0.03	1.37	0.36
<i>Phyllanthus emblica</i>	2.69	5.63	0.09	3.75	0.27
<i>Asparagus dumosus</i>	0.45	1.41	0.00	0.52	0.71
<i>Alangium salvifolium</i>	0.45	1.41	0.00	0.53	0.71
<i>Acacia sp</i>	1.79	5.63	0.04	2.48	0.18
<i>Hymenodyctyon orixense</i>	0.45	1.41	0.01	0.59	0.71
<i>Terminalia bellirica</i>	0.45	1.41	0.08	1.91	0.71
<i>Acacia nilotica</i>	1.35	4.23	0.02	1.80	0.24
<i>Aegle marmelos</i>	5.38	8.45	0.08	5.18	0.24
<i>Pterocarpus marsupium</i>	1.35	4.23	0.11	3.31	0.24
<i>Ziziphus mauritiana</i>	15.70	38.03	0.19	17.28	0.03
<i>Grewia tiliiaefolia</i>	3.14	7.04	0.04	3.35	0.20
<i>Anogeissus latifolia</i>	1.35	2.82	0.07	2.33	0.53
<i>Wrightia tinctoria</i>	67.28	92.96	1.30	68.11	0.02
<i>Grewia tenax</i>	0.90	2.82	0.01	1.07	0.36
<i>Cassia fistula</i>	1.35	2.82	0.02	1.42	0.53
<i>Acacia leucophloea</i>	6.28	15.49	0.19	8.97	0.08
<i>Balanites aegyptiaca</i>	0.90	1.41	0.01	0.75	1.42
<i>Holarrhena pubescens</i>	1.79	5.63	0.02	2.15	0.18
<i>Sterculia urens</i>	2.24	5.63	0.16	4.84	0.22
<i>Mallotus philippinensis</i>	3.14	9.86	0.07	4.52	0.10
<i>Mitragyna parvifolia</i>	0.45	1.41	0.03	0.97	0.71
<i>Derris indica</i>	1.79	4.23	0.04	2.21	0.32
<i>Capparis sepiaria</i>	0.45	1.41	0.01	0.56	0.71
<i>Acacia catechu</i>	20.63	35.21	0.34	21.12	0.05
<i>Butea monosperma</i>	3.59	8.45	0.12	5.30	0.16
<i>Azadirachta indica</i>	2.24	5.63	0.06	3.00	0.22
<i>Flacourtia indica</i>	1.79	4.23	0.01	1.78	0.32
<i>Dichrostachys cinerea</i>	1.79	4.23	0.01	1.77	0.32
<i>Catunaregam spinosa</i>	4.04	8.45	0.02	3.70	0.18
<i>Schrebera swietenoides</i>	1.35	1.41	0.07	2.12	2.13
<i>Lannea coromandelica</i>	4.04	11.27	0.13	6.13	0.10
<i>Schrebera swietenoides</i>	0.45	1.41	0.01	0.68	0.71
<i>Ficus arnottiana</i>	0.45	1.41	0.01	0.61	0.71
<i>Manilkara hexandra</i>	5.38	14.08	0.17	7.94	0.09
<i>Morinda pubescens</i>	0.45	1.41	0.01	0.62	0.71
<i>Tectona grandis</i>	40.82	59.15	1.07	46.86	0.04
<i>Terminalia alata</i>	1.35	2.82	0.05	1.90	0.53
<i>Boswellia serrata</i>	3.59	8.45	0.18	6.20	0.16

Species	Density/ha	Frequency %	TBA	IVI	A/F Ratio
<i>Holoptelia integrifolia</i>	2.69	8.45	0.04	3.49	0.12
<i>Dalbergia latifolia</i>	1.79	5.63	0.03	2.42	0.18
<i>Bombax ceiba</i>	0.90	2.82	0.04	1.54	0.36
<i>Diospyros melanoxylon</i>	12.56	23.94	0.22	13.60	0.07
<i>Schleichera oleosa</i>	1.35	4.23	0.01	1.57	0.24
<i>Ficus glomerata</i>	5.83	14.08	0.19	8.52	0.09
<i>Ehretia laevis</i>	1.79	5.63	0.01	2.09	0.18
<i>Moringa oleifera</i>	1.35	4.23	0.07	2.69	0.24

A.4. *Tectona grandis*-*Wrightia tinctoria* Community

This vegetation type occurred in the western part of the Gir WLS. Approximately 55 different tree species were found in this community. *Tectona grandis* (124.43), *Wrightia tinctoria* (29.12) were the dominant species. The other constant species with more than 25% occurrence values were *Tectona grandis* (98.21), *Wrightia tinctoria* (44.39), and *Acacia catechu* (27.80). The total tree density of this vegetation type was 238.64/ha. The diversity value is 2.30 and richness, evenness values were 55, 0.57. Details of distribution of vegetation sampling plots within this communities and species composition, status are given in Figure 3.11 and Table 3.8.

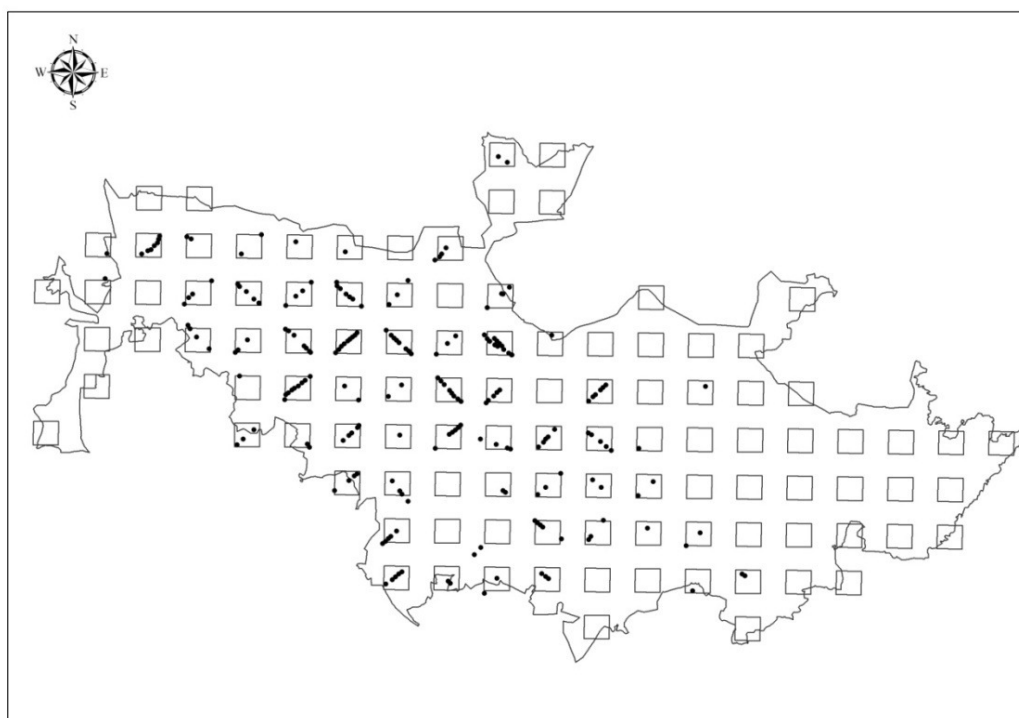


Figure 3.11: Distribution of Vegetation sampling plots classified in *Tectona grandis-Wrightia tinctoria* Community

Table 3.8: Species composition and status in *Tectona grandis-Wrightia tinctoria* Community

Species	Density/ha	Frequency %	TBA	IVI	A/F ratio
<i>Bauhinia racemosa</i>	0.14	0.45	0.00	0.21	2.23
<i>Polyalthia longifolia</i>	0.14	0.45	0.00	0.22	2.23
<i>Bauhinia purpurea</i>	5.71	15.25	0.07	7.48	0.08
<i>Emblca officinalis</i>	1.57	3.59	0.03	2.05	0.38
<i>Phyllanthus emblica</i>	3.14	8.52	0.08	4.97	0.14
<i>Tamarindus indica</i>	0.14	0.45	0.01	0.35	2.23
<i>Asparagus dumosus</i>	0.29	0.45	0.01	0.37	4.46
<i>Sapindus emarginatus</i>	0.14	0.45	0.00	0.19	2.23
<i>Acacia sp</i>	5.28	13.90	0.13	8.02	0.09
<i>Terminalia bellirica</i>	1.29	3.59	0.05	2.29	0.31
<i>Acacia nilotica</i>	1.14	2.24	0.01	1.27	0.71
<i>Aegle marmelos</i>	3.00	6.28	0.04	3.59	0.24
<i>Pterocarpus marsupium</i>	1.14	3.59	0.04	2.01	0.28
<i>Ziziphus mauritiana</i>	0.86	2.69	0.01	1.22	0.37
<i>Grewia tiliiaefolia</i>	8.00	19.28	0.10	9.96	0.07
<i>Anogeissus latifolia</i>	0.14	0.45	0.00	0.21	2.23
<i>Wrightia tinctoria</i>	24.28	44.39	0.43	29.12	0.04

Species	Density/ha	Frequency %	TBA	IVI	A/F ratio
<i>Grewia tenax</i>	0.14	0.45	0.00	0.19	2.23
<i>Cassia fistula</i>	1.43	4.48	0.02	2.10	0.22
<i>Acacia senegal</i>	0.57	1.79	0.02	0.99	0.56
<i>Zizyphus xylopyra</i>	1.71	4.48	0.01	2.11	0.27
<i>Haldinia cordifolia</i>	0.14	0.45	0.09	1.74	2.23
<i>Acacia leucophloea</i>	4.57	13.00	0.09	6.78	0.08
<i>Syzygium cumini</i>	0.14	0.45	0.02	0.57	2.23
<i>Holarrhena pubescens</i>	3.00	6.73	0.02	3.41	0.21
<i>Sterculia urens</i>	2.14	5.83	0.17	5.42	0.20
<i>Mallotus philippinensis</i>	1.57	3.59	0.04	2.28	0.38
<i>Derris indica</i>	0.14	0.45	0.00	0.21	2.23
<i>Capparis sepiaria</i>	0.14	0.45	0.00	0.19	2.23
<i>Acacia catechu</i>	15.42	27.80	0.29	18.66	0.06
<i>Butea monosperma</i>	2.14	6.73	0.04	3.38	0.15
<i>Azadirachta indica</i>	0.29	0.90	0.01	0.46	1.12
<i>Dichrostachys cinerea</i>	0.14	0.45	0.00	0.18	2.23
<i>Catunaregam spinosa</i>	1.29	3.59	0.01	1.69	0.31
<i>Lannea coromandelica</i>	5.14	12.11	0.11	7.24	0.11
<i>Schrebera seietenioides</i>	0.43	1.35	0.01	0.65	0.74
<i>Ficus arnottiana</i>	0.14	0.45	0.00	0.25	2.23
<i>Grewia asiatica</i>	0.14	0.45	0.00	0.23	2.23
<i>Manilkara hexandra</i>	2.43	5.83	0.09	4.06	0.22
<i>Morinda pubescens</i>	1.29	4.04	0.05	2.36	0.25
<i>Tectona grandis</i>	114.25	98.21	2.93	124.43	0.04
<i>Terminalia alata</i>	6.43	17.94	0.15	9.94	0.06
<i>Boswellia serrata</i>	3.28	6.73	0.13	5.43	0.23
<i>Delonix elata</i>	0.14	0.45	0.00	0.20	2.23
<i>Holoptelia integrifolia</i>	0.57	1.79	0.01	0.95	0.56
<i>Dalbergia latifolia</i>	2.43	6.73	0.05	3.70	0.17
<i>Bombax ceiba</i>	0.14	0.45	0.00	0.21	2.23
<i>Diospyros melanoxylon</i>	7.43	18.39	0.19	11.06	0.07
<i>Ficus racemosa</i>	0.14	0.45	0.01	0.32	2.23
<i>Schleichera oleosa</i>	0.43	0.90	0.02	0.75	1.67
<i>Ficus glomerata</i>	1.43	4.48	0.04	2.39	0.22
<i>Ficus benghalensis</i>	0.43	1.35	0.04	1.15	0.74
<i>Ehretia laevis</i>	0.14	0.45	0.00	0.20	2.23
<i>Moringa oleifera</i>	0.14	0.45	0.00	0.19	2.23
<i>Gmelina arborea</i>	0.29	0.90	0.00	0.42	1.12

A.5. *Terminalia alata-Tectona grandis-Lannea coromandelica-Acacia catechu* Community

This type of vegetation community occurred in the National Park and central east part of the Gir WLS. More than 40 different species of trees were found in this vegetation type. *Terminalia alata* (39.52), *Tectona grandis* (38.13), *Lannea coromandelica* (30.64), *Acacia catechu* (31.00) were the dominant trees of this community. The other constant species with the occurrence value more than 25% were *Terminalia alata* (47.06), *Tectona grandis* (45.10), *Lannea coromandelica* (45.10), *Acacia catechu* (43.14), *Boswellia serrata* (27.45), *Emblica officinalis* (25.49). The total tree density of this vegetation type was 179.84/ha. The diversity value was 2.92 and the richness, evenness values were 40, 0.79. Details of distribution of vegetation sampling plots within this communities and species composition, status are given in Figure 3.12 and Table 3.9.

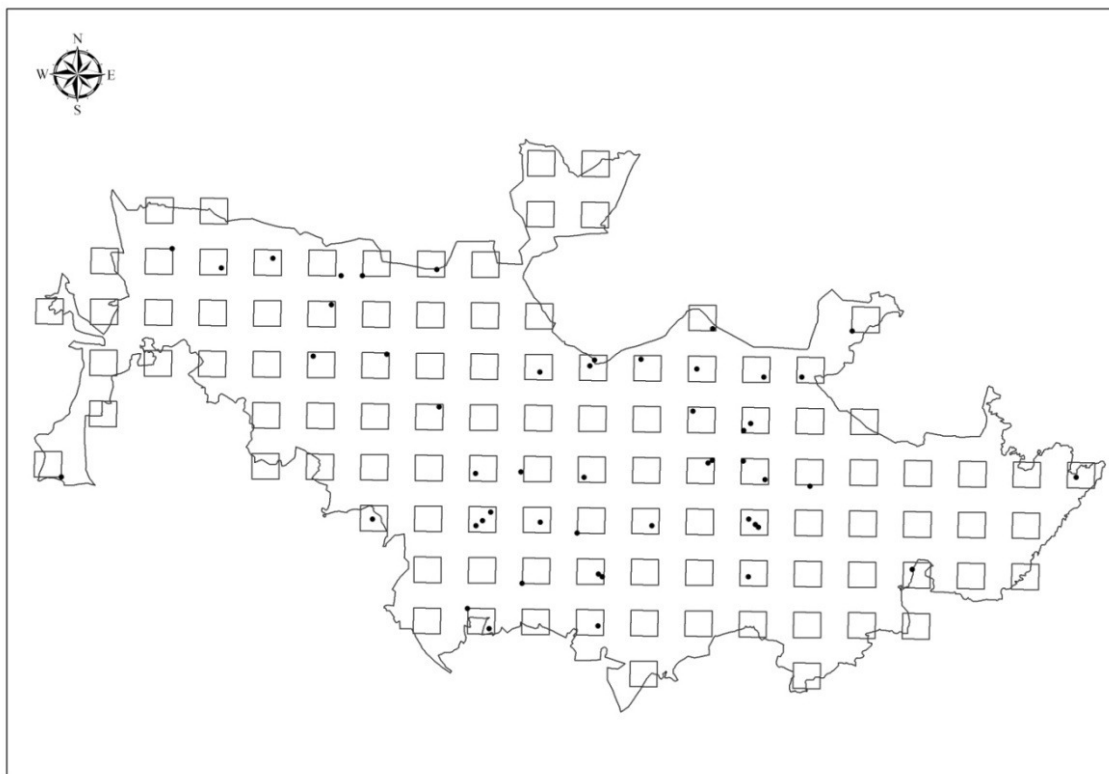


Figure 3.12: Distribution of Vegetation sampling plots classified in *Terminalia alata-Tectona grandis-Lannea coromandelica-Acacia catechu* Community

Table 3.9: Species composition and status in *Terminalia alata*-*Tectona grandis*-*Lannea coromandelica*-*Acacia catechu* Community

Species	Density/ha	Frequency %	TBA	IVI	A/F ratio
<i>Emblica officinalis</i>	12.49	25.49	0.19	17.99	0.06
<i>Phyllanthus emblica</i>	3.75	7.84	0.07	5.67	0.19
<i>Tamarindus indica</i>	1.25	3.92	0.03	2.48	0.26
<i>Sapindus emarginatus</i>	0.62	1.96	0.00	0.95	0.51
<i>Acacia sp</i>	2.50	7.84	0.03	4.07	0.13
<i>Terminalia bellirica</i>	2.50	7.84	0.21	8.64	0.13
<i>Aegle marmelos</i>	2.50	1.96	0.06	3.45	2.04
<i>Pterocarpus marsupium</i>	0.62	1.96	0.00	0.95	0.51
<i>Ziziphus mauritiana</i>	0.62	1.96	0.01	1.03	0.51
<i>Grewia tiliiaefolia</i>	2.50	5.88	0.02	3.22	0.23
<i>Anogeissus latifolia</i>	1.87	3.92	0.04	2.90	0.38
<i>Wrightia tinctoria</i>	1.25	3.92	0.02	2.08	0.26
<i>Grewia tenax</i>	1.25	3.92	0.01	1.97	0.26
<i>Cassia fistula</i>	0.62	1.96	0.01	0.99	0.51
<i>Acacia senegal</i>	1.25	1.96	0.02	1.74	1.02
<i>Acacia leucophloea</i>	0.62	1.96	0.05	2.17	0.51
<i>Syzygium cumini</i>	0.62	1.96	0.00	0.88	0.51
<i>Holarrhena pubescens</i>	0.62	1.96	0.00	0.88	0.51
<i>Sterculia urens</i>	0.62	1.96	0.00	0.91	0.51
<i>Mallotus philippinensis</i>	1.87	5.88	0.03	3.18	0.17
<i>Mitragyna parvifolia</i>	2.50	7.84	0.11	6.15	0.13
<i>Derris indica</i>	0.62	1.96	0.01	1.14	0.51
<i>Acacia catechu</i>	23.10	43.14	0.30	31.00	0.04
<i>Butea monosperma</i>	1.25	3.92	0.01	1.93	0.26
<i>Azadirachta indica</i>	1.25	3.92	0.02	2.29	0.26
<i>Flacourtia indica</i>	1.87	5.88	0.01	2.85	0.17
<i>Dichrostachys cinerea</i>	0.62	1.96	0.00	0.95	0.51
<i>Catunaregam spinosa</i>	5.00	5.88	0.07	5.90	0.45
<i>Lannea coromandelica</i>	16.24	45.10	0.42	30.64	0.03
<i>Manilkara hexandra</i>	7.49	21.57	0.34	17.84	0.05
<i>Tectona grandis</i>	23.73	45.10	0.55	38.13	0.04
<i>Terminalia alata</i>	24.35	47.06	0.58	39.52	0.03
<i>Boswellia serrata</i>	13.74	27.45	0.39	24.18	0.06
<i>Dalbergia latifolia</i>	2.50	7.84	0.07	5.05	0.13
<i>Diospyros melanoxylon</i>	10.62	21.57	0.26	17.74	0.07
<i>Schleichera oleosa</i>	1.25	3.92	0.03	2.35	0.26
<i>Ficus glomerata</i>	1.87	5.88	0.02	2.98	0.17
<i>Ficus benghalensis</i>	0.62	1.96	0.02	1.22	0.51
<i>Sterculia villosa</i>	0.62	1.96	0.00	0.92	0.51
<i>Gmelina arborea</i>	0.62	1.96	0.01	1.08	0.51

A.6. *Boswellia serrata*-*Acacia catechu* Community

This type of vegetation community occurred in the central part of the Gir WLS. 20 different tree species were observed in this community. *Boswellia serrata* (106.51), *Acacia catechu* (41.21) were the two most dominant species of trees found in this vegetation type. Other constant species with the occurrence value more than 25% were *Boswellia serrata* (90.32), *Acacia catechu* (54.84), *Manilkara hexandra* (29.03), and *Acacia spp* (29.03). The total tree density was 173.62/ha. The diversity value was 2.20 and the richness, evenness values were 20, 0.73. Details of distribution of vegetation sampling plots within this communities and species composition, status are given in Figure 3.13 and Table 3.10.

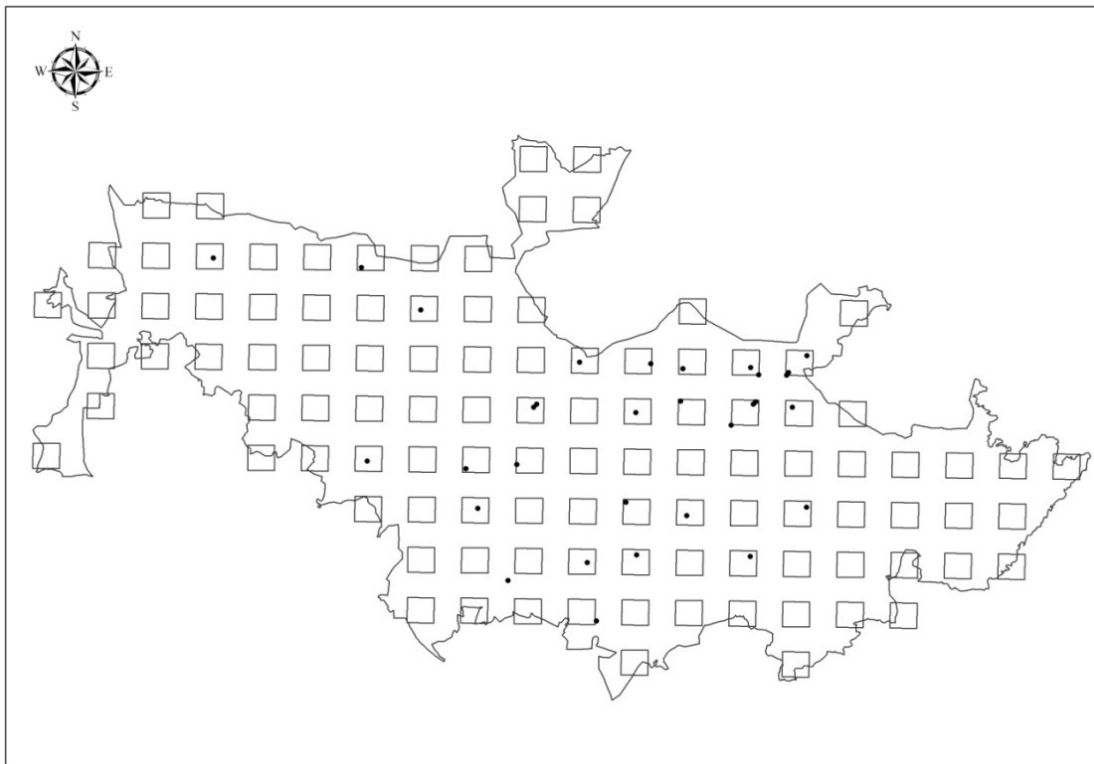


Figure 3.13: Distribution of Vegetation sampling plots classified in *Boswellia serrata*-*Acacia catechu* Community

Table 3.10: Species composition and status in *Boswellia serrata*-*Acacia catechu* Community

Species	Density/Ha	Frequency %	TBA	IVI	A/F Ratio
<i>Bauhinia purpurea</i>	6.16	16.13	0.09	10.66	0.07
<i>Emblica officinalis</i>	2.05	6.45	0.03	3.76	0.16
<i>Phyllanthus emblica</i>	1.03	3.23	0.01	1.75	0.31
<i>Asparagus dumosus</i>	4.11	3.23	0.03	4.15	1.24
<i>Acacia sp</i>	11.30	29.03	0.17	19.30	0.04
<i>Ziziphus mauritiana</i>	1.03	3.23	0.01	1.75	0.31
<i>Anogeissus latifolia</i>	1.03	3.23	0.01	1.72	0.31
<i>Wrightia tinctoria</i>	9.25	19.35	0.18	15.70	0.08
<i>Sterculia urens</i>	4.11	12.90	0.39	16.05	0.08
<i>Mitragyna parvifolia</i>	1.03	3.23	0.03	2.29	0.31
<i>Acacia catechu</i>	28.77	54.84	0.33	41.21	0.03
<i>Lannea coromandelica</i>	7.19	16.13	0.15	12.78	0.09
<i>Ficus arnottiana</i>	2.05	3.23	0.09	4.34	0.62
<i>Manilkara hexandra</i>	12.33	29.03	0.34	24.41	0.05
<i>Tectona grandis</i>	6.16	16.13	0.18	12.89	0.07
<i>Terminalia alata</i>	5.14	16.13	0.08	9.62	0.06
<i>Boswellia serrata</i>	65.75	90.32	1.64	106.51	0.03
<i>Dalbergia latifolia</i>	3.08	9.68	0.05	5.94	0.10
<i>Diospyros melanoxylon</i>	1.03	3.23	0.06	3.06	0.31
<i>Moringa oleifera</i>	1.03	3.23	0.02	2.13	0.31

A.7. *Acacia catechu*- *Ziziphus mauritiana* Community

This type of vegetation community occurred in the central, northern, central east part of the Gir WLS. More than 40 tree species were observed in this community. *Acacia catechu* (131.10), *Ziziphus mauritiana* (25.10) were the dominant species of this community. *Acacia catechu* (100.00) was considered to be the most constant species of this community. The total tree density of this vegetation type was 176.61/ha. The diversity value was 2.16 and the richness, evenness values were 40, 0.58. Details of distribution of vegetation sampling plots within this communities and species composition, status are given in Figure 3.14 and Table 3.11.

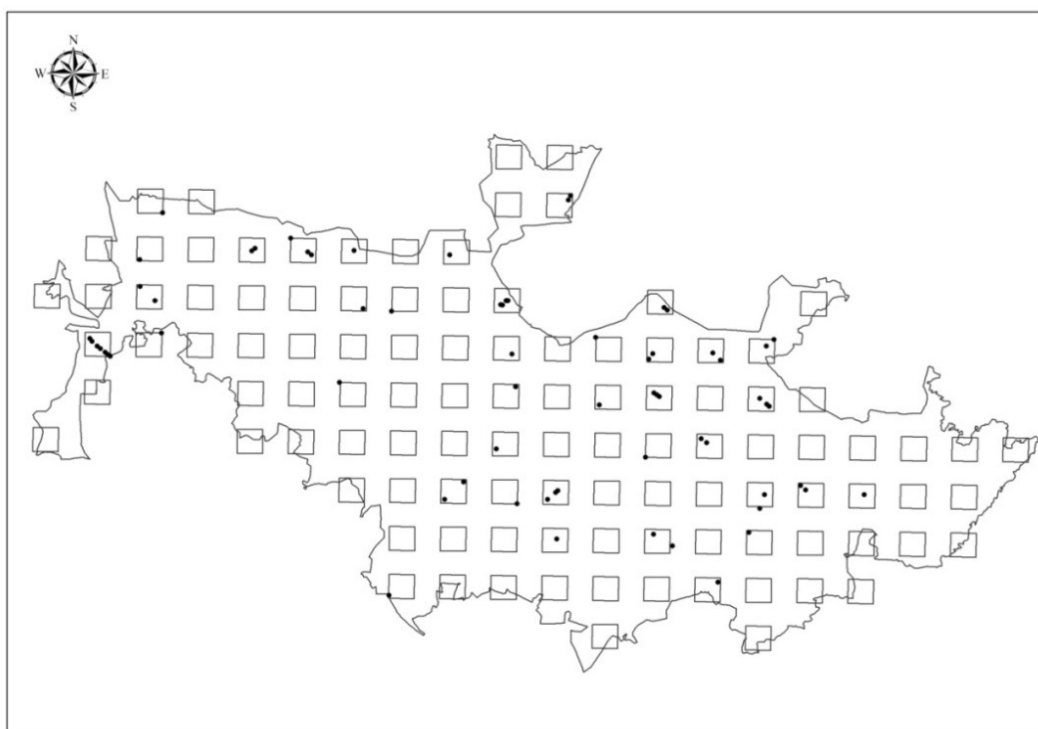


Figure 3.14: Distribution of Vegetation sampling plots classified in *Acacia catechu-Ziziphus mauritiana* Community

Table 3.11: Species status and composition in *Acacia catechu-Ziziphus mauritiana* Community

Species	Density/Ha	Frequency %	TBA	IVI	A/F Ratio
<i>Bauhinia purpurea</i>	1.45	4.55	0.03	3.27	0.22
<i>Embllica officinalis</i>	2.90	6.06	0.08	6.42	0.25
<i>Acacia sp</i>	0.97	3.03	0.01	2.09	0.33
<i>Hymenodyctyon orixense</i>	1.45	1.52	0.03	2.33	1.98
<i>Terminalia bellirica</i>	0.48	1.52	0.04	2.11	0.66
<i>Acacia nilotica</i>	3.38	7.58	0.06	6.47	0.18
<i>Aegle marmelos</i>	0.97	1.52	0.02	1.65	1.32
<i>Ziziphus mauritiana</i>	16.41	22.73	0.23	25.10	0.10
<i>Grewia tiliiaefolia</i>	1.45	3.03	0.02	2.70	0.50
<i>Anogeissus latifolia</i>	3.86	7.58	0.04	6.31	0.21
<i>Wrightia tinctoria</i>	0.97	3.03	0.01	1.85	0.33
<i>Grewia tenax</i>	0.48	1.52	0.00	0.91	0.66
<i>Acacia senegal</i>	1.45	4.55	0.03	3.46	0.22
<i>Zizyphus xylopyra</i>	2.41	7.58	0.02	4.71	0.13
<i>Haldinia cordifolia</i>	0.48	1.52	0.01	1.13	0.66
<i>Acacia leucophloea</i>	2.41	7.58	0.04	5.28	0.13
<i>Balanites aegyptiaca</i>	0.97	3.03	0.00	1.77	0.33

Species	Density/Ha	Frequency %	TBA	IVI	A/F Ratio
<i>Holarrhena pubescens</i>	0.97	3.03	0.01	1.93	0.33
<i>Mallotus philippinensis</i>	0.48	1.52	0.01	1.01	0.66
<i>Derris indica</i>	0.97	1.52	0.01	1.28	1.32
<i>Acacia catechu</i>	95.06	100.00	1.24	131.10	0.03
<i>Butea monosperma</i>	1.93	6.06	0.05	4.86	0.17
<i>Azadirachta indica</i>	0.48	1.52	0.01	1.20	0.66
<i>Dichrostachys cinerea</i>	1.93	4.55	0.02	3.26	0.29
<i>Catunaregam spinosa</i>	0.97	3.03	0.01	1.85	0.33
<i>Lannea coromandelica</i>	2.90	9.09	0.06	6.85	0.11
<i>Grewia asiatica</i>	0.97	3.03	0.01	1.99	0.33
<i>Parkinsonia aculeata</i>	0.48	1.52	0.02	1.36	0.66
<i>Manilkara hexandra</i>	0.48	1.52	0.11	4.58	0.66
<i>Morinda pubescens</i>	2.90	6.06	0.06	5.76	0.25
<i>Tectona grandis</i>	4.83	10.61	0.17	12.28	0.13
<i>Terminalia alata</i>	5.79	15.15	0.13	13.15	0.08
<i>Boswellia serrata</i>	5.31	9.09	0.20	13.00	0.20
<i>Delonix elata</i>	0.48	1.52	0.03	1.77	0.66
<i>Holoptelia integrifolia</i>	0.48	1.52	0.01	1.08	0.66
<i>Dalbergia latifolia</i>	0.48	1.52	0.01	0.99	0.66
<i>Diospyros melanoxylon</i>	2.90	6.06	0.06	5.70	0.25
<i>Schleichera oleosa</i>	0.97	1.52	0.02	1.65	1.32
<i>Ficus glomerata</i>	1.45	4.55	0.04	3.66	0.22
<i>Moringa oleifera</i>	0.97	3.03	0.02	2.13	0.33

A.8. *Anogeissus latifolia*, *Boswellia serrata* Community

This type of vegetation community occurred in the Gir east part. 34 different tree species belong to this community. The dominant species were *Anogeissus latifolia* (89.37), *Boswellia serrata* (26.54). The other constant species with the more than 25 % occurrence value were *Anogeissus latifolia* (86.67), *Acacia catechu* (30.67), and *Acacia nilotica* (32.00). The total tree density was 189.79/ha whereas the diversity value was 2.43. The richness and evenness values were 34 and 0.69. Details of distribution of vegetation sampling plots within this communities and species composition, status are given in Figure 3.15 and Table 3.12.

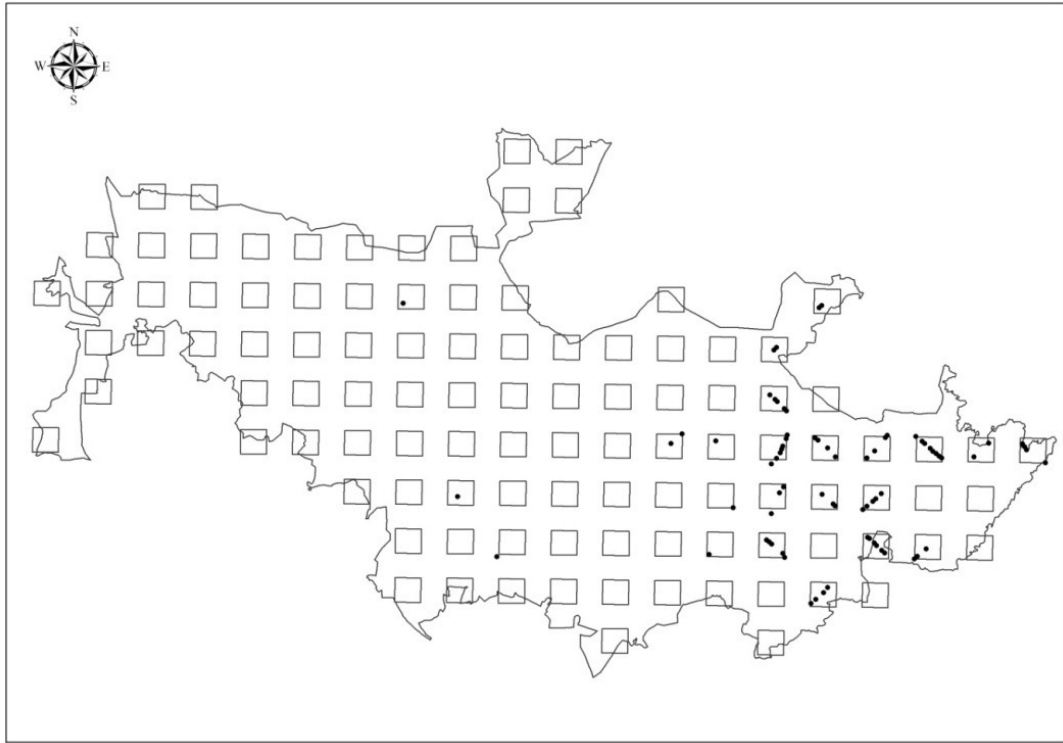


Figure 3.15: Distribution of Vegetation sampling plots classified in *Anogeissus latifolia*, *Boswellia serrata* Community

Table 3.12: Species composition and status in *Anogeissus latifolia*, *Boswellia serrata* Community

Species	Density/Ha	Frequency %	TBA	IVI	A/F Ratio
<i>Bauhinia purpurea</i>	1.70	5.33	0.03	3.20	0.19
<i>Emblica officinalis</i>	1.28	4.00	0.02	2.36	0.25
<i>Acacia sp</i>	0.43	1.33	0.00	0.63	0.75
<i>Acacia nilotica</i>	15.74	32.00	0.28	24.06	0.05
<i>Ziziphus mauritiana</i>	2.55	8.00	0.02	4.15	0.13
<i>Anogeissus latifolia</i>	70.21	86.67	1.13	89.37	0.03
<i>Wrightia tinctoria</i>	2.55	8.00	0.08	5.52	0.13
<i>Grewia tenax</i>	0.43	1.33	0.00	0.65	0.75
<i>Acacia senegal</i>	15.32	22.67	0.26	20.84	0.09
<i>Acacia leucophloea</i>	6.81	18.67	0.10	11.19	0.06
<i>Balanites aegyptiaca</i>	0.85	2.67	0.00	1.27	0.38
<i>Sterculia urens</i>	0.43	1.33	0.03	1.36	0.75
<i>Mitragyna parvifolia</i>	0.43	1.33	0.00	0.63	0.75
<i>Acacia catechu</i>	16.17	30.67	0.32	24.95	0.05
<i>Phoenix sylvestris</i>	0.85	1.33	0.02	1.20	1.50
<i>Butea monosperma</i>	2.98	9.33	0.04	5.25	0.11
<i>Azadirachta indica</i>	2.55	8.00	0.06	4.96	0.13
<i>Dichrostachys cinerea</i>	2.13	5.33	0.02	3.11	0.23

Species	Density/Ha	Frequency %	TBA	IVI	A/F Ratio
<i>Catunaregam spinosa</i>	0.43	1.33	0.00	0.63	0.75
<i>Lannea coromandelica</i>	4.68	12.00	0.14	9.41	0.10
<i>Schrebera seietenioides</i>	0.85	2.67	0.02	1.64	0.38
<i>Grewia asiatica</i>	0.43	1.33	0.02	1.03	0.75
<i>Manilkara hexandra</i>	8.09	21.33	0.22	15.75	0.06
<i>Morinda pubescens</i>	1.28	4.00	0.02	2.26	0.25
<i>Tectona grandis</i>	1.28	4.00	0.03	2.59	0.25
<i>Terminalia alata</i>	6.81	18.67	0.18	13.16	0.06
<i>Boswellia serrata</i>	13.19	24.00	0.52	26.54	0.07
<i>Delonix elata</i>	0.85	2.67	0.20	6.11	0.38
<i>Holoptelia integrifolia</i>	0.43	1.33	0.01	0.80	0.75
<i>Dalbergia latifolia</i>	0.43	1.33	0.01	0.73	0.75
<i>Annona squamosa</i>	0.43	1.33	0.00	0.65	0.75
<i>Diospyros melanoxylon</i>	6.38	16.00	0.18	12.36	0.08
<i>Schleichera oleosa</i>	0.43	1.33	0.01	0.86	0.75
<i>Moringa oleifera</i>	0.43	1.33	0.01	0.80	0.75

A.9. *Terminalia alata*, *Acacia leucophloea*, *Anogeissus latifolia*, *Butea monosperma* Community

This type of community occurred in the North-Eastern part of the Gir WLS. 27 different species were found in this vegetation type. *Terminalia alata* (80.08), *Acacia leucophloea* (52.75), *Anogeissus latifolia* (37.46), *Butea monosperma* (27.55) were the dominant species found in this community. Other constant species with more than 25% of occurrence values were *Terminalia alata* (59.09), *Acacia leucophloea* (56.82), *Anogeissus latifolia* (25.00), *Butea monosperma* (25.00). The total tree density of this vegetation type was 127.39/ha. The diversity value was 2.40. The richness and evenness values were 27 and 0.73. Details of distribution of vegetation sampling plots within this communities and other statistical analysis are given in Figure 3.16 and Table 3.13.

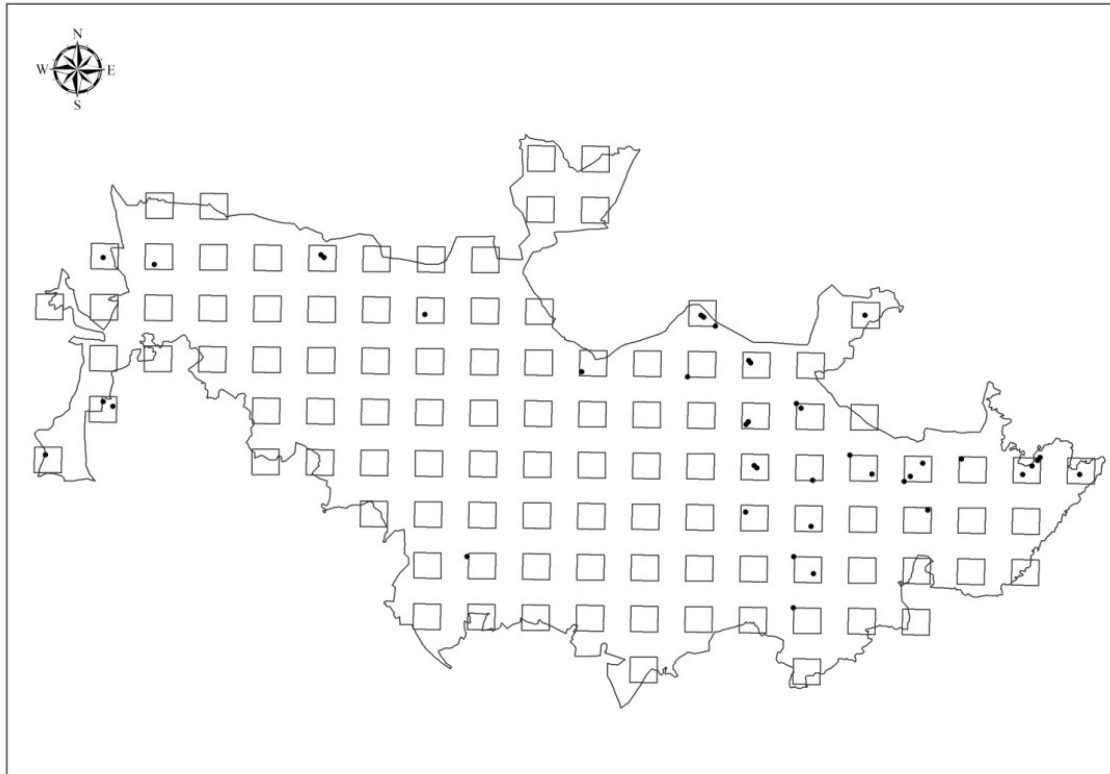


Figure 3.16: Distribution of Vegetation sampling plots classified in *Terminalia alata*, *Acacia leucophloea*, *Anogeissus latifolia*, *Butea monosperma* Community

Table 3.13: Species composition and status in *Terminalia alata*, *Acacia leucophloea*, *Anogeissus latifolia*, *Butea monosperma* Community

Species	Density/Ha	Frequency %	TBA	IVI	A/F Ratio
<i>Bauhinia purpurea</i>	1.45	4.55	0.02	3.94	0.22
<i>Emblca officinalis</i>	3.62	11.36	0.04	9.00	0.09
<i>Acacia nilotica</i>	9.41	18.18	0.09	18.66	0.09
<i>Ziziphus mauritiana</i>	2.17	4.55	0.03	4.67	0.33
<i>Grewia tiliiaefolia</i>	1.45	4.55	0.01	3.32	0.22
<i>Anogeissus latifolia</i>	19.54	25.00	0.25	37.46	0.10
<i>Grewia tenax</i>	0.72	2.27	0.01	1.66	0.44
<i>Acacia senegal</i>	1.45	4.55	0.02	4.07	0.22
<i>Zizyphus xylopyra</i>	0.72	2.27	0.01	2.14	0.44
<i>Acacia leucophloea</i>	23.89	56.82	0.26	52.75	0.02
<i>Balanites aegyptiaca</i>	0.72	2.27	0.01	1.87	0.44
<i>Mallotus philippinensis</i>	0.72	2.27	0.02	2.29	0.44
<i>Capparis sepiaria</i>	0.72	2.27	0.01	1.66	0.44
<i>Acacia catechu</i>	3.62	11.36	0.05	9.59	0.09

Species	Density/Ha	Frequency %	TBA	IVI	A/F Ratio
<i>Butea monosperma</i>	10.13	25.00	0.20	27.55	0.05
<i>Flacourtia indica</i>	1.45	4.55	0.01	3.32	0.22
<i>Dichrostachys cinerea</i>	0.72	2.27	0.00	1.58	0.44
<i>Lannea coromandelica</i>	0.72	2.27	0.01	1.87	0.44
<i>Manilkara hexandra</i>	5.07	11.36	0.07	11.55	0.12
<i>Tectona grandis</i>	0.72	2.27	0.03	2.89	0.44
<i>Terminalia alata</i>	32.57	59.09	0.64	80.08	0.03
<i>Boswellia serrata</i>	0.72	2.27	0.00	1.58	0.44
<i>Holoptelia integrifolia</i>	0.72	2.27	0.01	2.11	0.44
<i>Dalbergia latifolia</i>	1.45	4.55	0.02	3.74	0.22
<i>Annona squamosa</i>	0.72	2.27	0.00	1.58	0.44
<i>Schleichera oleosa</i>	0.72	2.27	0.01	2.14	0.44
<i>Ficus benghalensis</i>	1.45	2.27	0.10	6.93	0.88

A.10. *Acacia nilotica*, *Butea monosperma*, *Ziziphus mauritiana* Community

This type of vegetation community was mainly found in the eastern part of the Gir WLS. The dominant species were *Acacia nilotica* (103.49), *Butea monosperma* (34.55), *Ziziphus mauritiana* (26.03). The constant species with the more than 25% occurrence values were *Acacia nilotica* (88.41), *Butea monosperma* (39.13), *Ziziphus mauritiana* (30.43). The total tree density of this vegetation type was 196.62/ha. The diversity value was 2.29. The richness and evenness values were 30 and 0.67. Details of distribution of vegetation sampling plots within this communities and species composition, status are given in Figure 3.17 and Table 3.14.

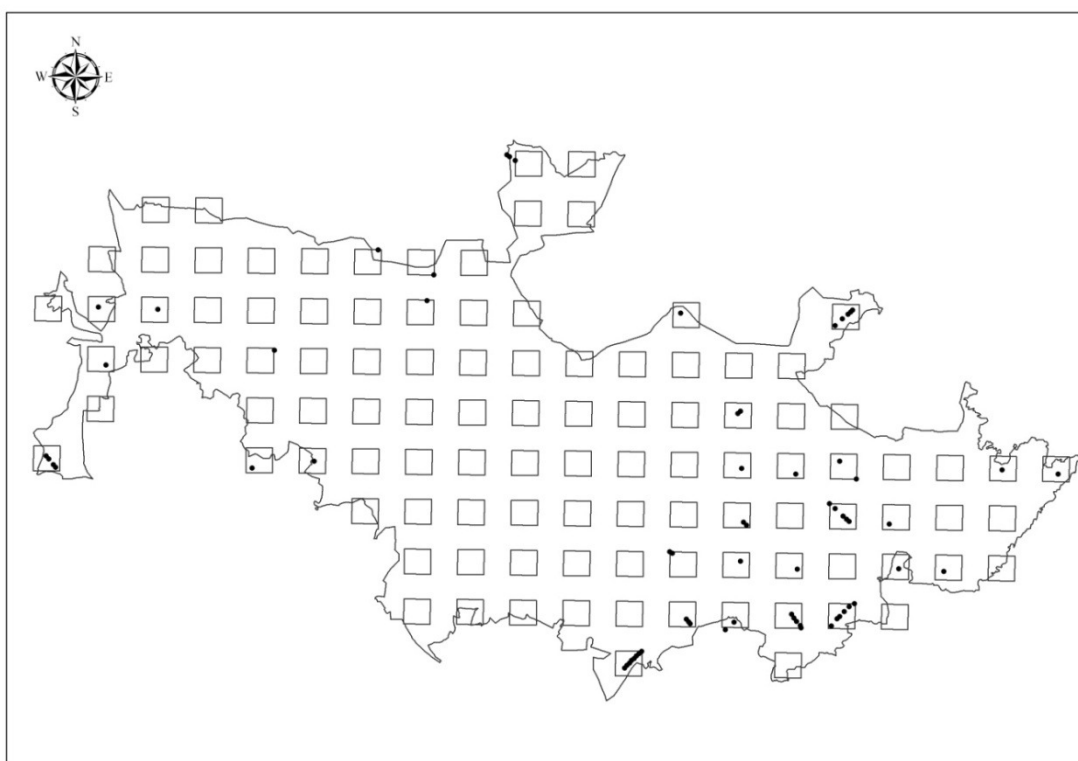


Figure 3.17: Distribution of Vegetation sampling plots classified in *Acacia nilotica*, *Butea monosperma*, *Ziziphus mauritiana* Community

Table 3.14: Species status and composition in *Acacia nilotica*, *Butea monosperma*, *Ziziphus mauritiana* Community

Species	Density/Ha	Frequency %	TBA	IVI	A/F Ratio
<i>Bauhinia purpurea</i>	4.62	14.49	0.05	7.96	0.07
<i>Acacia sp</i>	1.38	4.35	0.06	3.66	0.23
<i>Acacia nilotica</i>	77.08	88.41	1.42	103.49	0.03
<i>Ziziphus mauritiana</i>	18.46	30.43	0.28	26.03	0.06
<i>Grewia tiliiaefolia</i>	0.92	2.90	0.02	1.84	0.35
<i>Anogeissus latifolia</i>	4.15	7.25	0.17	8.89	0.25
<i>Cassia fistula</i>	0.46	1.45	0.00	0.76	0.69
<i>Acacia senegal</i>	15.23	24.64	0.31	23.41	0.08
<i>Acacia leucophloea</i>	9.23	24.64	0.14	15.78	0.05
<i>Balanites aegyptiaca</i>	0.46	1.45	0.00	0.76	0.69
<i>Syzygium cumini</i>	2.31	1.45	0.04	2.80	3.45
<i>Mallotus philippinensis</i>	0.46	1.45	0.01	1.02	0.69
<i>Capparis sepiaria</i>	0.46	1.45	0.01	0.91	0.69
<i>Acacia catechu</i>	6.92	13.04	0.08	9.68	0.13
<i>Phoenix sylvestris</i>	0.46	1.45	0.00	0.76	0.69
<i>Butea monosperma</i>	24.46	39.13	0.39	34.55	0.05
<i>Azadirachta indica</i>	6.00	14.49	0.13	10.82	0.09

Species	Density/Ha	Frequency %	TBA	IVI	A/F Ratio
<i>Flacourtia indica</i>	0.92	2.90	0.01	1.73	0.35
<i>Dichrostachys cinerea</i>	5.08	10.14	0.04	6.76	0.15
<i>Lannea coromandelica</i>	1.38	2.90	0.01	1.97	0.52
<i>Schrebera seietenioides</i>	0.92	2.90	0.01	1.65	0.35
<i>Ficus arnottiana</i>	0.92	2.90	0.23	7.35	0.35
<i>Manilkara hexandra</i>	0.92	2.90	0.02	1.88	0.35
<i>Morinda pubescens</i>	1.38	4.35	0.02	2.62	0.23
<i>Tectona grandis</i>	0.46	1.45	0.01	0.87	0.69
<i>Terminalia alata</i>	1.85	4.35	0.04	3.22	0.31
<i>Delonix elata</i>	0.46	1.45	0.01	1.02	0.69
<i>Holoptelia integrifolia</i>	2.31	7.25	0.06	4.93	0.14
<i>Diospyros melanoxylon</i>	5.54	13.04	0.10	9.33	0.10
<i>Ficus benghalensis</i>	1.38	4.35	0.06	3.58	0.23

B. Remote Sensing Approach

B.1. Image Classification: The unsupervised classification of the remotely-sensed data indicated that a high degree of inter-class separability was not present and only a few useful groupings could be identified. To assess the degree of correspondence between the ecological classes identified by the TWINSPAN analysis and those identifiable in the remotely-sensed data attention was therefore focused on only the classification producing up to five groups. The first three groups essentially separated pure teak and teak associated forest classes from other forest classes. The other two forest classes were non teak bearing forest classes. The three teak bearing forest classes were Moist Teak forests; *Wrightia*-Teak forests where *Wrightia* were dominated than Teak, Teak-*Wrightia* forests where teak dominated *Wrightia* presence. Other forest classes were *Boswellia-Acacia* forest, *Anogeissus-Boswellia-Acacia* forests.

The class groupings identified from the unsupervised classifications of the remotely sensed data therefore to some extent, resembled those derived from the TWINSPAN analysis. Together these results indicated that some 4-5 groups of forest classes were separable in the remotely sensed data set and that these groups corresponded broadly to the ecological groups identified from the TWINSPAN analysis. From both sets of classifications it was apparent that Teak associated forests, *Boswellia-Acacia* forests, and *Anogeissus-Boswellia-Acacia* forests were separable to reasonable levels but further inter-class discrimination was unlikely.

On the basis of these results an attempt was made to map ecological groups which might assist conservation at the site from the Landsat TM image. For this it was, however necessary to accommodate the other classes present in the image. These classes were water bodies, barren land, grassland, agricultural or burnt area, and fallow land. All these classes were spectrally distinct from the others and so could be classified accurately. The 10 classification categories (Figure 3.18) consisted of:

1. Moist Teak Forests,
2. *Wrightia*- Teak- *Acacia nilotica* Forests,
3. Teak- *Wrightia* Forests,
4. *Accacia-Boswellia* Forests,
5. *Anogeissus-Boswellia-Accacia* Forests,
6. Bare land representing soil that has almost no vegetation,
7. Water, where this class mainly represents water stored behind dams,
8. Agriculture land including crop and fallow land as well as rainfed and irrigated land.
9. Grassland.
10. Fallow land.

The dominant and constant species of each of these vegetation types are given in Table 3.15.

Table 3.15: Dominant and constant species of major vegetation types in Gir protected area

Forest Type	Dominant Species (Based on IVI Value)	Constant Species (Frequency>25%)
Moist Teak	<i>Tectona grandis, Ziziphus mauritiana</i>	<i>Ziziphus mauritiana, Tectona grandis, Acacia nilotica, Acacia catechu, Acacia auriculiformes</i>
Wrightia-Teak- Acacia nilotica	<i>Wrightia tinctoria, Tectona grandis, Acacia nilotica, Butea monosperma, Ziziphus mauritiana</i>	<i>Wrightia tinctoria, Acacia nilotica, Tectona grandis, Butea monosperma, Ziziphus mauritiana, Acacia catechu, Bauhinia purpurea, Ziziphus mauritiana</i>
Teak-Wrightia	<i>Tectona grandis, Wrightia tinctoria</i>	<i>Tectona grandis, Wrightia tinctoria, Acacia catechu</i>
Acacia-Boswellia	<i>Acacia catechu, Boswellia serrata, Terminalia alata, Tectona grandis, Lannea coromandelica, Ziziphus mauritiana</i>	<i>Acacia catechu, Boswellia serrata, Terminalia alata, Tectona grandis, Lannea coromandelica, Emblica officinalis, Manilkara hexandra</i>
Anogeissus-Boswellia-Accacia	<i>Anogeissus latifolia, Boswellia serrata, Terminalia alata, Acacia leucophloea, Anogeissus latifolia, Butea monosperma</i>	<i>Anogeissus latifolia, Acacia catechu, Acacia nilotica, Terminalia alata, Acacia leucophloea, Anogeissus latifolia, Butea monosperma</i>

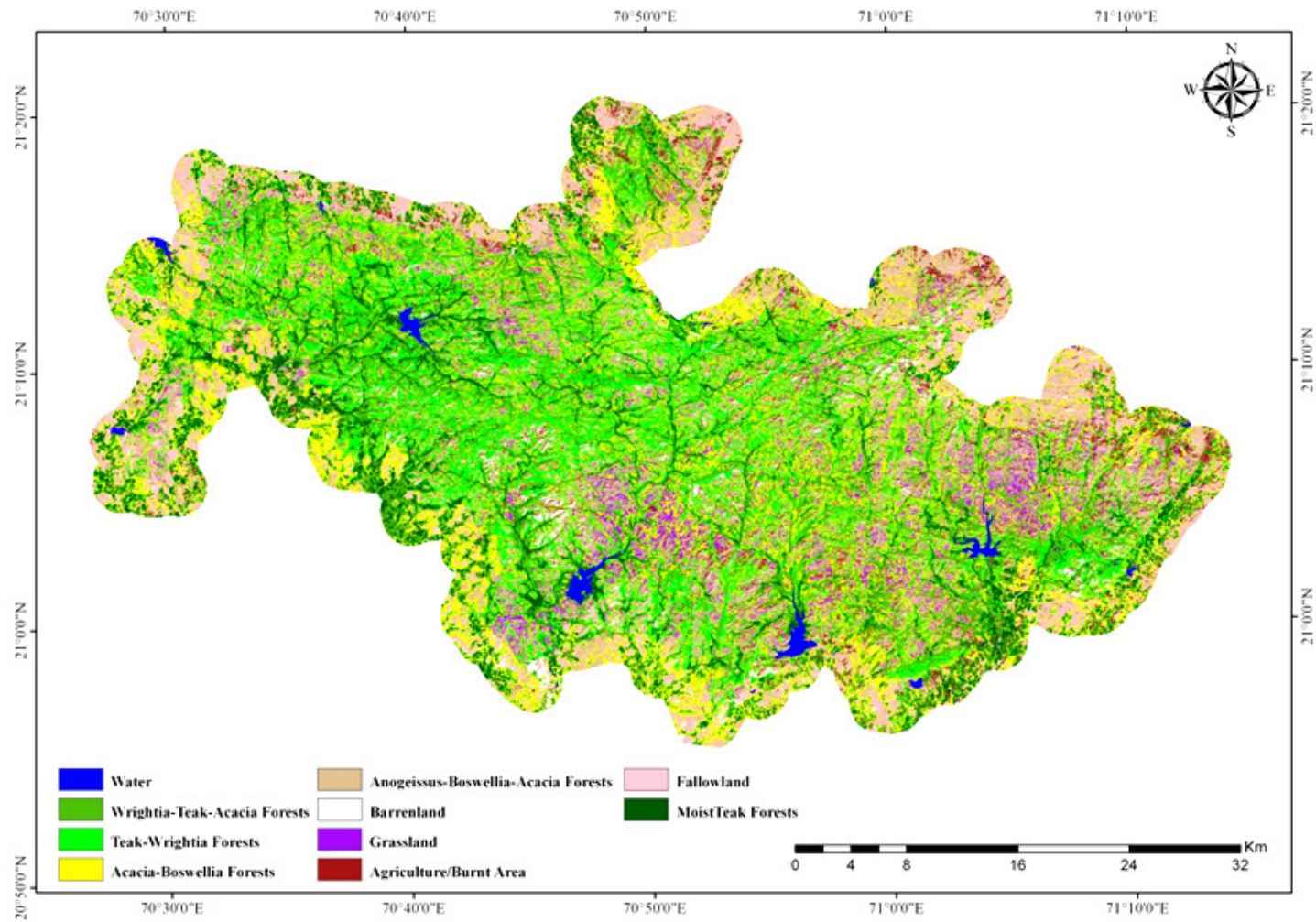


Figure 3.18: Land use/cover pattern of Gir protected area, based on Landsat TM image of 2009

Detailed descriptions of the above five vegetation types and other land use/cover classes are as follows:

1. Moist Teak Forests

This forest type was represented by two different communities of trees, *Ziziphus mauritiana*-*Tectona grandis* community and *Tectona grandis* Community. More than 50 different trees were found in this forest type. Teak (*Tectona grandis*) was the most dominant tree species of this forest type and distribution of teak was governed by the precipitation. Other dominating trees found were *Ziziphus mauritiana*, *Acacia catechu*, *Acacia auriculiformes*, and *Acacia nilotica*. Throughout its stretch from west to east within the sanctuary, teak was dominating in the western and central part of the sanctuary. *Ziziphus mauritiana* was distributed all over the stretch while presence of *Acacia auriculiformes* was mainly found in central-eastern part of the sanctuary and *Acacia nilotica* was dominating in eastern part of the sanctuary where teak was totally absent.

2. Wrightia-Teak- Acacia nilotica Forests

This forest type was represented by two different vegetation communities, *Wrightia tinctoria*-*Tectona grandis* Community and *Acacia nilotica*, *Butea monosperma*, *Ziziphus mauritiana* community. More than 50 tree species were found in this forest type. While the *Wrightia tinctoria*-*Tectona grandis* Community mainly found in the western and central part of the sanctuary, the *Acacia nilotica*, *Butea monosperma*, *Ziziphus mauritiana* Community was distributed in the eastern part of the sanctuary. In the western part, this forest type was dominated by *Wrightia tinctoria*, *Tectona grandis* and other constant species were *Acacia catechu* and *Bauhinia purpurea*. In the eastern part, this forest type was dominated by *Acacia nilotica*, *Butea monosperma*. *Ziziphus mauritiana* was found all over this forest type.

3. Teak-Wrightia Forests

This forest type was spread throughout the sanctuary but more area in western part of the sanctuary was under this forest type than in the eastern part. Approximately 55 different tree species occurred in this forest type. *Tectona grandis*, *Wrightia tinctoria* were the dominant

species in the western and central part of the sanctuary while in the eastern part of the sanctuary *Acacia catechu* was the most common species.

4. *Acacia-Boswellia* Forests

This forest type was patchily distributed in National Park area, central part of the sanctuary, fringes area and represented by three different vegetation communities namely, *Terminalia alata-Tectona grandis-Lannea coromandelica-Acacia catechu* community, *Boswellia serrata-Acacia catechu* community, *Acacia catechu-Ziziphus mauritiana* community. More than 50 different tree species occurred in this forest type. In the national park area and few places in the central part of the sanctuary *Terminalia alata*, *Tectona grandis*, *Lannea coromandelica* were the dominant tree species while *Boswellia serrata*, *Acacia catechu* dominated in the central part of the sanctuary and in the central-east portion of the sanctuary *Acacia catechu* was the dominant species.

5. *Anogeissus-Boswellia-Accacia* Forests

This forest type was found in the eastern and north eastern portion of the sanctuary and it was patchily distributed. More than 40 different tree species were found in this forest type. The most dominating tree species of this forest type was *Anogeissus latifolia*, distribution of which was totally restricted to the eastern Gir. Other dominating species were *Boswellia serrata*, *Terminalia alata*, *Butea monosperma*. Different *Acacia spp* were also found in this forest type and they were spatially distributed separately from west to east. *Acacia catechu* was distributed all over the sanctuary while distribution of *Acacia leucophloea* was mostly found in the central part and *Acacia nilotica* was found in the eastern part of the sanctuary.

Kappa statistic used to measure classification accuracy was found to be .82 (Table 3.16), showing significant amount of agreement between the actual and classified vegetation map.

Table 3.16: Kappa statistics showing accuracy of the image classification

Symmetric Measures					
		Value	Asymp. Std. Error ^a	Approx. T ^b	Approx. Sig.
Measure of Agreement	Kappa	.821	.032	21.654	.000
N of Valid Cases		200			

C. Effect of changing grain size on landscape indices

With the change in grain size through spatial aggregation, the responses of the 17 class-level metrics fell into two general groups: metrics showing consistent scaling relations (Type I) and metrics showing unpredictable scaling behavior (Type II). The first group was further divided into those showing both consistent and robust scaling relations (Type IA) and those showing consistent but less robust scaling relations (Type IB).

Of the 17 metrics, 5 belonged to Type IA: Number of Patches (NP) (Figure 3.19), Patch Density (PD) (Figure 3.20), Total Edge (TE) (Figure 3.21), Edge Density (ED) (Figure 3.22), and Landscape Shape Index (LSI) (Figure 3.23); and 7 to Type IB: Largest Patch Index (LPI) (Figure 3.26), Square Pixel Index (SqP), Mean Patch Size (MPS) (Figure 3.24), Patch Size Standard Deviation (PSSD) (Figure 3.27), Patch Size Coefficient of Variation (PSCV) (Figure 3.25), Area-Weighted Mean Shape Index (AWMSI), and Area-Weighted Mean Fractal Dimension (AWMFD).

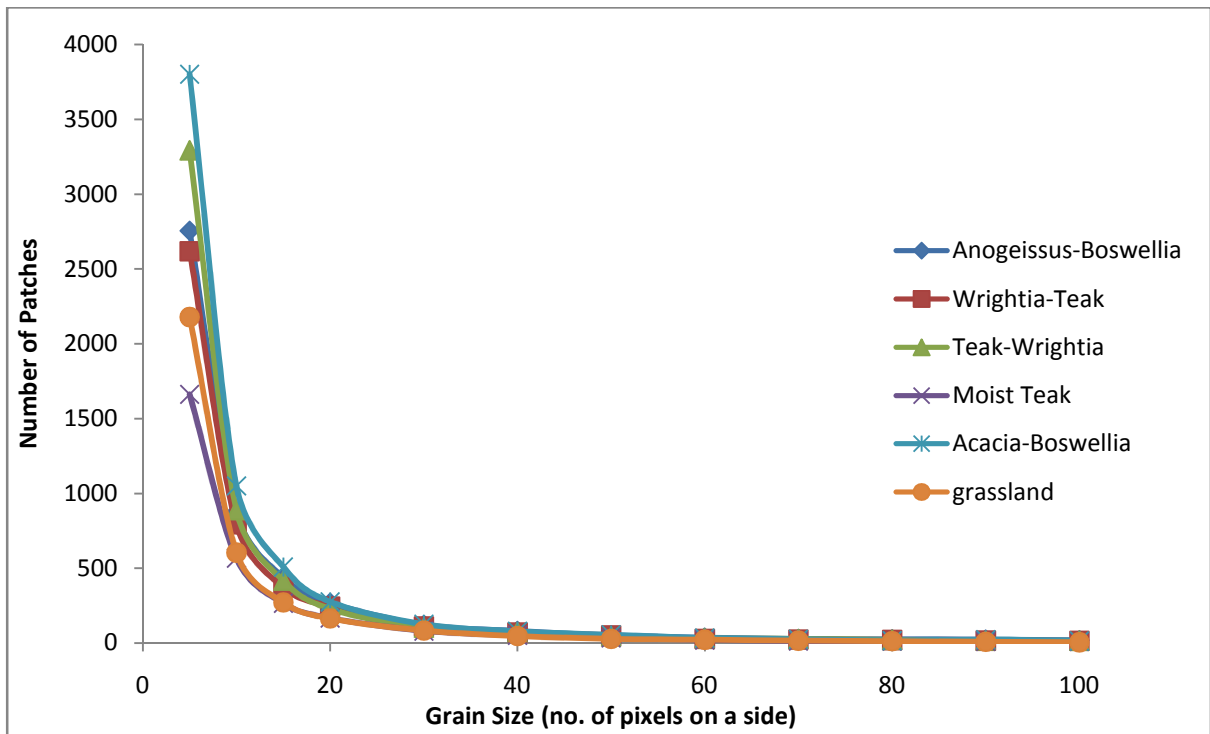


Figure 3.19: Scalograms showing effects of changing grain size on landscape metrics (Type IA scaling behavior), Relation of number of patches with the change in grain size

Type II metrics included: Class Area (CA), Percent of Landscape (CA %), Mean Patch Shape Index (MSI), Mean Patch Fractal Dimension (MPFD), and Double-Log Fractal Dimension (DLFD). The values of these indices varied unpredictably with changing grain size, resulting in response curves of various forms – relatively constant, monotonic changes, or fluctuations. This suggested that these metrics were highly sensitive to the specific patterns of the landscapes under study, and thus general scaling relations were not possible to derive.

Some metrics showed similar scalograms because of mathematical similarity. For example, CA and CA (%) showed exactly the same pattern because $CA (\%) = CA/A$, where A is the landscape area which is a constant in the case of changing grain size.

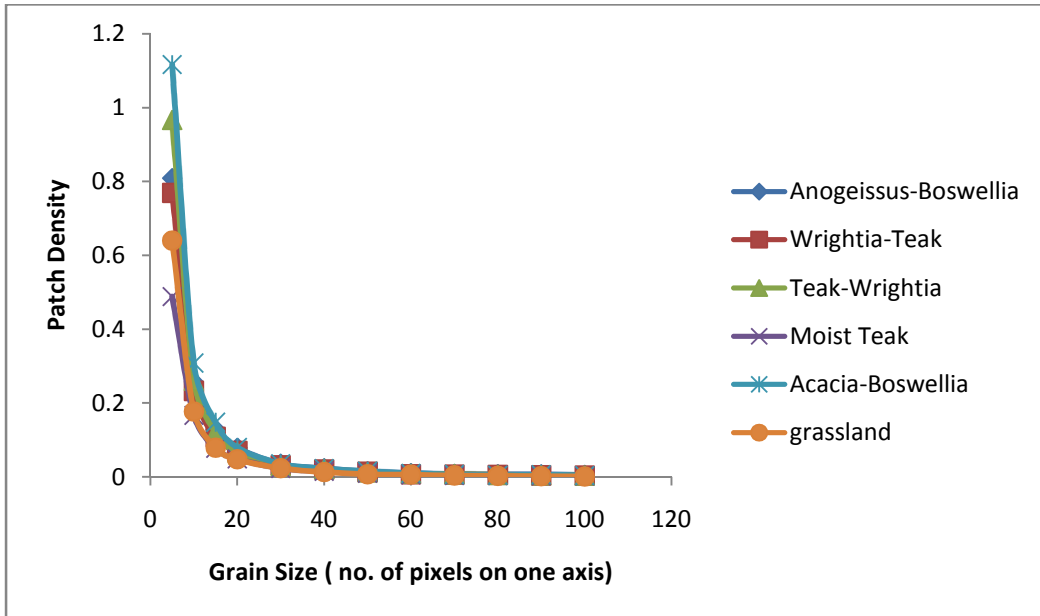


Figure 3.20: Scalograms showing effects of changing grain size on landscape metrics (Type IA scaling behavior), Relation of patch density with the change in grain size

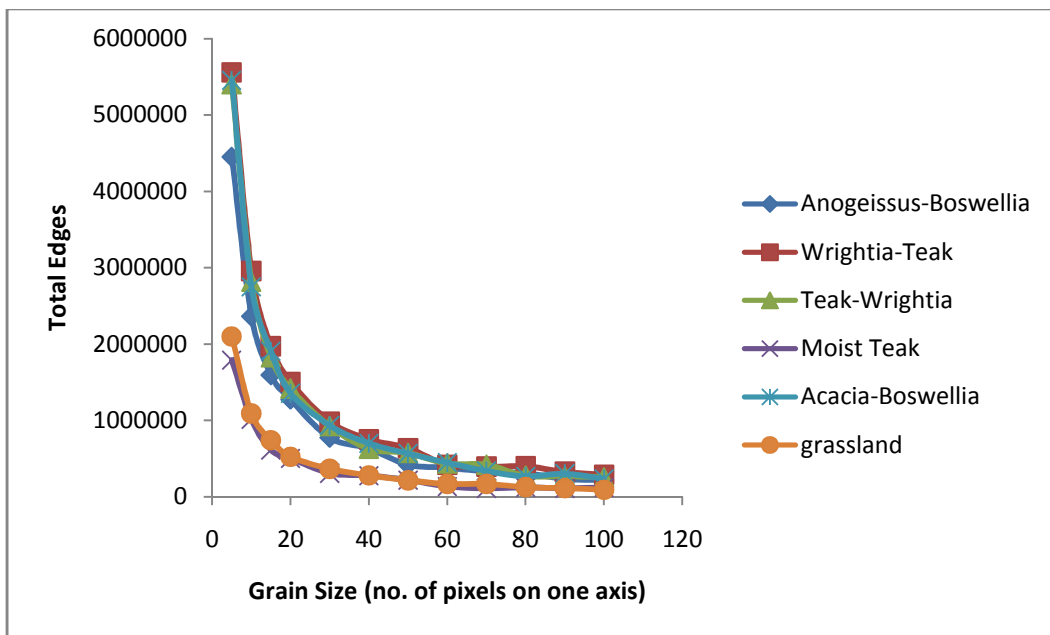


Figure 3.21: Scalograms showing effects of changing grain size on landscape metrics (Type IA scaling behavior), Relation of total edges with the change in grain size

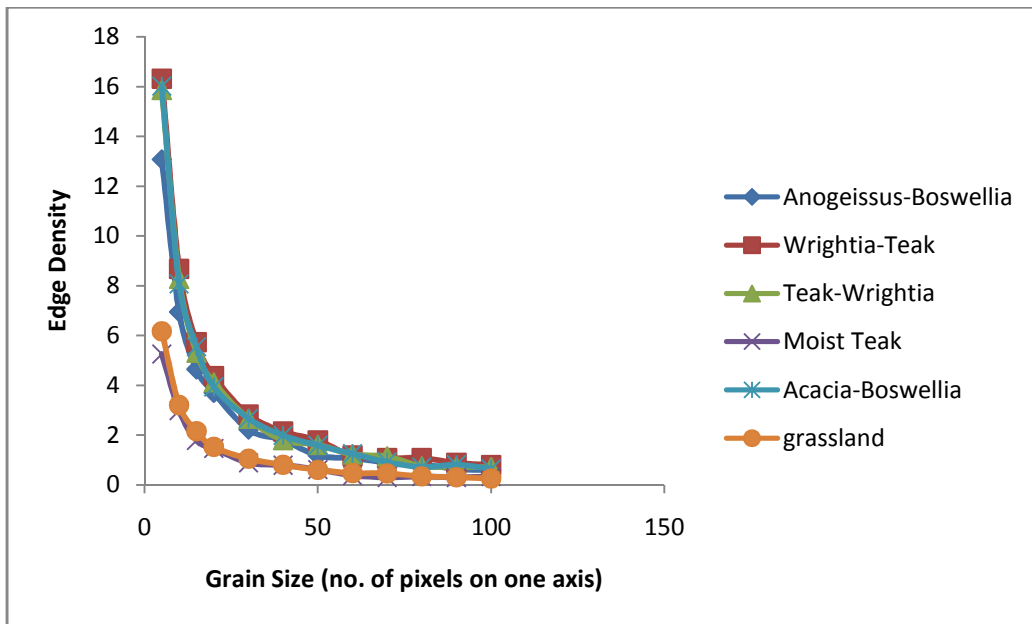


Figure 3.22: Scalograms showing effects of changing grain size on landscape metrics (Type IA scaling behavior), Relation of edge density with the change in grain size

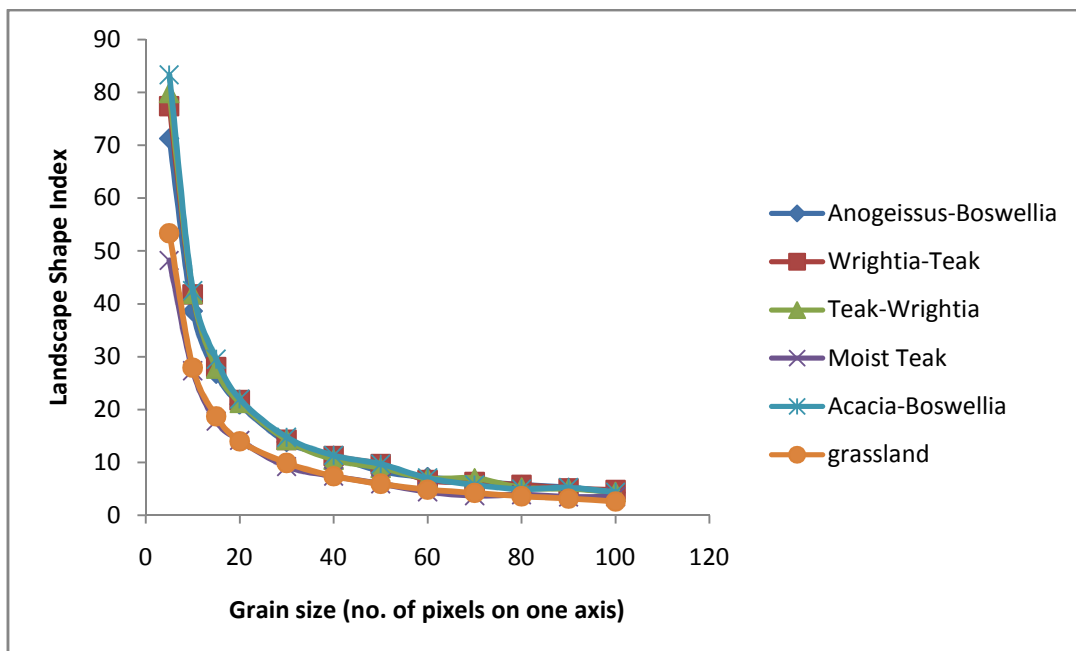


Figure 3.23 Scalograms showing effects of changing grain size on landscape metrics (Type IA scaling behavior), Relation of landscape shape index with the change in grain size

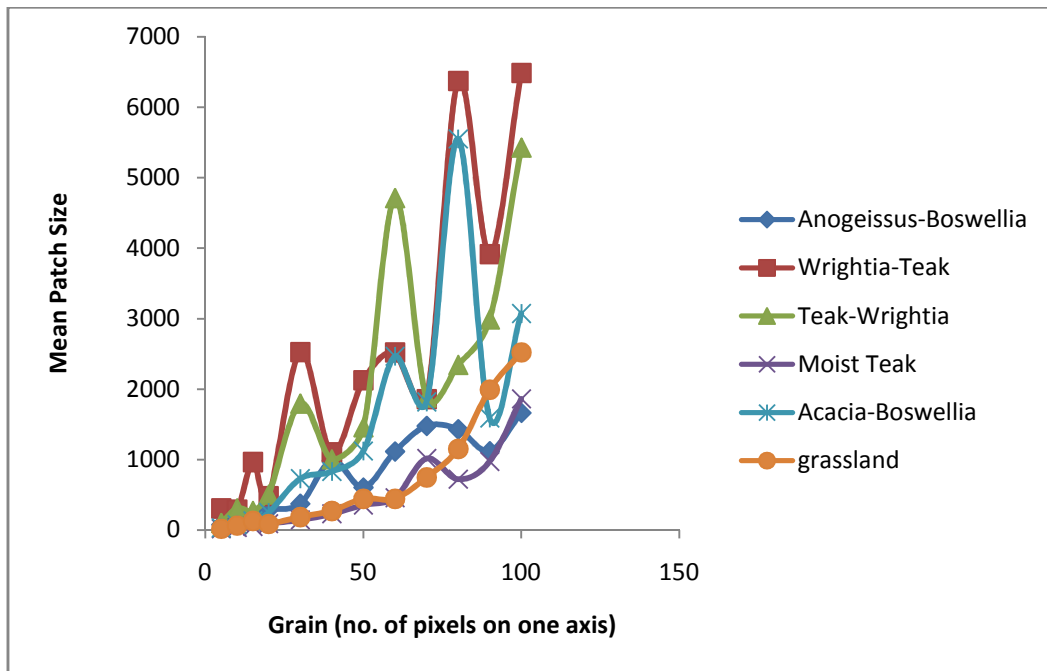


Figure 3.24: Scalograms showing effects of changing grain size on landscape metrics (Type IB scaling behavior), Relation of mean patch size with the change in grain size

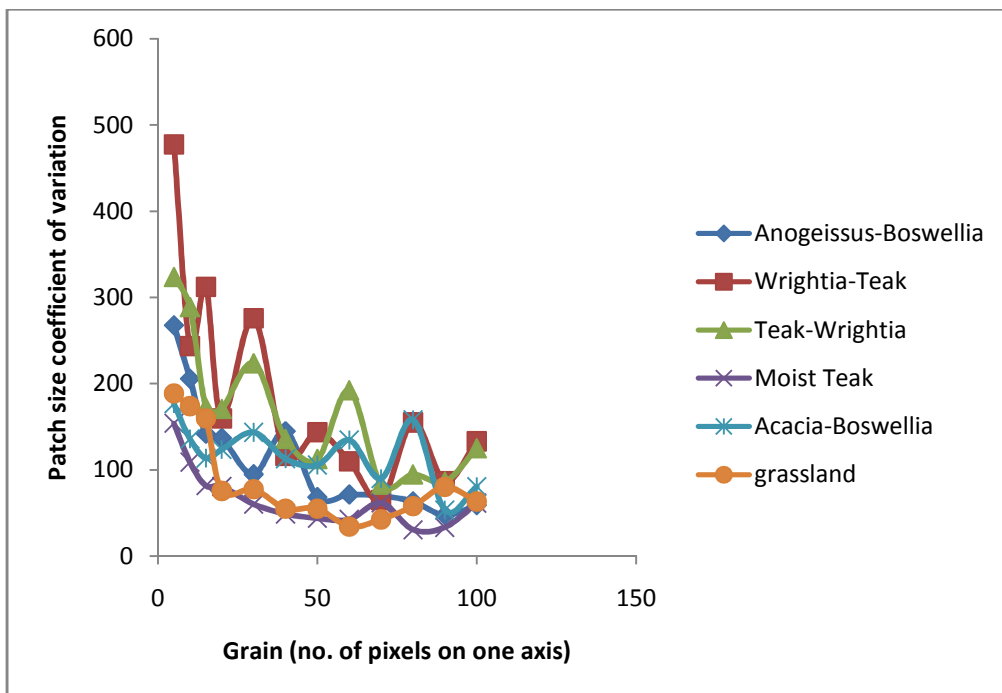


Figure 3.25: Scalograms showing effects of changing grain size on landscape metrics (Type II scaling behavior), Relation of patch size coefficient of variation with the change in grain size

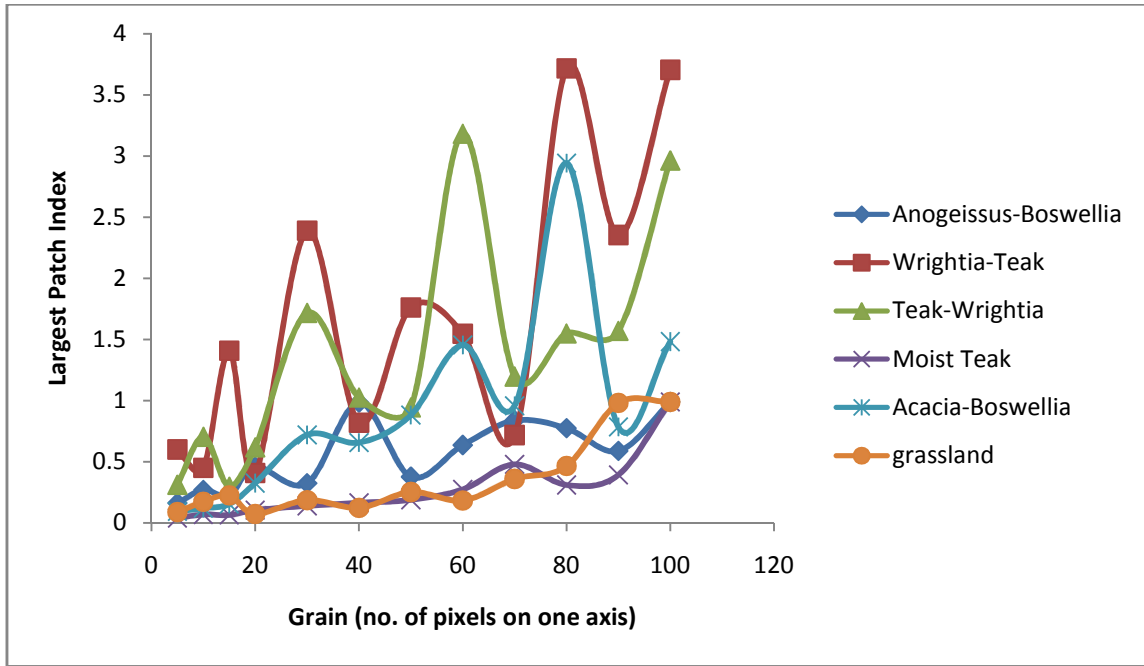


Figure 3.26: Scalograms showing effects of changing grain size on landscape metrics (Type II scaling behavior), Relation of largest patch index with the change in grain size

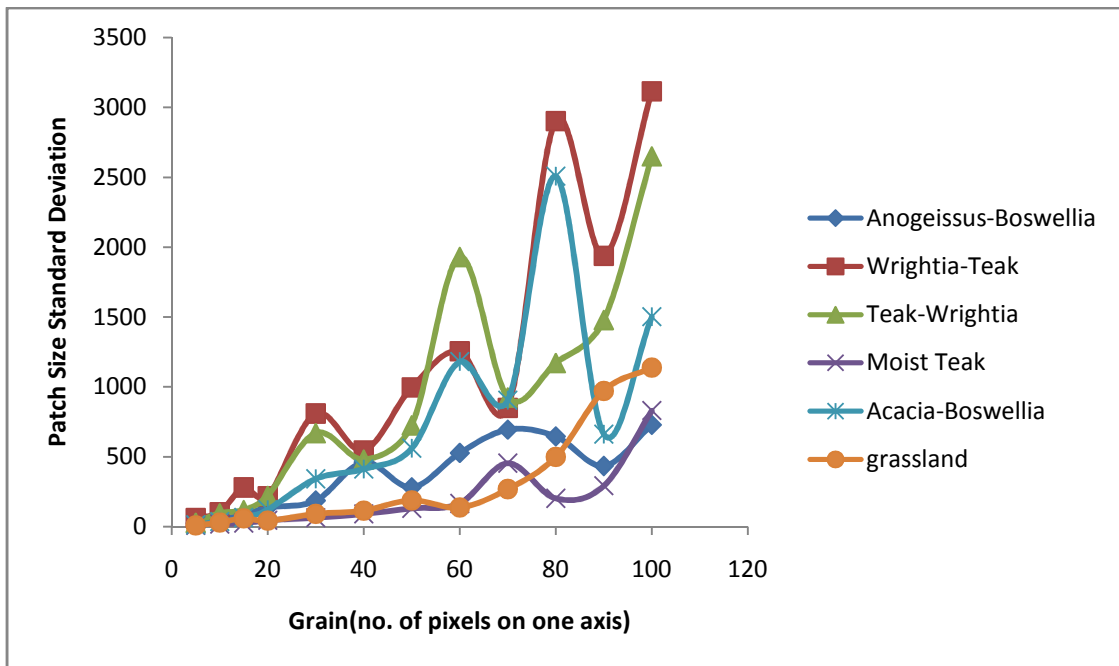


Figure 3.27: Scalograms showing effects of changing grain size on landscape metrics (Type II scaling behavior), Relation of patch size standard deviation with the change in grain size

On the other hand, MSI and MPFD exhibited similar patterns in response to both changing grain size and extent because of their mathematical similarity:

$$\frac{MSI}{MPFD} = \frac{\sum_{i=1}^N \left(\frac{0.25P_i}{\sqrt{a_i}} \right)}{\sum_{i=1}^N \left(\frac{\ln(0.25P_i)}{\ln\sqrt{a_i}} \right)}$$

That is, while MSI is simply a perimeter-area ratio normalized based on the square shape and averaged over all patches, MPFD requires that both the numerator and denominator be log-transformed before the summation for the entire class across the landscape.

Similarly, the scale response curves of AWMSI and AWMFD resembled each other because:

$$AWMSI/AWMFD = \frac{\sum_{i=1}^N \left[\left(\frac{0.25P_i}{\sqrt{a_i}} \right) \left(\frac{a_i}{A} \right) \right]}{\sum_{i=1}^N \left(\frac{\ln(0.25P_i)}{\ln\sqrt{a_i}} \left(\frac{a_i}{A} \right) \right)}$$

In this case, AWMFD seemed more preferable because it was able to suppress somewhat the abrupt large fluctuations that occurred with AWMSI, so that a comparison between patch types became more feasible.

3.8 Discussion

LANDSAT provided scientific researchers with a valuable imagery that can be utilized for detecting terrestrial land cover conditions, and tracking land vegetation, agricultural activity, urban growth, and surface hydrology (Cohen and Goward 2004) and proved to be valuable in qualitative and quantitative terrestrial land cover changes (Coppin and Bauer 1994, Collins and Woodcock 1996, Woodcock et al. 2001). Multiple images were used in identifying and monitoring reclaimed lands (Pax-Lenney et al. 1996) and helped to avoid confusion between fallow and barren lands in delta regions. Multivariate discriminant function was applied to Landsat images which allowed six major land cover classes to be classified to an accuracy of 87% (Wood and Foody 1989). The spectral and segment based classification levels of Landsat TM was merged to generate a land use maps with enhanced accuracy up to 78% (Guindon et al. 2004)

Despite the heterogeneity of the landscape from the precipitation and vegetation point of view, especially from west to east corner of the Gir protected area, Landsat images were

found to be efficient in delineating the major landscape classes. Accuracy levels of more than 80% are considered adequate enough for reliable classification of land use/cover types (Sabins 1997). Importantly, areas of different classes on the study area were estimated. The use of automated signature building method such as the Isodata algorithm based on statistical differences might fail to capture the difference between different vegetation communities but supplemented by representative field data improved the ability to discriminate between these classes, but on the expense of higher cost. Other landscape elements can be easily discriminated using automated signature building algorithms. Thus collecting field data on these classes is not cost effective. Agricultural land extends from irrigated, rainfed areas followed by completely irrigated dryland farms. Mango orchards also extend over a wide range of landscapes with varying types of associated farming practices. Sometimes, the distinction between scrubland and forestland was vague where some of the so called forests are no more than a dense scrubland. On the other hand reflectance of old mango orchard and canopies of bigger trees were similar in reflectance increasing probability of classifying mango orchard as forestland. It was difficult to discriminate between pastureland and agricultural lands because of the marginal to semi dryness of the area while some agricultural lands especially fallow lands were being mistakenly identified as pastureland.

The final map assessment demonstrated satisfactory accuracy figures. An important validating element of the classification obtained was the result derived from the multivariate analysis of the phytosociological relevés. In all cases the various plant communities obtained through the TWINSpan classification corresponded to one of the final classes obtained in the digital classification. However even when the procedures proved adequate for the definition of broad land cover types along the study area, it should be noted that some floristic differences shown by the TWINSpan analysis were not represented on the final map include forest communities sharing similar to identical physiognomic characteristics. This problem already addressed by many authors in different sites (Singh 1987, Tuomisto et al. 1994) was due to the fact that satellite sensor systems measure reflected and emitted radiation from plants and from the ground surface (given a clean atmosphere) which may be shared by various vegetation types as long as they show a similar structure and plants show similar spectral signatures (Wilkie and Finn 1996).

The results of this study showed that changing grain size had significant effects on both the class- and landscape-level metrics. Overall, more metrics showed consistent scaling relations with changing grain size at both the class and landscape levels – indicating that effects of

changing spatial resolution are generally predictable. While the same metrics tended to behave similarly at the class level and the landscape level, the scale responses at the class level were much more variable. This study supports the gradually accepted concepts: there is no single “correct” or “optimal” scale for exemplifying spatial heterogeneity (Wu 2004), and comparison between landscapes using pattern indices must be based on the same spatial resolution and extent. Landscape metrics with simple scaling relations reflect those landscape features that can be extrapolated or interpolated across spatial scales. This study demonstrated that the spatial scale at which these patterns were quantified influenced the result and measurements made at different scales might not be comparable. Furthermore, qualitative and quantitative changes in measurements across spatial scales would differ according to how scale is defined. Thus the definition and methods of changing scale must always be explicitly stated. It is important to define the scale of ecological data in terms of both grains and extent. The identification of properties that do not change or change predictably across scales would simplify the extrapolation of measurements from fine scales to broad scales. However, although it may be possible to identify simple relationships between landscape parameters measured at different scales, the exact relationship varies across landscapes and does not permit extrapolation from one region to another.

Numerous ecological applications require information on the abundance and configuration of different habitat or cover types that is provided by class level metrics. The results verify that information is lost in coarser grained spatial data. In general, information about the less frequent land cover types is most easily lost, but the rate of loss depends upon their spatial arrangement. Because landscape metrics calculated at an inappropriate thematic resolution may lead to the so called "ecological fallacy" for correlation and regression analysis (Wu 2006, Buyantuyev and Wu 2007), it is critically important to identify an appropriate level of detail surely relevant for ecological processes of interest. The effects of varying pixel sizes was examined on several indices of diversity and evenness, and found that all of them were unresponsive to varying grain size (Wickham and Riitters 1995). As they suggested, it might be due to the fact that only a narrow range of grain sizes was measured (4 pixel sizes: 4X4 m², 12X12m², 28X28 m², and 80X80 m²). Landscape metric scalograms (the response curves of landscape metrics to changing grain size or extent), as an alternative of the single values of the metrics, should be used for characterizing, comparing, and monitoring landscape patterns. It seems that landscape metric scalograms are more likely to be useful for linking landscape pattern to ecological processes because both pattern and process in ecological systems often operate on multiple scales (Wu et al. 2002).

Chapter 4

**Variations in the Landscape Patterns
and Vegetation Cover between 1998 and
2009 in the Semi arid Gir Wildlife Sanctuary**

4.1 Introduction

Habitat loss and fragmentation beside other issues such as climate change, invasive species, disease, and overexploitation produce detrimental effects on species and ecosystems (Ewers and Didham 2006) and are the primary causes of declines in biological diversity worldwide (Wilcove et al. 1998). Hence, assessing and monitoring the state of the earth surface is a basic requirement for global change research (Committee on Global Change Research, National Research Council 1999, Lambin et al. 2001, Jung et al. 2006). In the last two decades, increased understanding of ongoing and emerging environmental issues and the prerequisite to attempt for sustainable management of natural resources have engrossed the attention of the scientific community and International and regional organizations to assess and monitor such changes on a regular basis (Cihlar 2000, Achard et al. 2001, DeFries et al. 2002). The study of Land Use and Land Cover Change (LUCC) is a joint core project of the International Geosphere-Biosphere Program (IGBP) and the International Human Dimensions of Global Environmental Change Programme (IHDP) (IGBP 1993, Ojima et al. 1994, Turner et al. 1994, IGBP/IHDP 1995). Many nations and International organizations have dedicated considerable attention to LUCC (Fischer and Ermoliev 1996, IIASA 1998, Kuninori 1999), including International organizations such as the United Nations Environment Programme (UNEP), United Nations Food and Agriculture Organization (UN/FAO), International Institute of Applied Systems Analysis (IIASA) and International Geographical Union (IGU).

Increasing human activity across the globe has rapidly accelerated the pace of land cover/use change, resulting in extensive changes in the spatial arrangement of native habitats and biodiversity loss (Collinge 1996). Habitat modification (including habitat

loss, degradation and fragmentation) is now considered as the major threats of ecosystem degradation by human activities (Whitfield et al. 2002). Recent changes in agricultural management and strengthening have altered the equilibria between biota and patterns in both farmed and natural landscapes (Poudevigne et al. 2002). More land was turned into cropland in the 30 years after 1950 than in the 150 years between 1700 and 1850 (Millenium Ecosystem Assessment 2005). Most of the world's biomes have experienced a 20%-50% conversion to human use (Collinge 2009). Worldwide, about 50% of tropical dry forests were transformed to other uses by 1950, and another 10% was lost between 1950 and 1990. Nearly 70% of the natural cover of temperate grasslands disappeared by 1950, and an additional 15% has vanished since then (Collinge 2009). As a consequence, efforts to protect biological diversity have shifted from protection of distinct threatened species to unified strategies that protect ecosystems or habitats (McNeely et al. 1990, Lauer and Whistler 1993). The signing of the biodiversity Convention at the Rio "Earth Summit" Conference in 1992, by a number of countries, establishes the global political assurance of retaining biodiversity (Ledoux et al. 2000).

Changes in landscape patterns are reliant on the spatial and temporal scales of observation (Meentemeyer and Box 1987, Turner 1989, Malingreau and Belward 1992) which happens over long periods of time as an outcome of changes in population (Turner et al. 1990, 1993). These changes are driven by seasonal and inter-annual climatic disparities, long term climatic shifts, vegetation succession and human or natural instabilities (Hobbs 1990). The interaction between these changes leads to explicit landscape dynamics. The monitoring of landscape spatial pattern at a variety of temporal scales has several applications (Turner 1989) and as land transformation patterns vary in their spatial patterns, they may also change significantly in their impact on vital ecological processes (Franklin and Forman 1987, Kareiva and Wennergren 1995, Fahrig 1997, 2003, Harrison and Bruna 1999, McGarigal and Cushman 2002). Environments within regenerating stands change quickly and therefore, stand condition information must be updated occasionally. A thorough understanding of the patterns, the reasons and both the social and ecological consequences of historical changes improve our ability to predict future landscape dynamics and design more effective landscape management approaches (Kienast 1993).

4.2 Literature Review

Remote sensing practices by means of satellite imagery are effective methods to document the rates and patterns of alteration in forest ecosystems and can deliver useful information to help resolve disagreements about future management directions (Green et al. 1994, Cohen et al. 1995). This technique governs and measure the differences in a variety of land cover features over time (Jha and Unni 1994), providing a cost-effective method to characterize varying land-cover conditions and to monitor changes at multiple scales (Lunetta et al. 2002) especially over large areas (Langley et al. 2001, Nordberg and Evertson 2003). Remote Sensing is progressively used to map land cover (Achard et al. 2002, Mayaux et al. 2005), and wildlife habitat allowing a rapid valuation across areas on a steady basis that may be too extensive or too challenging to survey on the ground. At regional to global scales, the only practical means to map and monitor land cover is through satellite remote sensing (Tucker et al. 1985, Townshend et al. 1991, Skole 1994, DeFries et al. 1998).

Because of the potential capability for systematic observations at various scales, remote sensing technology extends possible data archives from present time to over several decades back. The progression of Geographical Information system (GIS) technology further availed the planners of a powerful tool which, by incorporating spatial data collected from different sources, different formats, permits overlaying of two or more maps to be carried out with ease (Wheeler and Ridd 1985, Williams 1985, Berry and Berry 1988, Hathout 1988, Lindhult et al. 1988). Sensible and accurate change detection of Earth's surface features provided the foundation for better understanding of relationships and interactions between human and natural phenomena to better manage and use resources.

Consecutive aerial photographs have been regularly employed by planners to detect land use alteration over a period of time in a region (Avery 1965, Faulkner 1968, Richter 1969, Adeniyi 1980, Lo and Wu 1984). Based on the possibility of analysis, there are two types of study of land use/cover change: Global or Continental Scales; and Regional Scales. In the last few years, landscape researchers tried to understand temporal and spatial patterns of landscape changes so as to develop complete models

of land cover dynamics (Iverson 1988, Turner and Ruscher 1988, Baudry 1993, Simpson et al. 1994, Medley et al. 1995). Because of the advantages of repetitive data acquisition, its synoptic view, and digital format appropriate for computer processing, remotely sensed data, such as Thematic Mapper (TM), Satellite Probatoired' Observation de la Terre (SPOT), radar and Advanced Very High Resolution Radiometer (AVHRR), have become the major data sources for different change detection application during the past decades.

Satellite imagery has been extensively used to define landscape structure and change in tropical forests (Myers 1980, Jenkins 1987, Nelson et al. 1987, Sader and Joyce 1988, Fearnside et al. 1990, Green and Sussman 1990, Bjorndalen 1992, Skole and Tucker 1993, Tucker et al. 1994). Many efforts have been made at the global or national scale using NOAA/ AVHRR data, rather than more comprehensive regional land use studies using Thematic Mapper (TM) and SPOT data, are less common (Skole et al. 1997, Shi et al. 2000). Most studies are concerned with applications in deforestation, urban sprawl and land use change in common. Land use change within agricultural areas is also occasionally addressed. Primary work on land cover change analysis using remote sensing (Nelson and Holben 1986, Tucker et al. 1991, Lambin and Strahler 1994a, b) was based on the analysis of multi-temporal Normalized Difference Vegetation Index (NDVI) series produced by the National Oceanic and Atmosphere Administration (NOAA) from Advanced Very High Resolution Radiometer (AVHRR) images. Today there are numerous techniques to perform change detection with satellite data (Jensen 1996, Gong and Xu 2003).

Fewer studies have been commenced in temperate zone (Hall et al. 1991, Morrison et al. 1991, Ripple et al. 1991, Fiorella and Ripple 1993, Luque et al. 1994, Spies et al. 1994, Turner et al. 1996, Zheng et al. 1997, Sachs et al. 1998) with few more remote sensing studies on forests in Northeast Asia (Tian et al. 1995, Shao et al. 1996, Zheng et al. 1997). In other tropical and temperate regions, change detection techniques in a time series of imagery have been used to monitor changes in land use, shifting cultivation, vegetation phenology, and pasture development and to assess deforestation, crop stress and damage (Singh 1989, Collins and Woodcock 1996, Cohen et al. 1998). Digital change detection allowed quantification of temporal phenomena in multirate satellite imagery (Tian et al. 1995, Coppin and Bauer 1996)

used available vegetation maps and aerial photography to describe the patterns and rate of change of a boreal forest landscape in the inner Mongolian Autonomous Region in northeastern China.

A huge number of change detection techniques have been established to acquire operational monitoring since the advent of the orbital system (Lillestrand 1972). Change detection is the practice of identifying changes in the state of an object or phenomenon by observing it at different times (Singh 1989). The urgent need to develop accurate assessments of the rate and scale of land-cover changes has led to the plethora of change detection and classification techniques. These techniques have been formulated, applied and evaluated, and several reviews of change detection techniques have been conducted (Singh 1989). Change Detection has occasionally been accomplished by the comparison of the two independent classification results (Joyce et al. 1980, Burns 1985, Tucker et al. 1985). Overall, change detection encompasses the application of multi temporal datasets to quantitatively examine the temporal effects of the phenomenon. The technique involved manually interpreting land use categories from aerial photographs for each period of time and measuring their areas. By comparing the area between two or more periods of time, the predominant land use changes in the region can be detected.

Several algorithms have been developed for change detection, e.g. (1) Change detection using write function memory insertion (Price et al. 1992, Jensen et al. 1993); (2) Multidate composite image change detection (Fung and Ledrew 1987, Eastman and Fulk 1993); (3) Image algebra change detection (Green et al. 1994) using univariate image differencing (Weismiller et al. 1977, Williams and Stauffer 1978); (4) Image regression (Jensen 1983); (5) Image rationing (Howarth and Wickware 1981); (6) Transformed band differencing such as Vegetation index differencing (Nelson 1983); and (7) Multispectral change vector magnitude and direction (Malila 1980). Change vector-based procedures have also been employed recently to data having high temporal dimensionality (Malila 1980, Lambin and Strahler 1994a,b); (8) Hybrid approaches based on a mix of categorical and radiometric change information have also been proposed and evaluated (Colwell and Weber 1981, Johnson et al. 1998); (9) Manual on screen digitization of change (Lacy 1992, Wang et al. 1992, Light 1993); (10) Post-classification comparison change detection (Rutchev and

Velcheck 1994); (11) Knowledge based vision systems for detecting change (Wang 1993, Gong et al. 1996); and so on.

Other approaches used for change detection include: (1) Thresholding of a single spectral feature, for example, difference channel (2) Principal component transformation or other linear transformations to merge two temporal datasets (Byrne et al. 1980, Richards 1980, Collins and Woodcock 1996); (3) Analogue images of bands describing the change (Eyton 1983, Häme 1991); (4) Direct classification of a multirate dataset (Swain 1978, Häme 1991, Varjo 1994); (5) Decision Tree Classifiers (Lozano-Garcia and Hoffer 1985); (6) Residual computation of a regression model between two images (Frank 1984, Olsson 1994, Jha and Unni 1994) etc.

Before applying any of these change detection techniques, it is a thumb rule to fulfill the following conditions: (1) Accurate registration of multi temporal images; (2) Precise radiometric and atmospheric calibration or normalization between multi-temporal images (3) Similar phenological states between multi-temporal images; and (4) Selection of same spatial and spectral resolution images if possible. The selection of a suitable change detection algorithm is very important (Jensen 1996). Post classification comparison change detection is the most frequently used quantitative method of change detection, though every error in the individual date classification map is presented in the final change detection map (Rutchev and Velcheck 1994). Therefore, it is imperative that the individual classification maps used in the post-classification change detection method be as accurate as possible (Augenstein et al. 1991).

In spite of several evaluations of these techniques (Weismiller et al. 1977, Singh 1989, Stow et. al 1990) no standard techniques have yet been adopted (Macleod and Congalton 1998). Any change detection technique has its own set of advantages and disadvantages, and no single approach can be considered optimal or applicable to all cases. The major disadvantage of most existing methods is that they require high quality training data, which are often difficult to obtain in change detection. Among the many factors governing selection of a change detection strategy are information requirements, spectral coverage, data availability and quality, image processing

resources, analyst skill and experience, phenomenological knowledge, time and cost constraints, and the importance of labeling that are detected. Amongst all the techniques, change vector analysis (CVA) considered as the most valuable technique for the change detection studies. The other techniques used in this study are markov analysis and cellular automata (CA) markov analysis. Details of the above techniques and their advantages, disadvantages are stated below:

4.2.1 Change Vector Analysis (CVA)

Change vector analysis (CVA) is a radiometric technique, the main utility of which is the finding of all changes present in the input multispectral data (Malila 1980). The change vector analysis by Malila (1980) is markable example of approaches to both distinguish and identify changes automatically. The method uses spectral and spatial information to accomplish a spectral-spatial clustering on a two temporal image dataset in order to find homogeneous areas. First, homogeneous areas on each image are located, and then the change vector is calculated between the area spectral mean in the earlier and later images. The magnitude of the vector gives the magnitude of the change while the angle gives the type of change. Thus not only the change type, but also its intensity is obtained. The two temporal images used can be original channels or derived spectral bands such as “greenness” and “brightness” (Kauth and Thomas 1976).

The possible advantages of CVA over some other methods include : (1) ability to simultaneously process and investigate change in all multispectral input data layers (as opposed to selected bands); (2) avoidance of compounding of spatial-spectral errors often inherent in multi-date classifications; (3) the capability to detect changes both in land cover and condition; and (4) computation and separation of multidimensional change vector components, and composition of change images that retain this information and facilitate change interpretation and labeling. CVA, unsupervised classification and visual interpretation of aerial photographs were used to detect land-cover change and found that the combination of CVA and unsupervised classification provided more powerful interpretation of change than either method alone (Silapaswan et al. 2001).

The most critical pre-processing requirements for CVA are the accurate geometric registration and radiometric normalization of the input data. While information requirements, study objectives or research hypotheses will dictate the spatial-spectral-temporal coverage to be employed. Image to image registration quality is serious in change detection. Mis-registration leads to change detection commission errors that corrupt results, and are extremely difficult or impossible to mitigate via post processing intervention. Radiometric normalization of the input data is equally critical in radiometric change detection procedures such as CVA, both to minimize detection of spurious changes due to poor normalization (i.e. reduce commission error) and ensure that changes of small magnitude can be detected and measured (i.e., minimize omission error). Particularly for long term studies, selection of anniversary acquisition dates (i.e. the same dates on separate years) is advisable for change detection processing to reduce the effects of variability.

4.2.2 Markov Analysis

Markov examines two qualitative land cover images from different dates and produces a transition matrix, a transition areas matrix and a set of conditional probability images. A Markovian method is one in which the state of a system at time 2 can be predicted by the state of the system at time 1 given a matrix of transition probabilities from each cover class to every other cover class. A transition probability matrix expresses the probability that a pixel of a given class will change to any other class (or stay the same) in the next time period. It also produces a transition area matrix which expresses the total area (in cells) probable to change in the next time period and a set of conditional probability images – one for each land cover class. These maps express the likelihood that each pixel will belong to the selected class in the next time period. These are called conditional probability maps as this probability is conditional on their current state (Eastman 2000).

4.2.3 Cellular Automata Markov (CA-MARKOV) Analysis

4.2.3.1 Cellular Automata (CA)-Markov Model Description

Cellular Automata (CA)-Markov Chain analysis modeling technique, set in IDRISI Andes software (Eastman 2000), includes two techniques: Markov Chain analysis and CA. The Markov Chain analysis clarifies the probability of land cover alteration from one period to another by developing a transition probability matrix between t1 and t2. The Markov module is based on the first law of Geography by using a contiguity rule (Cabral and Zamyatin 2006). According to the rule a pixel that is near one specific land cover category is more liable to become that category than a pixel that is farther. CA is integrated into the Markovian approach to add the spatial character to the model. CA-Markov models the change of several classes of cells by using a Markov transition matrix; a suitability map and a neighborhood filter.

4.2.3.2 Implementing and validating the model

The analytical power of the model is assessed by linking the result of the simulation t2 to a real classified map of t2 by means of using three different Kappa statistics (Ponitus 2000, Dushku et al. 2003): Kappa for no information (K_{no}), Kappa for location ($K_{location}$), and Kappa for quantity ($K_{quantity}$). K_{no} defines the overall accuracy of the simulated map. $K_{location}$ confirms the model's ability to predict location. $K_{quantity}$ is a measure of validation of the simulations to predict quantity perfectly. If the predictive power of a model is considered strong then it is used to make future projections assuming that the transition mechanism verified between t1 and t2 is going to be repeated.

4.3 Background information of Forest Dynamics and Management Systems of Gir (1878 – 2009)

Gir forests comprise of forest areas of both the ex-Junagadh and ex-Baroda states. The past history for those two tracts is therefore somewhat different (Joshi and Karamchandani 1976).

Junagadh Gir Forests

1878-1914: The forests were managed by the revenue department. A permit holder on payment of prescribed fees was allowed to cut the material in any manner he liked from a given area. As a result, all the best trees were removed leaving behind stunted, hollow trees.

1915-1934: As per the plan by Ratnagar in 1920, the area was divided into two working circles. Western working circle retained mixed deciduous forests with teak forming 80% of the crop. The coppice with standard system, with a rotation of 30 years and a closure of 10 years for the exploited coupes was recommended with no cleanings and thinning. Eastern working circle retained non-teak areas bearing mainly thorny species and large grassy blanks. A sanctuary for lions was formed into Devalia block and Ronia vidi in Malpara block was worked for hay collection.

1935-1954: The simple coppice system i.e. clear felling with reservation of fruit trees e.g. rayan, ambo, amli, arithi was introduced. The closure period was reduced to 3 years and *rab* method was introduced for regeneration. This system was followed till 1952, and the number of fruit tree species was increased to cover ambo, amli, rayan, ravano, aritha, simlo, vad, papal, bahedo, jambu, pipalo, khair.

1965-1972: Acharya revised the working plan of Gir forests for the period from 1954-55 onwards and his working plan was continued. Five different working circles were created as under.

Superior Teak Working Circle: Well stocked good quality teak areas were included in this working circle. Rotation of 40 years was fixed. Modified clear felling with certain reservations was prescribed. This was supplemented with 5 percent artificial regeneration, cleaning in 5th year and thinning in 15th and 25th years with a closure for 6 years.

Inferior Teak Working Circle: Open and poor teak forests were involved in this circle, with a rotation of 30 years. Modified clear felling with reservation was

prescribed. Most of the prescriptions of superior working circle were made applicable except that only one thinning in 10th year was prescribed.

Improvement Working Circle: All the open scrub type non-teak forests (except *vidis*) were included in this circle. Light and conservative “Improvement felling system” with felling cycle of 20 years, soil conservation measures with 5 percent plantation and closure of 10 years was prescribed.

Grass (Fodder) Working Cycle: Tree growth was worked on a felling cycle of 10 years. Artificial regeneration of fruit, fodder and shade trees in percent of coupe area was prescribed.

Sarasia Gir Forests

During the period from 1884-1894, when the management of forests was with the Revenue department, misuse was permitted through contractors on permit system. As a result, serious damage to forests intensified. The protection was poor and fire conservancy was not known. During the period of ex-Baroda state, Sarasia Gir was treated as grazing ground and the main item of revenue was grazing fees and sale of grass. As a result of this, damage to fodder trees happened because of extensive lopping. From the year 1954-55 these forests were brought under Acharya’s working plan.

Joshi Plan (1976 to 1985): The Acharya plan was revised in 1969 to change traditional method of working of forests to wildlife management and the revision work was completed in 1976. The entire working plan of the Gir forests was based on the concept of biodiversity conservation and improvement of habitat in favour of wildlife. Most of the Sanctuary and National Park area was managed under National Park Habitat Improvement Working Circle and the Watershed Conservation Working Circle. The rest of the area excluding forest settlement, protected forests and reservoirs were included in Grasslands and Coastal Plantation Working Circle.

The total area under the Joshi plan was 1602.6 sq km. In order to fulfill the objects of management, an area of 258.71 sq km. has been constituted as “National Park”. A

buffer zone of a width of about 1 km was left undisturbed all around the area of the National Park. Total area of this buffer zone was 100.86 sq km. The remaining area was constituted into the following working Circles.

Habitat Improvement Working Circle: The area allotted to this working circle was 292.41sq km. It included all the teak bearing areas with teak forming more than 10% of the growing stock. The entire crop was of coppice origin with a large proportion of old and mature trees. The forest was worked mainly with the idea of preservation and improvement of habitat.

Watershed Conservation Working Circle: The area allotted to this working Circle was 704.10 sq km. This working circle included all areas containing sparse growth of non teak species and formed catchment areas of many rivers flowing throughout this region. These forests were worked mainly with the idea of soil and moisture conservation and practically no exploitation was prescribed.

Grass-land Working Circle: The working circle comprised of 164.58 sq km of grass producing areas locally known as *vidis*. Improvement measures for reserve *vidis* were prescribed for 10 years.

Plantation Working Circle: The area allotted to the working circle was 20.086 sq km. It included all the areas wherein plantations had been carried out in past along the coastal border in Gir East and Gir West Divisions and all other such areas (in charge of Forest Department) proposed to be covered by such plantations in future.

Recreation (Overlapping) Working Circle: The entire area of the Gir Sanctuary i.e. 1412 sq km was covered by this working circle. The special object of management was to create conditions for the development of Wildlife tourism.

4.4 Objectives

To date, there have been very few studies of landscape pattern or change for the Gir PA. Qureshi et al. 2004 used LISS 3 imagery to classify forests. The current study

represents the first landscape change analysis using remotely sensed data in the Gir wildlife sanctuary. The objectives of this research are:

1. To assess the rate of change in land use and vegetation patterns over a 10-year period and to consider the implications of the rates and patterns of change on future ecological conditions.

Main questions of interest were the following:

- (a) What is the pattern of change in the landscape?
- (b) What habitat types are vulnerable to change and why?

Hypothesis for the objective

H₀: Land use cover changes over years are similar across different habitat types or random.

H_a: Human driven land uses or land cover increases with time.

4.5 Methodology

The post-classification comparison approach was employed for detection of land use/cover changes, by comparing independently produced classified land use/cover maps. The main advantage of this method was its capability to provide descriptive information on the nature of changes that occurs. The following approaches were considered appropriate to fulfill above objectives. Field data was collected in a systematic manner, described in chapter three (section 3.6). Landsat TM satellite imageries were obtained of different years to understand the temporal changes occurred in the place. The spatial distributions of each of the classes were extracted from each of the land use/cover maps by means of GIS functions. Change Vector Analysis (Malila 1980) was used to check the direction and magnitude of the change. Markov analysis (Eastman 2000) was also applied to understand the probability of changes of one land cover type to other. For this study, we used Cellular Automata (CA)-Markov Chain analysis modeling technique, embedded in IDRISI Andes software (Eastman 2000) for predicting future landscape pattern.

The Markov Chain analysis model was executed using the Markov module. The first step in the model was to develop a transition probability matrix for each of the land

cover classes between the years 1998 and 2002, and this in turn was used as an input for modeling land cover change for 2009. Based on the probability matrix obtained from Markov analysis, simulated land cover map of 2009 was created with the cellular automata markov analysis tool (CA-MARKOV). The validation of the result obtained from this technique was done using the actual classified image of 2009 with the simulated image of 2009. CA analysis was carried out with the CA-Markov module, which uses the output from the above stated Markov Chain analysis and transition suitability image collection, and applies a contiguity filter. Later images of 1998 and 2009 was analyzed using Markov Change Analysis and based on the probability matrix obtained, simulated land cover map of 2020 was created with the cellular automata markov analysis tool (CA-MARKOV). First the probability matrix was obtained from Markov analysis and then assuming the transition probability matrix remain unchanged, simulated land cover map of 2020 was produced. FRAGSTAT (McGarigal and Marks 1995) was run on the classified image of 1998, 2002 and 2009. The trend of change observed in values of different indices with the change of land cover type is discussed.

The techniques applied to classify individual scene is discussed in chapter 3 (Section 3.6).

4.5.1 Strategies used for Land use/cover change detection

4.5.1.1 Satellite Images and Reference Data

Three satellite images from the Landsat program (path-row 149/45) were used in this study. All these images were acquired by Landsat 5 Thematic Mapper (TM) sensor on 17th November, 1998, 4th November, 2002 and 1st December, 2009. These images were cloud free over the study area. Observations at anniversary dates were selected because by using non-anniversary dates into consideration, the number of possible combinations of observation dates increases dramatically and the range of change magnitudes for different combination of dates is much wider than for anniversary dates. Thus the comparison of anniversary dates does indeed provide some control over seasonal variations.

During the post monsoon season in 2010 (November – January, 2011), I gathered ground reference data from more than 900 plots. I recorded vegetation structure and

composition, slope and aspect for all of these reference sites. The location of these sites recorded with a GPS (Garmin 72) (Chapter 3, Section 3.6).

4.5.1.2 Classification of the 1998, 2002 and 2009 TM scene

All bands of the TM image except the thermal band were used in PCA to extract information and transforming a set of correlated spectral bands of image data into an equivalent set of uncorrelated components of the data. The stacked PC layers were initially classified using the Isodata algorithm (Unsupervised classification) into 100 classes and then regrouped into the ten major classes under investigation based on the field study (see chapter three for details). The 10 classification categories consisted of:

1. Moist Teak Forests.
2. *Wrightia*- Teak Forests
3. Teak-*Wrightia* Forests
4. *Boswellia*-*Accacia* Forests
5. *Anogeissus*-*Boswellia*-*Accacia* Forests
6. Bare land representing soil that has almost no vegetation.
7. Water, where this class mainly represents water stored behind dams.
8. Agriculture land including crop and fallow land as well as rainfed and irrigated land.
9. Grassland.
10. Fallow land

4.5.1.3 Classification accuracy analysis

Accuracy assessment was performed for 1998, 2002 and 2009 land cover maps using topographical maps, local knowledge, and systematic sampling design. Initially classified images were evaluated visually, and the effects of topographic normalization were noted. The image was evaluated quantitatively with ground truth data points. Several other types of ground truth maps, such as drainage network, road network, and settlement areas were included in the ground truth map. For the classified image of 2009, accuracy was determined based on a stratified random sample of 200 (out of total 900 points) points distributed throughout the area. Kappa statistics (Landis and Koch, 1977), which assess overall accuracy by incorporating

individual errors of omission and commission, were also calculated (Chapter 3, section 3.7, table 3.16)

Identification of respective land use/cover classes and the accuracy of the classified images of 1998 and 2002 were assessed by identifying bigger patches (more than 1 ha) and using ground truth points of 2009, distributed in those bigger patches. A detailed literature review of past vegetation studies and management plans assured that in last ten years, main forest types in the area did not change so much but area under one forest type might have changed to other forest type, following process of succession or human disturbance. Use of satellite images of the same season every year assured the same reflectance values of respective forest types.

Landscape pattern analysis was performed on the raster coverages with FRAGSTATS v2.0. (McGarigal and Marks 1995). Numerous indices of landscape pattern were calculated. FRAGSTATS calculated a number of spatial metrics for each patch, for each cover class and for the entire landscape. We analyzed all metrics for the land cover classes of all dates (1998, 2002 and 2009) but selective results showing significant changes over time are presented here.

The following indices are presented here:

- (1) The total number of landscape patches was used as an overall measure of landscape fragmentation. This index exhibits a unimodal relationship with land cover proportions because maximum patch number occurs at intermediate levels and decreases at both high and low cover proportions (Gustafson and Parker 1992).
- (2) Mean patch size (Area mean) is a similar measure of fragmentation but one that exhibits a pattern opposite to that of patch number because patch size is usually largest in homogeneous landscapes.
- (3) Total landscape edge was calculated by summing the length of all patch boundaries in the landscape. Both the number of patches in a landscape and the complexity of patch boundaries influence the amount of edge in a landscape.

(4) Patch density is a limited, but fundamental, aspect of landscape pattern. Patch density has the same basic utility as number of patches as an index, except that it expresses number of patches on a per unit area basis that facilitates comparisons among landscapes of varying size. Of course, if total landscape area is held constant, then patch density and number of patches convey the same information. Like number of patches, patch density often has limited interpretive value by itself because it conveys no information about the sizes and spatial distribution of patches.

4.6 Results

The classified land use/cover maps for the study area in 1998, 2002 and 2009 are shown in Figure 4.1, 4.2 and 4.3. The trends in all the land use/cover changes are summarized in table 4.1. Viewed as a time series, the quality and quantity of different forest types have substantially varied during the 11-year period from 1998 to 2009.

Table 4.1: Land use/cover changes for Gir PA, as extracted from Landsat images, 1998-2009

Land use/cover types	1998		2002		2009	
	Area in Hectare	Percentage (%)	Area in Hectare	Percentage (%)	Area in Hectare	Percentage (%)
Water	2443.59	1.69	728.28	0.50	1168.47	0.81
<i>Wrightia</i> -Teak Forests	60255.09	41.67	35870.67	24.81	33278.67	23.02
Teak- <i>Wrightia</i> Forests	9589.50	6.63	35061.75	24.25	29277.45	20.25
<i>Boswellia</i> - <i>Acacia</i> Forests	36483.66	25.23	35671.32	24.67	27719.46	19.17
<i>Anogeissus</i> - <i>Boswellia</i> - <i>Acacia</i> Forests	17050.05	11.79	18121.59	12.53	25652.25	17.74
Agri-Barren-Fallow land	1766.25	1.22	3744.09	2.59	8651.43	5.98
Grassland	8838.09	6.11	7686.00	5.32	9794.61	6.77
Moist Teak Forests	8166.51	5.65	7709.04	5.33	9050.40	6.26

Teak-*Wrightia* forests increased in 2002 four times from its initial coverage of area in 1998, and covered approximately the same area in 2009. The *Wrightia*-Teak forest areas covered 60255 ha in 1998 and decreased to 33278 ha in 2009. On the other hand

Teak-*Wrightia* forests increased substantially from 9589 ha in 1998 to 29277 ha in 2009. The *Boswellia-Acacia* forests occupied 36483 ha in 1998 and decreased to 27719 ha in 2009. The agriculture, Barren and fallow land jointly occupied 1766 ha in 1998 that increased to 8651 ha in 2009. Though there was not substantially increase in grassland and moist teak forest area but area of both the land cover types increased in 2009. The trends in land use/cover changes for all the classes are shown in figure 4.4.

Wrightia-Teak forest type has decreased by 26976 ha during the period 1998-2009 while the Teak-*Wrightia* forests increased by the area of 19688 ha. Comparison of transition probabilities of these three main succession stages of teak associated forests is described in figure 4.5. The probability of switching towards a more dense community is indicated from this analysis.

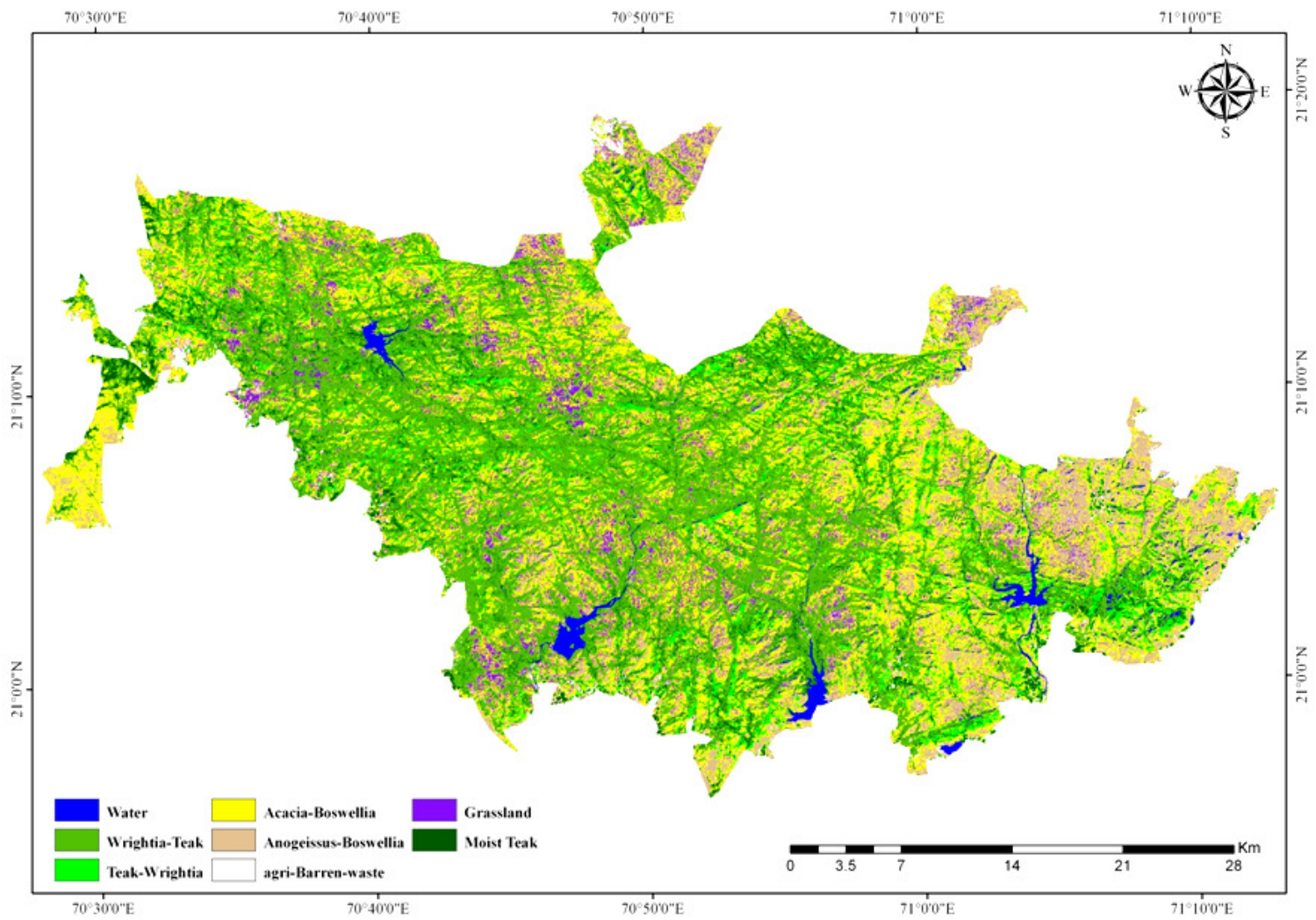


Figure 4.1: Land use/cover map of Gir PA in1998

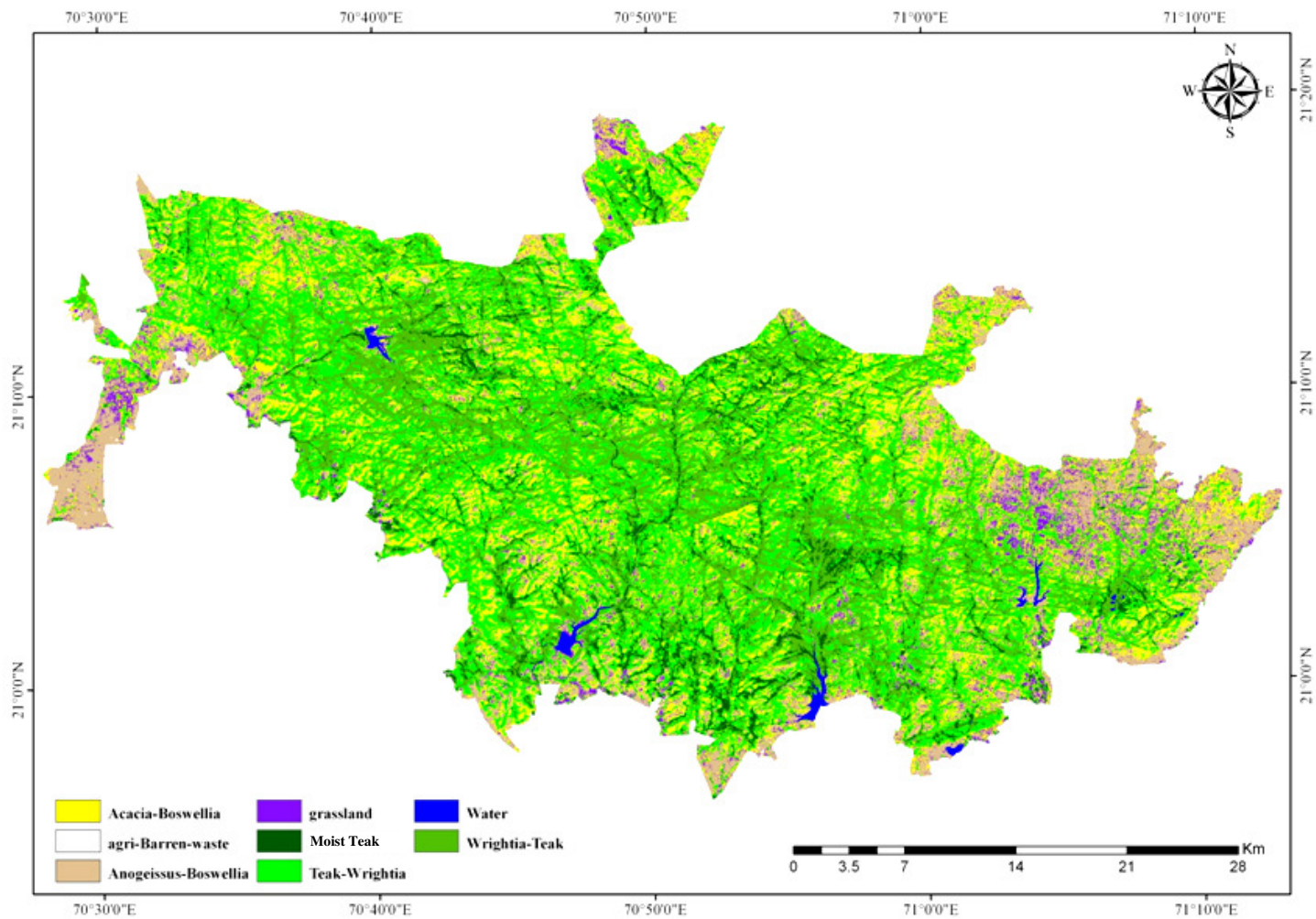


Figure 4.2: Land use/cover map of Gir PA in 2002

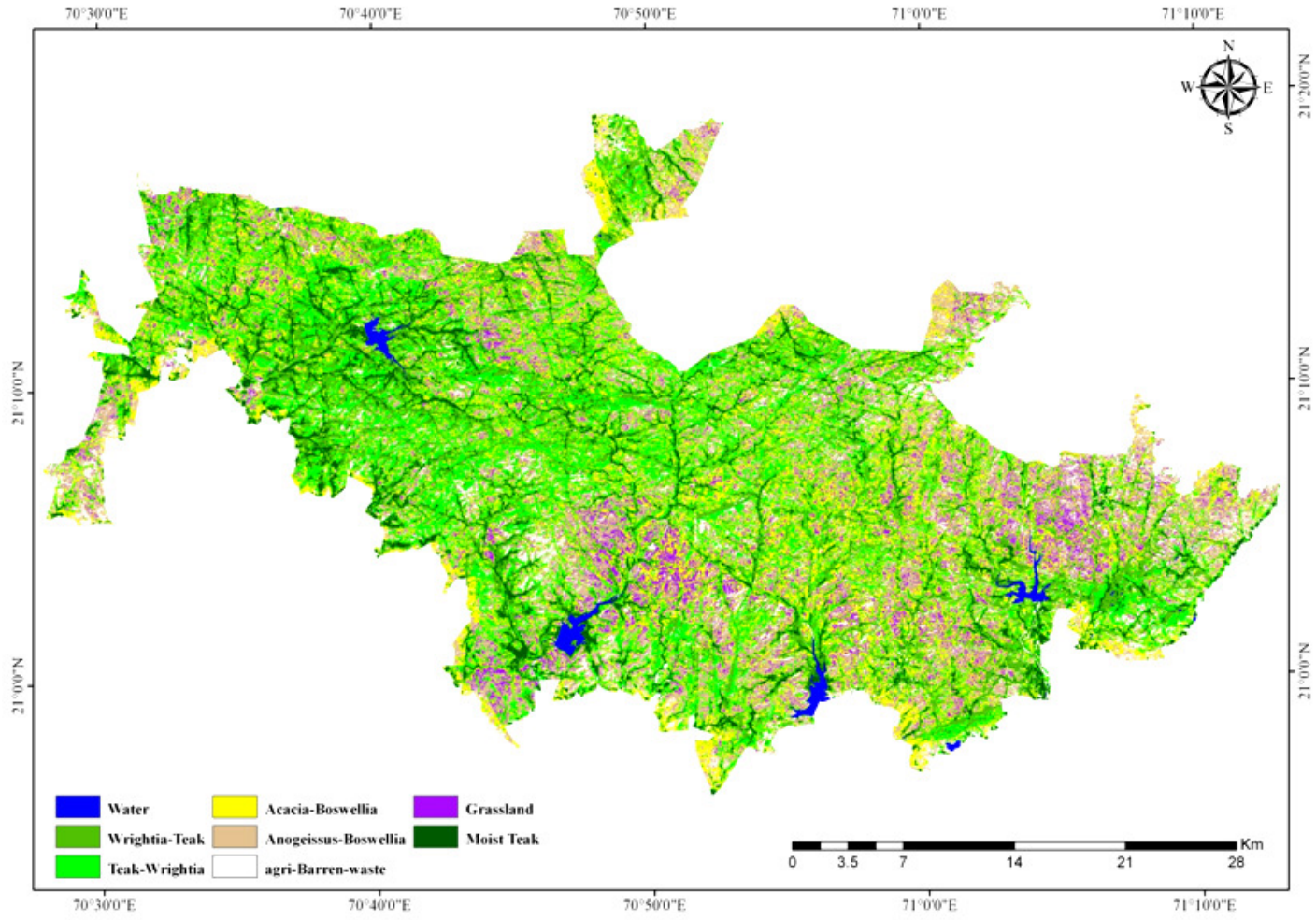


Figure 4.3: Land use/cover map of Gir PA in 2009

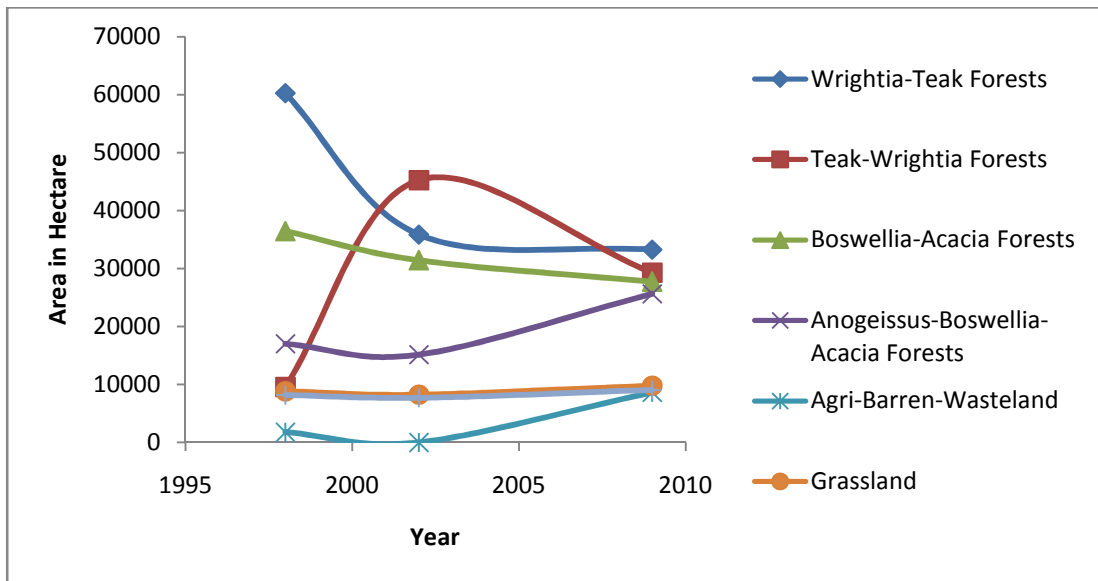


Figure 4.4: Trends in different land use/cover changes from 1998-2009.

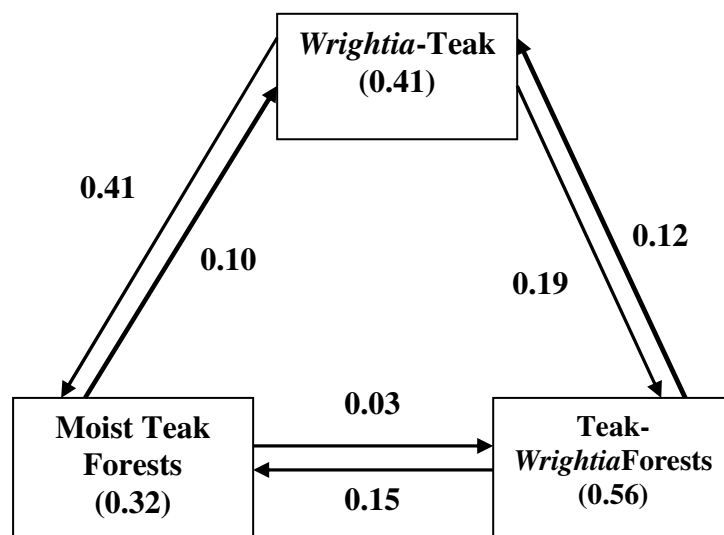


Figure 4.5: Box and arrow diagram illustrating land cover transition probabilities for the Gir Protected Area 1998-2009. Values within boxes are self replacement probabilities, whereas values positioned on arrows are transition probabilities.

The pattern of changes observed in teak associated forest types had effect on spatial distribution of forest too. As the moist teak forests became denser, number of patches of this forest type decreased while number of patches increased for the *Wrightia*-Teak forest type and remained almost constant for the Teak-*Wrightia* forest type (Figure 4.6).

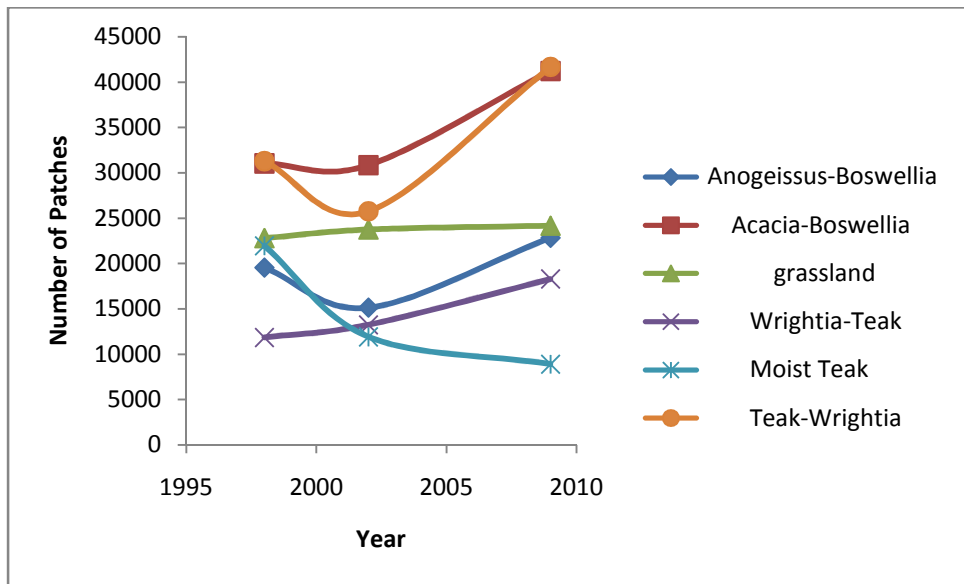


Figure 4.6: Changes in number of patches from 1998-2009

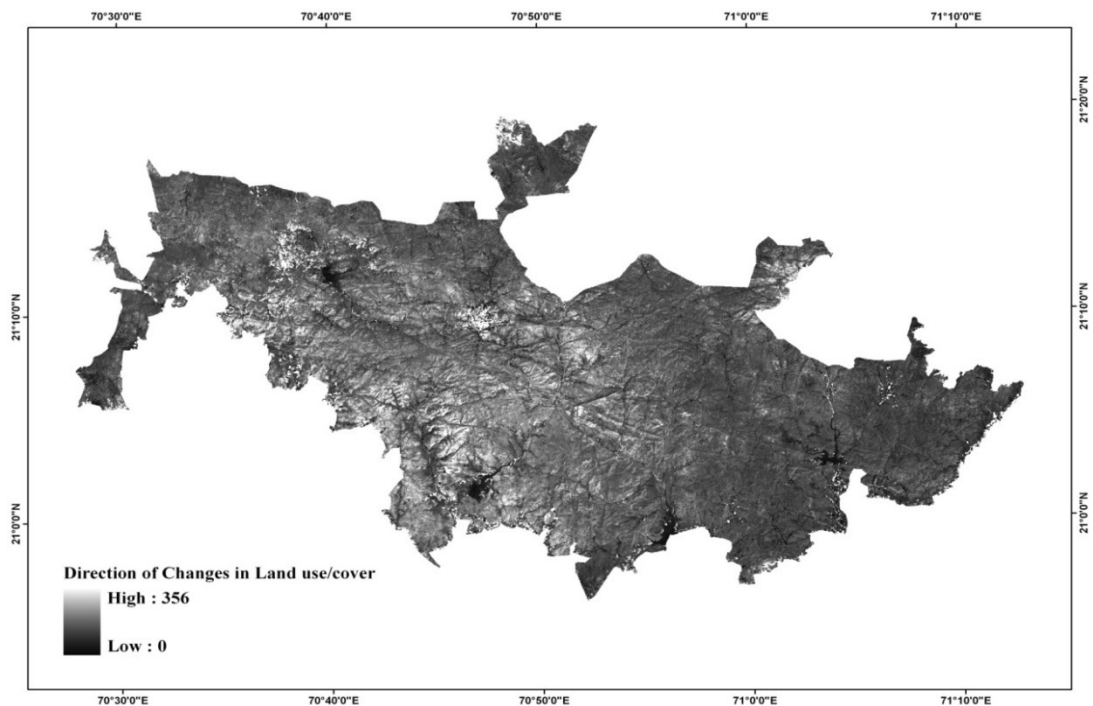


Figure 4.7: Direction of changes in land use/cover obtained from CVA of images of 1998 and 2009

The change has not taken place in all directions but mainly around river channels or water bodies in western part of the Gir. The distribution of these forests from river channels are like moist teak forests to *Wrightia*-teak forests then to teak-*Wrightia* forests. Direction of change image obtained from CVA strongly supports this view with higher values in areas under these forest types around river channels in western Gir mainly (Figure 4.7).

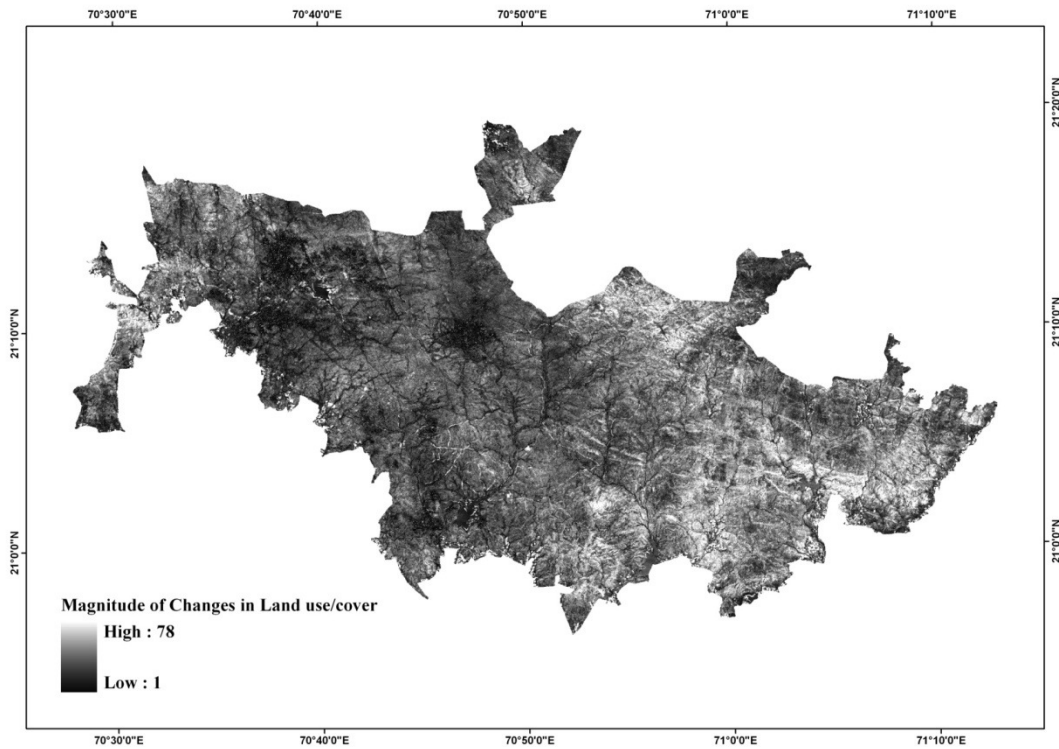


Figure 4.8: Magnitude of changes obtained from CVA of images of 1998 and 2009

Other notable land cover changes include the increase in *Anogeissus-Boswellia-Acacia* forests type from 17050 ha in 1998 to 25652 ha in 2009 with its spatial distribution mainly in eastern portion of the study area where teak associated forest types are less common or absent. The gradual changes in vegetation pattern in eastern Gir, where *Anogeissus-Boswellia-Acacia* forests increased and *Acacia-Boswellia* forests decreased over time are also visible from the magnitude of changes image obtained from CVA (Figure 4.8). Number of patches increased for both the forest types during the study period (Figure 4.6) but the mean patch size decreased for *Acacia-Boswellia* whereas it almost remained same for *Anogeissus-Boswellia* forest type (Figure 4.9). Average patch size increased for the moist teak forest type as small patches begin to coalesce into larger ones and decreased for the *Wrightia-Teak* forest types as the area under this forest type reduced following succession (Figure 4.9).

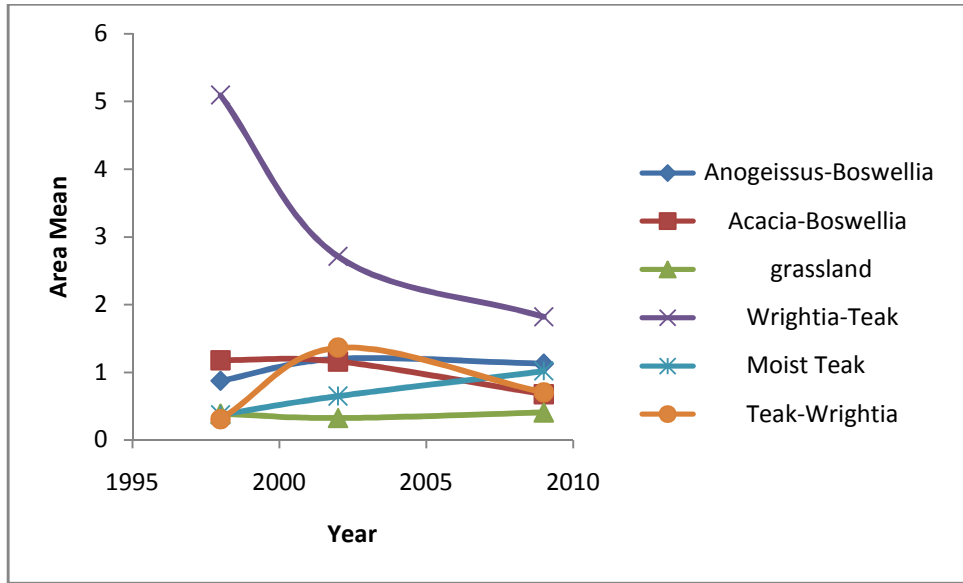


Figure 4.9: Changes in average size of patches from 1998-2009

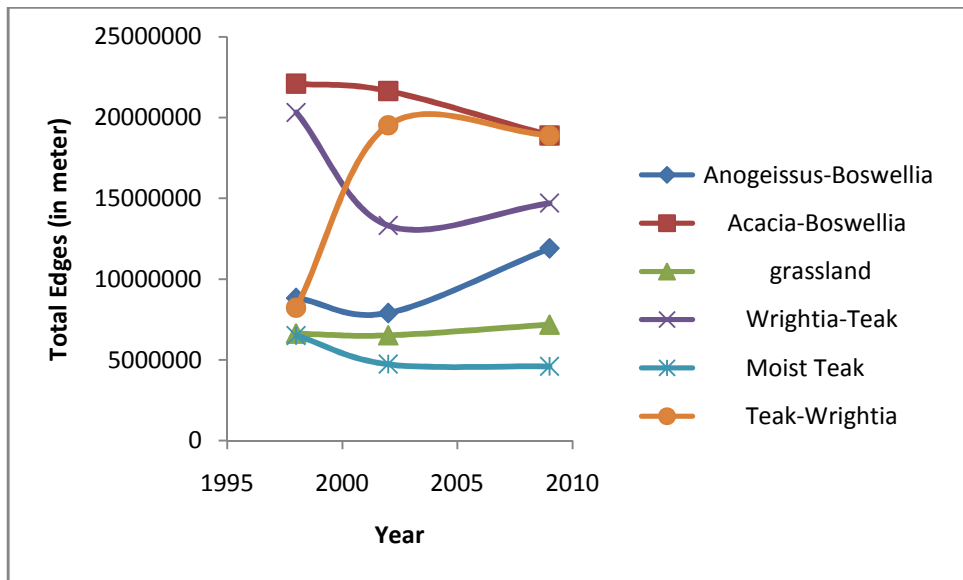


Figure 4.10: Changes in total edges from 1998-2009

The rate of agricultural encroachment and the amount of open area have substantially increased in 2009 increasing the fragmentation and patchiness of the area with an uneven spatial direction. Total edges for *Anogeissus-Boswellia* forest types increased during the study period as a result of spread of both of these forest types over the study period (Figure 4.10) whereas for all other forest types it decreased following the decrease in area under these forest types over time.

Changes in patch density of different forest types occurred as an accumulated effect of increase of area under specific vegetation type as it happened in case of increase of patch density of *Anogeissus-Boswellia* forests, result of succession of *Wrightia*-Teak to Teak-*Wrightia* and moist teak forests where patch density decreased in the *Wrightia*-Teak forest whereas it increased in the later teak associated forest types. Clearance of vegetative patches of *Acacia-Boswellia* showed decrease in patch density too (Figure 4.11).

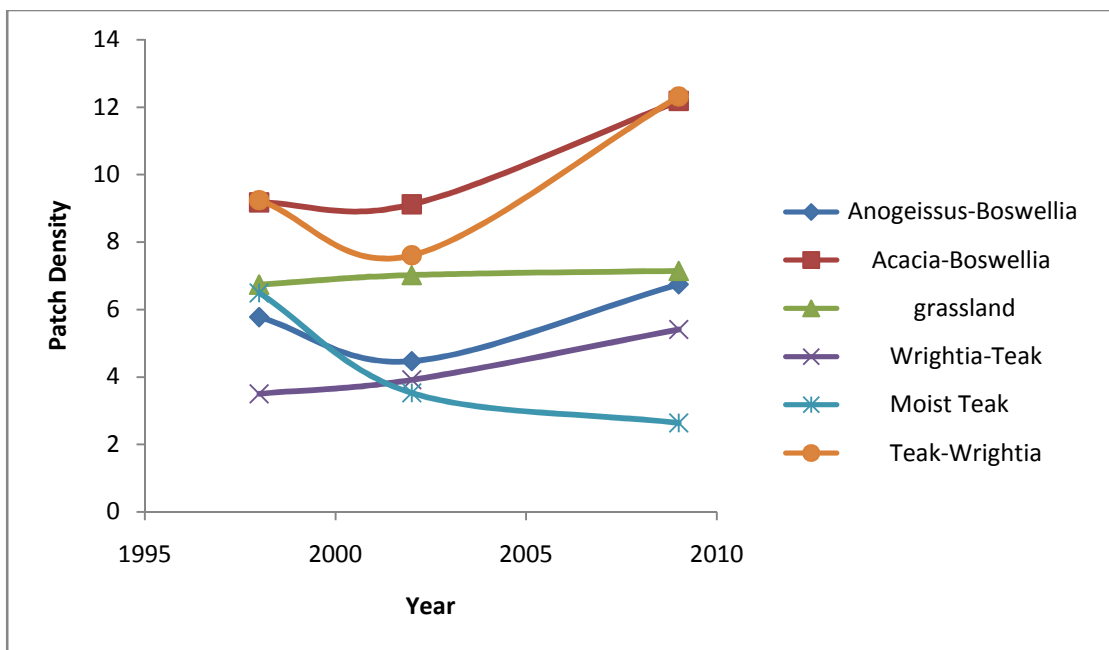


Figure 4.11: Changes in patch density from 1998-2009

Visual interpretation of the land cover modeling results showed that the simulated map (Figure 4.12) for the year 2009 is reasonably similar to the actual classified map of that year. A more detailed analysis was accomplished using the Kappa variations. The closer the values of these indices are to 100%, the stronger the agreement is between two maps. The K_{no} , which also gives the overall accuracy of simulation, is calculated to be 70%.

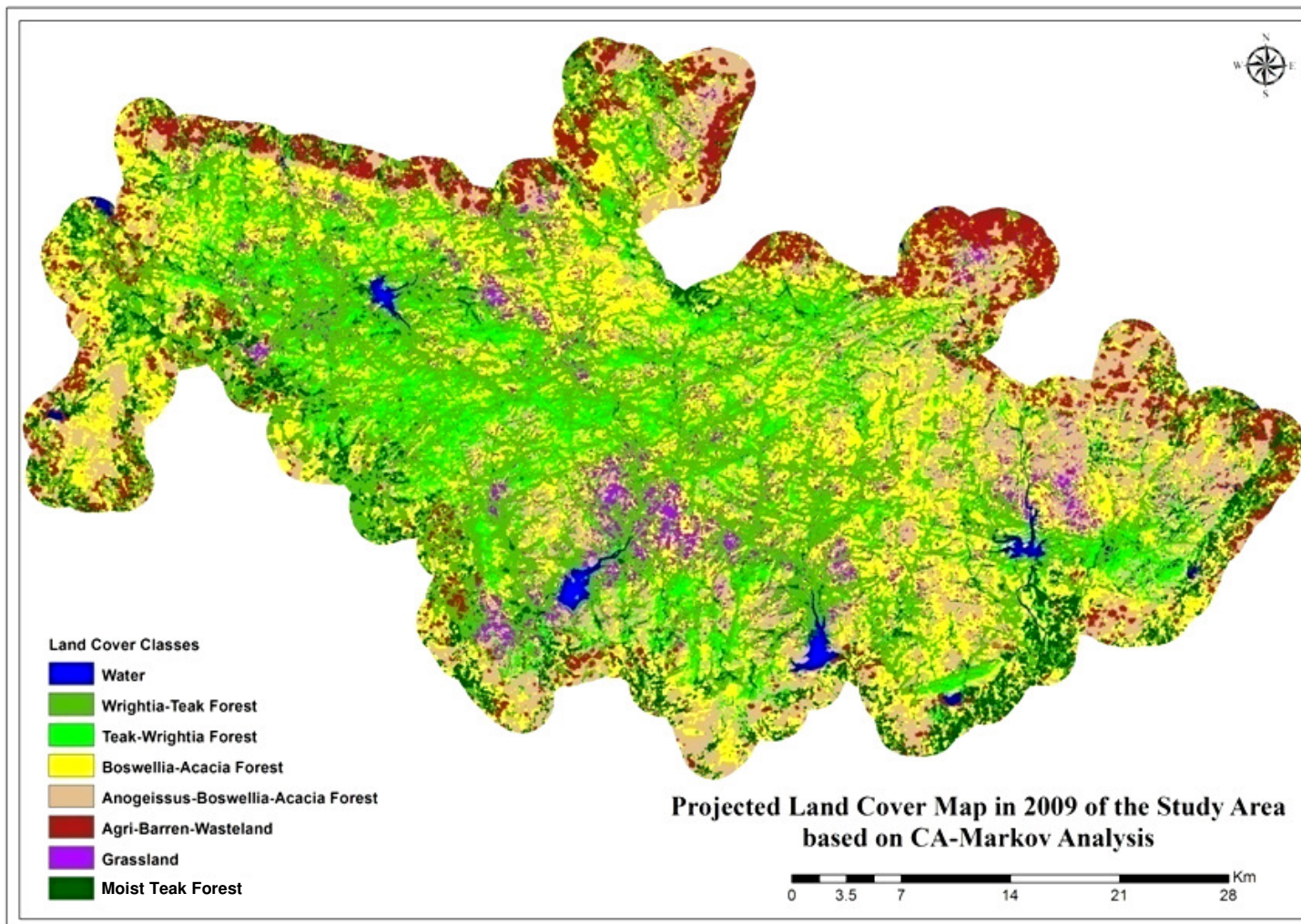


Figure 4.12: Projected land cover map of the study area of the year 2009, based on CA-Markov analysis

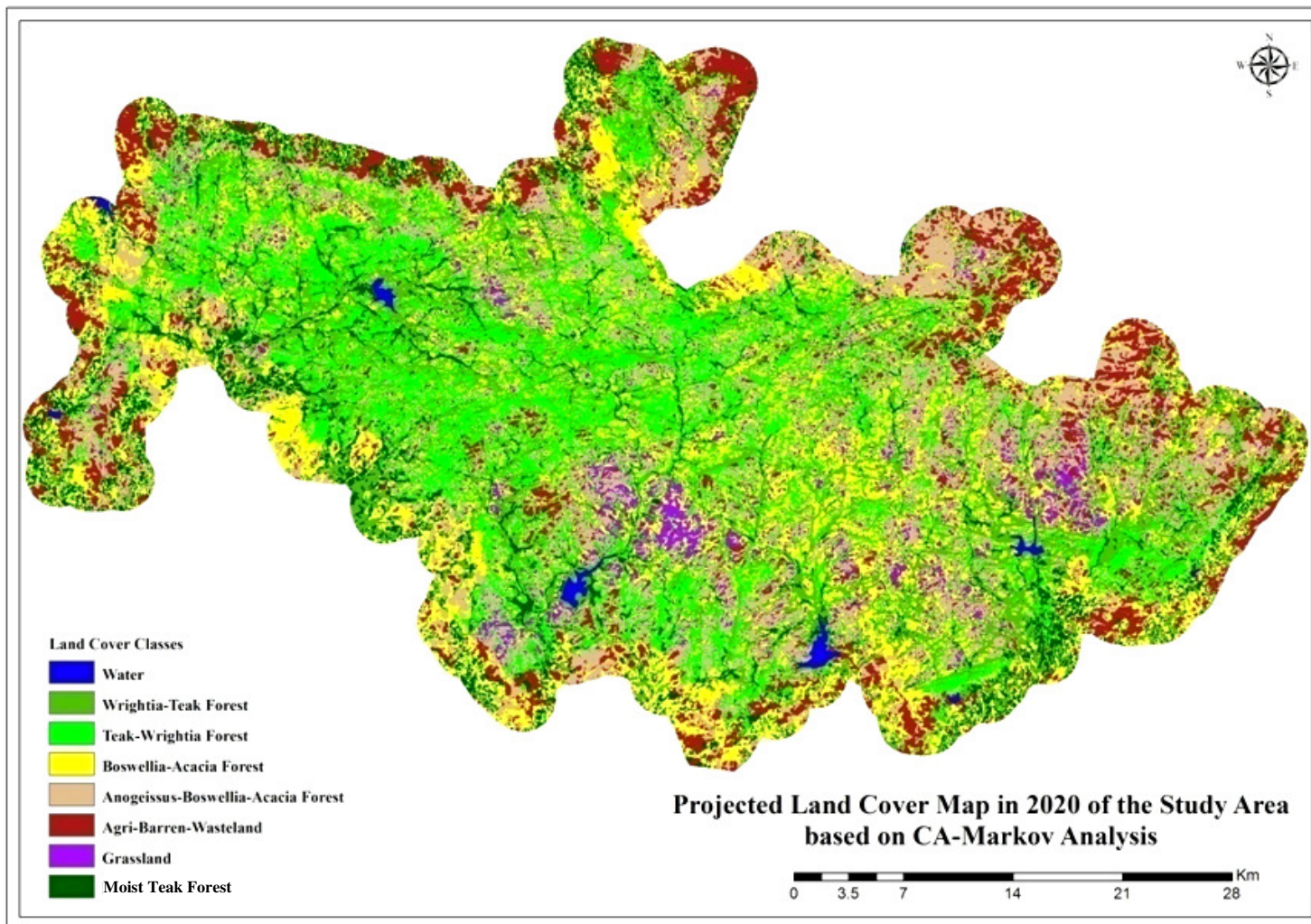


Figure 4.13: Projected land cover map of the study area of the year 2020, based on CA-Markov analysis

Table 4.2: Cross tabulation of simulated land cover map of 2020 with actual land cover map of 2009 (number of pixels)

	Actual Land Cover Map 2009										
	Classes	No Data	Water	<i>Wrightia</i> - Teak Forests	Teak- <i>Wrightia</i> Forests	<i>Boswellia</i> - <i>Acacia</i> Forests	<i>Anogeissus</i> - <i>Boswellia</i> - <i>Acacia</i> Forests	Agri- Barren- Fallowland	Grassland	Moist- Teak Forests	Total
Simulated Land Cover Map 2020	No Data	1966538	0	0	0	0	0	0	0	0	1966538
	Water	153	11909	6	0	0	0	36	0	0	12104
	<i>Wrightia</i> - Teak Forests	0	5343	345442	3378	1694	234	34439	89	30010	420629
	Teak- <i>Wrightia</i> Forests	0	0	52606	340612	27157	713	444	300	15191	437023
	<i>Boswellia</i> - <i>Acacia</i> Forests	0	0	83038	24379	308211	6009	13938	3895	2562	442032
	<i>Anogeissus</i> - <i>Boswellia</i> - <i>Acacia</i> Forests	0	0	0	977	102767	360723	14244	16206	0	494917
	Agri-Barren- Fallowland	0	1047	170	0	6563	100205	146251	4243	0	258479
	Grassland	0	0	184	0	14649	5456	0	87934	0	108223
	Moist Teak Forests	0	0	8396	177	263	187	31417	1256	131454	173150
	Total	1966691	18299	489842	369523	461304	473527	240769	113923	179217	4313095

The model performed well in the ability to specify location correctly ($K_{location} = 71\%$), and also in the ability to specify quantity ($K_{quantity} = 71\%$). Thus the classified map of 2009 was used for the future projection of 2020 (Figure 4.13). A cross-tabulation that describes the changes in land cover classes included in the study is given in Table 4.2. CA-Markov has the ability to simulate transition among any number of classes and the nature of the simulation is bidirectional. The table shown above demonstrates the number of pixels that are expected to change from 2009 to 2020. The diagonal of the matrix indicates the number of pixels that have persisted during the simulation, while the off-diagonal shows the number pixels that changed class.

4.7 Discussion

Land cover changes can influence ecosystems and regional sustainable development directly and/or indirectly by affecting a wide range of processes, such as the movement of nutrients through plants, soils, water and the atmosphere as well as greenhouse gasses and the movement of soil and water within catchments. It is thus very important for governmental decision making at local, national and regional scales to obtain real time information on land cover and its temporal and spatial changes.

The land cover in the study area was found to have changed over the study period; in particular, the areas under teak associated vegetation types found in western, central portion of the Gir PA, as well as areas under *Anogeissus-Boswellia-Acacia* forest, *Acacia-Boswellia* forest types. Land use/cover maps of the study area for the years 1998, 2002 and 2009 were obtained using an object-oriented approach and the derived maps provided new information on spatio-temporal distributions of vegetation types. Results obtained from classification were validated and employed for further change analysis and modeling. A critical analysis of the nature of land use/cover change was addressed and quantified using landscape metrics. Each of the metrics provided information on the nature of each index for the study site. The spatial simulation technique employed, produced satisfactory results that were confirmed by various Kappa summaries. This allowed us to project land change until the year 2020. Future projections presented an acceptable output for short term forecasting. This study has contributed to the understanding of spatial and temporal dynamics of vegetation growth of the Gir protected area and will form a basis for better planning.

A cyclone that swept through Gir in November 1982, uprooted about 2.8million trees (Daniel 1984), and the extraction of these trees continued for a few years, disturbing the harmony of Gir. Literature review stated that the forest cover which was receding in the 1960's and 70's, has consistently improved. Dense foliage has increased across Gir throughout the last two decades and in the time period (1985-2005) forest cover has improved in the forest and a large portion of land became dense or moderately dense forest (Singh 2007). Though any record is absent of the extent of the different qualities of forests before 1960, but records of the last five decades shown that Gir was never as dense as it is today. The trend in land use/cover changes observed in this study supported the ongoing debate of thickening of teak dominated forests type in the study area. The increase in the area of Teak dominated forests resulted from the transition of *Wrightia*-Teak forests into both Teak-*Wrightia* and moist teak forests as almost half of the area under cover type of *Wrightia*-Teak disappeared from this landscape during the study period. The spatial patterns of Teak-*Wrightia* forests expansion indicated the gradual shift of the *Wrightia*-Teak forests type with *Wrightia* domination to the Teak-*Wrightia* forests type with Teak domination. The direction of this change was mainly in central and western part of the study area where teak dominates rather than the eastern part of the sanctuary. The management plan (Singh and Kamboj 1996) prescribed the thinning of forests on an experimental basis; but no action has been taken since the felling of trees in the sanctuary is a highly delicate matter.

This study demonstrated that Gir WLS is effective in preserving dry deciduous forests and lion's habitat against the alarming rate of transformation of forests into human conducive land use classes outside protected area (Jhala et al. 2012). Results were similar to those reported from many protected areas throughout the tropics. Bruner et al. 2001, showed that 93 protected areas in tropical forests were successful in slowing forest loss when the surrounding landscapes were largely deforested. Similar results were presented by a land cover change study of 198 tropical protected areas (Defries et al. 2005) and a detailed study of deforestation in the Sarapiquí region in Costa Rica (Sánchez-Azofeifa et al. 2005). In all of these studies the protected areas were surrounded by rapidly changing landscapes. Although they became increasingly more isolated, they maintained high forest cover within their boundaries, potentially providing strongholds for species and biodiversity conservation (Songer et al. 2009).

Multi-scale species distribution modeling for Asiatic lions and its major prey species, understanding the impact of spatial scale in species distribution modeling

5.1. Introduction

Managing wildlife populations depends largely upon understanding and predicting their habitat requirements (Clark et al. 1993), and evaluating habitat feature precisely (Van Horne 1983). Information of the habitat requirements of populations and individuals of a species is essential for planning successful conservation efforts (Pereira and Itami 1991, Akcakaya and Atwood 1997, Gibson et al. 2004). Habitat suitability, defined as the potential to support a species (Clark et al. 1993), is a measure of the presence of essential environmental parameters, and relates to abiotic and biotic environments. Maps demonstrating the spatial arrangement of suitable habitat for a species can be a support to define the actions required to manage their habitat properly, and it is now accepted that the spatial prediction of species distributions form a core component of conservation planning (Manel et al. 1999, Guisan and Zimmermann 2000, Jaberg and Guisan, 2001, Austin 2002, Scott et al. 2002).

Recent research commends that patterns in population abundance are a product of the distribution of resources essential to fulfill the species niche requirements (Brown et al. 1995). The size and spatial arrangement of suitable habitat can influence the long-term persistence of some faunal species, predominantly if the species is susceptible to habitat destruction (Lindenmayer and Possingham, 1996, Lindenmayer et al. 1999). Recognizing the spatial relationship amongst organisms and environmental features is unquestionably significant for understanding autecology of species (Cowles 1899, Grinnell 1917). Resource and habitat selection may benefit species coexistence as well as being essential driving forces in evolution and speciation (Lack 1933, McPeck 1996, Morris 2003). Understanding the basis of habitat choice thus has important consequences for explaining the distribution of organisms, as well as helping to

differentiate among habitats of different quality for effective management. Information of those habitat features indispensable for the sustainability of a species can also provide vital information to wildlife managers dealing with reintroductions, translocations and the development of new protected areas (Araujo and Williams 2000, Rotenberry et al. 2006, Stamps and Swaisgood 2007).

As a useful tool in ecology, biogeography and evolution, predictive modeling of species' distributions has gained increasing importance (Guisan and Thuiller, 2005) and has been effectively applied in conservation ecology (Rushton et al. 2004, Rodríguez et al. 2007). A wide variety of statistical and machine learning approaches have been introduced, mostly in conjunction with geographic information systems (GIS) and remote-sensing (Pereira and Itami 1991, Fitzgerald and Lees 1992, Aspinall and Veitch 1993, Franklin et al. 2000). Many of the alternative statistical methods have also been developed (Guisan and Zimmermann 2000). Several forms of regression analysis prevail in the literature as for example, generalised linear models (Austin and Cunningham 1981, McCullagh and Nelder 1989) and generalised additive models (Hastie and Tibshirani 1990, Yee and Mitchell 1991). Logistic regression using presence-absence survey data appear to be progressively popular as the statistical model to be used (Franklin 1995, Guisan and Zimmermann 2000, Scott et al. 2002). But the absence of information about the areas where species are absent complicates the use of common ecological modelling tools, such as logistic regression or classification and regression trees (Guisan and Zimmermann 2000).

Species distribution models have used species–environmental relationships to predict the distribution of species across complex landscapes (Verner et al. 1986, Guisan and Zimmermann, 2000, Manly et al. 2002). The distributions of abundance in animal populations are spatially heterogeneous (Brown et al. 1995, Ives and Klofer 1997, Keitt et al. 2002). This heterogeneity can expressively affect population dynamics through parameters such as breeding performance, survival rates and dispersal (Aars and Ims 2000, Paradis et al. 2002). Furthermore patterns in the abundance of one or a small number of species may have crucial effects on the structure and functioning of many communities through density-dependent interactions with other species and large scale alterations of the structure of vegetation (Paine 1992, Palomares et al. 1995, Weltzin et al. 1997, Kotliar 2000).

Fitting a species distribution model involves a series of steps, each demanding a number of choices and well-justified decisions (Ferrier et al. 2002, Guisan and Thuiller 2005). Grain (resolution) or scale is one key factor that may affect predictions together with study extent, a component of spatial scale (Wiens 2002) and a major feature in ecology (Holling 1992) and ecological modelling (Huettmann and Diamond 2006). Scale has been given importance by many ecologists when obtaining and interpreting ecological data. Scalar features of ecological observation and analysis have turned out to be common, and now considered important in prioritizing research objectives (Levin 1992), designing organism-centered sampling methods (Wiens 1989), and extrapolating process from observed patterns (Turner et al. 1989, Turner 2005). These ideas have encouraged a growing number of “multi-scalar analyses” intent on describing ecological phenomena at more than one observational scale.

These models of species’ distributions quantify the relationships between species and their environments, helping in identifying the factors that determine the observed distribution patterns and thus to guide conservation planning, particularly in endangered species (Rushton et al. 2004). Nevertheless actual conservation actions need predictive and precise habitat models (Heglund 2002), which in turn claim better assessments of error and uncertainties at an appropriate scale (Guisan and Thuiller 2005). The selection of the suitable scale for modeling species-habitat relationships has become an important issue (Austin 2002, Guisan and Thuiller 2005), as the results of habitat models can depend upon the spatial scale at which the variables are measured (Lantz et al. 2007). To deal with this problem, a hierarchical habitat modeling approach at consecutively smaller scales has recently been employed (Martínez et al. 2003, Berg 2008). However, regardless of the substantial advances of these methods in recent years, limited analyses have explored the combination of different scales to obtain a final integrated result that might improve predictive ability (Anadoń et al. 2007, Lantz et al. 2007, McAlpine et al. 2008).

Important questions therefore include: Is there an optimal grain for fitting species distribution models? What is the effect of changing grain size on species distribution model performance? Choosing a grain size for modeling is partially a technical issue. For example, grain size is linked to the grid cell size of available environmental data (Graham et al. 2004), characteristics of the species data (e.g. geographical accuracy,

sample size, field survey constraints, or autocorrelation structure) (Guisan and Hofer 2003, Gottschalk et al. 2005, Linke et al. 2005, Huettmann and Diamond 2006) or computer power (i.e. too many cells may require too demanding computer resources). Grain size is also a vital ecological as well as management issue. Changing the grain size can affect the sensitivity of a phenomenon, such as patterns of presence or abundance (Johnson et al. 2002, Tobalske 2002, Wiens 2002, Graham and Hijmans 2006), or affect the significance of the output for management applications (Araújo et al. 2005).

Few reasons are there why altering grain size (spatial scale) could have significance on the performance of species distribution models. For instance, at a fine grain (fine scale), the risk that a wrong geographical location of a species record samples a cell representing a different habitat than the one where the species actually occurred increases, while the opposite will be observed when aggregating data towards coarser grains. Though, when coarsening the grain, the risk of a fitting environmental conditions that do not occur together but nearby in the field increases and can make the model recognize false combinations of suitable environmental conditions for a species. This is likely most vital for sessile organisms (Guisan and Thuiller 2005).

Ecological patterns depend on processes acting at different scales. For instance, at a regional scale, climate and history primarily explain species distribution, but at local scale, biotic interactions are the main process regulating distributions (Levin 1992). Ecological processes are also all together influenced by issues acting across a wide range of scales. The choice of scales in ecological studies thus certainly influences the results that can be demonstrated. This choice should therefore depend on the species studied and the questions raised (Wiens 1989). Wildlife research and management have conventionally focused on small spatial scales. More in recent times it has been recognized that animals respond to habitat factors at coarser spatial scales (Freemark and Merriam 1986). As each species responds to the environment at a unique range of scales (Levin 1992), there is no single accurate spatial scale at which to describe species-habitat relationships (Wiens 1989). Thus multi-scale approaches are necessary (Bissonette 1997, Cushman and McGarigal 2004) and are indeed becoming more and more common in studies of species-habitat relationships (Carroll et al. 1999, Fuhlendorf et al. 2002, Lawler and Edwards 2002, Thompson and McGarigal 2002,

Zabel et al. 2003, Fischer et al. 2004). Many studies, however, are conducted at a few random chosen scales (Zabel et al. 2003, Johnson et al. 2004), only few include a distinct range of scales (Fuhlendorf et al. 2002, Lawler and Edwards 2002), and very few explore species-habitat relationships along a continuous range of scales (Thompson and McGarigal 2002).

Multi-scale approaches have been developed to study ecological systems (Cushman and McGarigal 2002, Grand and Cushman 2003) and to explore the significance of a range of scales in the distribution of animals and in the structure of their communities (Herrando and Brotons 2002). The rapidly advancing GIS technology and new powerful statistical tools have helped to address spatial scale questions in species-habitat relationships (Guisan and Zimmermann 2000, Manly et al. 2002). Landscape ecologists are perhaps most familiar with the significance of spatial scale in the design of ecological studies. Matching the grain and the extent of the measurement of pattern with the scales at which processes in question work is often critical to understand the system under study (Wiens 1989). Many tools have been proposed to describe spatial heterogeneity (Gustafson 1998, Cullinan and Thomas 1992, McGarigal and Marks 1995). But, as each species, including humans, respond to the environment on its own suit of scales in space and time (Kolasa and Rollo 1991, Wiens et al. 1993), the spatial pattern of their actions is very complex and interwoven. It is thus difficult task to relate patterns described by such metrics to biodiversity as a whole (O'Neill et. al. 1996, McGarigal and Marks 1995).

5.2. Justification of the Study

Accelerating habitat changes magnified the importance of understanding the role of habitat characteristics in determining the species distribution. In this context, the primary concern of this study was to apply remote sensing and GIS techniques to map areas that vary according to their habitat potential for lions and its major ungulate prey species in the Gir protected area, recognizing that comprehensive field surveys could not be performed and study the impact of spatial scales in the habitat suitability prediction. Animal populations are often sustained by a subset of the total area used by the population, due to variances in habitat suitability across the landscape (Weins and Rotenberry 1981, Pulliam 1988). By means of predictive species distribution and

ecological niche modeling, we analyzed the species habitat relationships of these species at different spatial scales by varying the grain size. Analyzing habitat associations in this way assumes that animals recognize different types of patterns at different spatial scales – generally that they distinguish finer detail in habitat structure or heterogeneity at finer spatial scales (Hildén 1965, Hutto 1985).

5.3. Objectives

The primary objectives of this study were to:

- (1) Determine whether the influence of environmental predictors on species' distribution was scale-dependent;
- (2) Determine which remotely sensed variables contribute most to the species distribution models;
- (3) Identify threshold effects where changes in environmental variables abruptly influence species occurrence.

Hypothesis for the objective:

For modeling habitat suitability for lions:

Ho: Lions do not exhibit habitat preference i.e. use habitats in proportion to their availability.

Ha: Lions exhibit habitat preferences i.e. use some habitats more in comparison to their availability.

For modeling habitat suitability for principal prey species:

Ho: Ungulates use habitats in proportion to their availability i.e. they do not exhibit habitat preferences.

Ha: Ungulates use some habitats more in comparison to their availability and they exhibit habitat preferences.

5.4. Methodology

The whole study area was divided into grids of 2km X 2km and systematically searched for the sign of herbivore species (pellet) (Figure 5.1). In each of these sampling grids, one 2.5 km transect was laid diagonally covering most of the area of the grid. In this transect, at 250 meter intervals, plots of 5 meter radius were laid (n=900) to collect data on species presence. Sampling took place 6 days a week from

the month of October 2010-January 2011. Environmental variables thought to determine utilization of habitat by animals were extracted to 500 m X 500 m, 1 Km X 1 Km, 2.5 Km X 2.5 Km, 5 Km X 5 Km grids covering the entire study area. For each pellet sampling plot, vegetation around were quantified (described in chapter three, section 3.6). The ground data describing vegetation types was used with satellite imageries to formulate vegetation map. The vegetation map was used as a template describing habitat for the animals. Distance to the nearest water was computed using the GIS. Land use/cover data, Climatic variables (Table 5.1), Normalized Difference Vegetation Index (NDVI), gross primary production, elevation, human disturbance, and fire occurrence points were obtained from open GIS archives (Figure 5.2, 5.3, 5.4). Since this study was done at local scale, so all climatic variables were not used in all models, since there was very less difference in values of different variables. Projections, grid cell size and alignment, and spatial extent were manipulated to ensure consistency across all data layers using Arcmap 9.2 (Environmental Systems Research Institute, Redlands, CA, USA). All files were projected to UTM 42N (WGS84 datum) with a grid cell size of 500m, 1km, 2.5km and 5km.

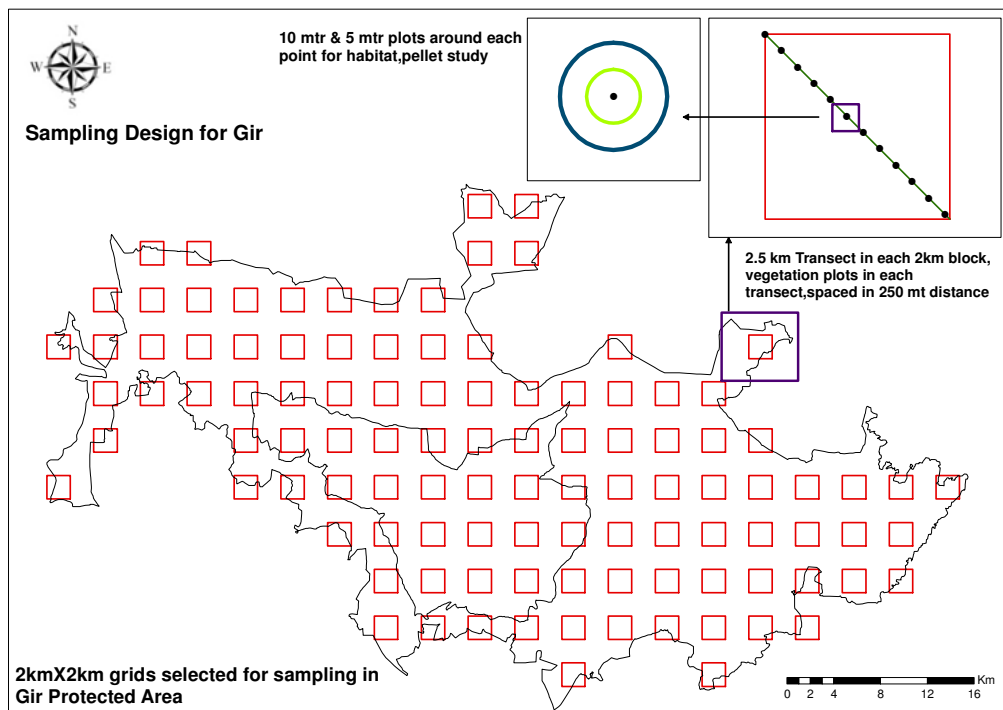


Figure 5.1: Sampling Design for field studies

Table 5.1: Different variables used in the species distribution model along with their sources

S. No	Type	Abbreviation	Variables	Reference
1	Habitat Composition	LULC	Land use/cover information	FSI,2009, vegetation map generated from the study
2	Habitat Configuration		Landscape indices	FSI,2009, vegetation map generated from the study
3	Topography	DEM	Digital elevation model	http://asterweb.jpl.nasa.gov/gdem.asp
4	Water Resource	EURVR	Euclidean distance to river channels	Digitized dataset
5	Anthropogenic Pressure	EURD	Euclidean distance to roads	Digitized dataset
6	Anthropogenic Pressure		Human footprint	http://sedac.ciesin.columbia.edu/data/set/wildareas-v2-human-footprint-geographic
7	Climatic Variables	BIO1	Annual Mean Temperature	Hijmans et al. (2005)
8	Climatic Variables	BIO2	Mean Diurnal Range (Mean of monthly (max temp - min temp))	Hijmans et al. (2005)
9	Climatic Variables	BIO3	Isothermality (BIO2/BIO7) (* 100)	Hijmans et al. (2005)
10	Climatic Variables	BIO4	Temperature Seasonality (standard deviation *100)	Hijmans et al. (2005)
11	Climatic Variables	BIO5	Max Temperature of Warmest Month	Hijmans et al. (2005)
12	Climatic Variables	BIO6	Min Temperature of Coldest Month	Hijmans et al. (2005)
13	Climatic Variables	BIO7	Temperature Annual Range (BIO5-BIO6)	Hijmans et al. (2005)
14	Climatic Variables	BIO8	Mean Temperature of Wettest Quarter	Hijmans et al. (2005)
15	Climatic Variables	BIO9	Mean Temperature of Driest Quarter	Hijmans et al. (2005)
16	Climatic Variables	BIO10	Mean Temperature of Warmest Quarter	Hijmans et al. (2005)
17	Climatic Variables	BIO11	Mean Temperature of Coldest Quarter	Hijmans et al. (2005)
18	Climatic Variables	BIO12	Annual Precipitation	Hijmans et al. (2005)
19	Climatic Variables	BIO13	Precipitation of Wettest Month	Hijmans et al. (2005)
20	Climatic Variables	BIO14	Precipitation of Driest Month	Hijmans et al. (2005)
21	Climatic Variables	BIO15	Precipitation Seasonality (Coefficient of Variation)	Hijmans et al. (2005)
22	Climatic Variables	BIO16	Precipitation of Wettest Quarter	Hijmans et al. (2005)
23	Climatic Variables	BIO17	Precipitation of Driest Quarter	Hijmans et al. (2005)
24	Climatic Variables	BIO18	Precipitation of Warmest Quarter	Hijmans et al. (2005)
25	Climatic Variables	BIO19	Precipitation of Coldest Quarter	Hijmans et al. (2005)

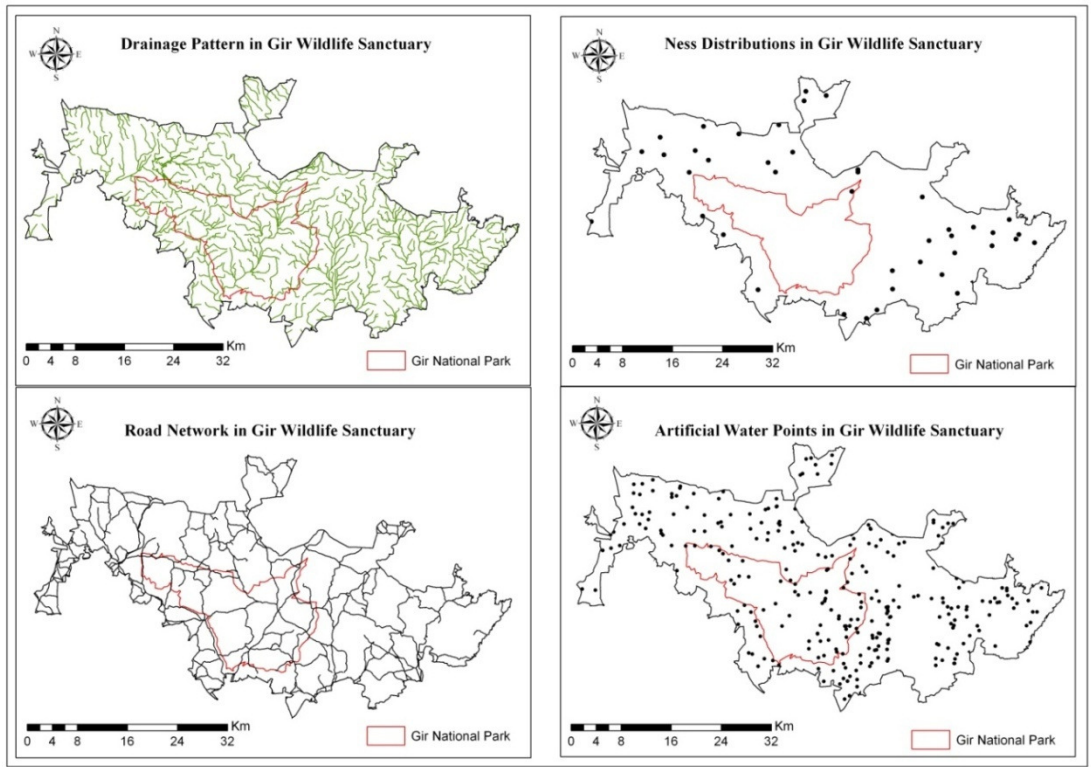


Figure 5.2: Drainage pattern, road network, nesses locations, artificial waterholes distribution within Gir protected area

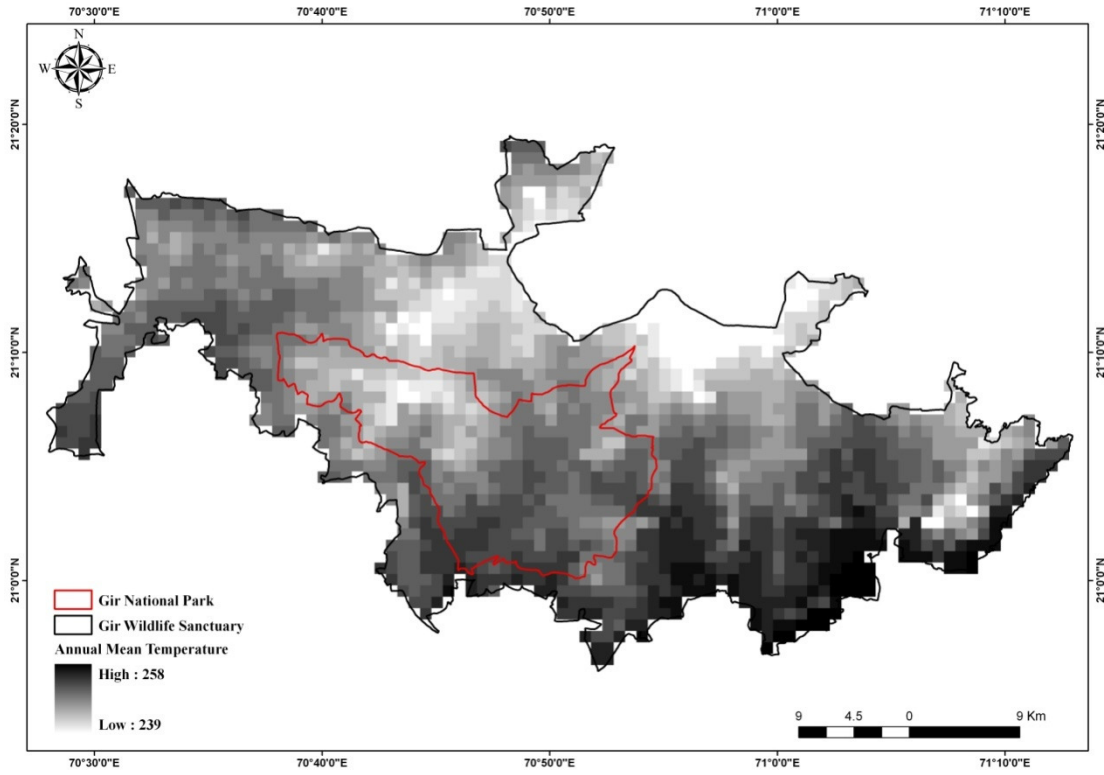


Figure 5.3: Annual Mean Temperature profile of Gir

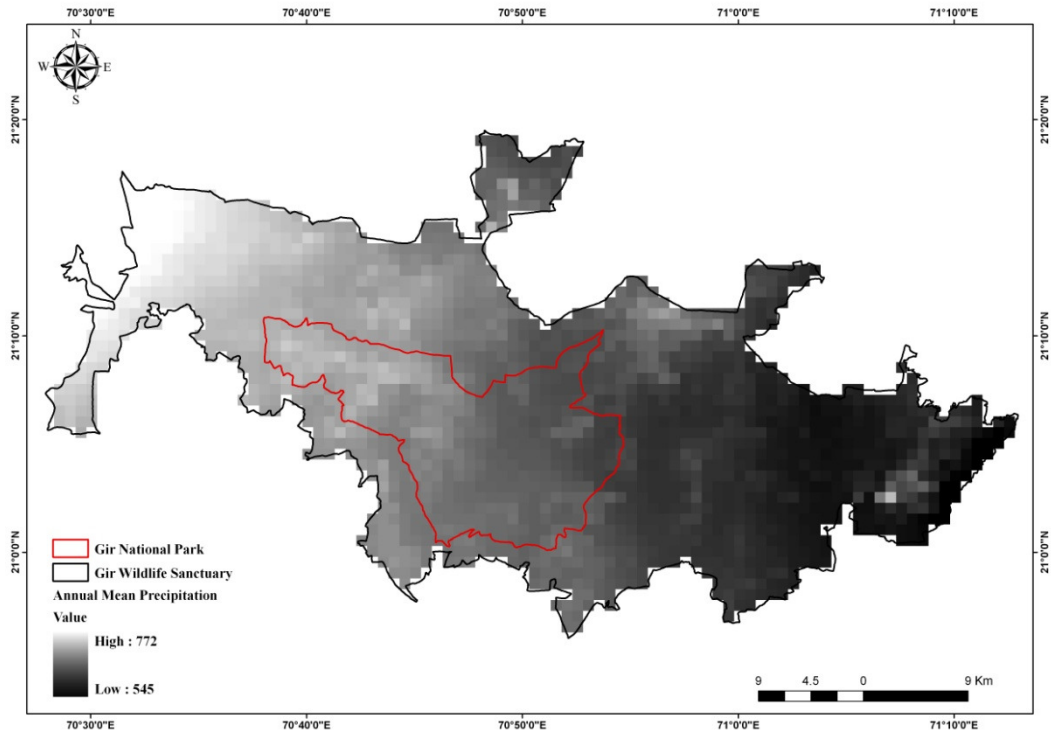


Figure 5.4: Annual Precipitation profile of Gir

5.5. Study species

Information on species account, history, past and present distributions and the present conservation scenario of Asiatic lions (*Panthera leo persica*) is explained in the introduction and other chapters (chapter 6A and 6B). Here I have discussed about the principal prey species of lions.

The revolutionary work of Schaller (1967) in Kanha Tiger Reserve was the first ecological description of a number of the common ungulate species found in the Indian subcontinent. Later, there have been numerous studies on ungulates in this region (Eisenberg and Lockhart 1972, Berwick 1974, Saratchandra and Gadgil 1975, Dinerstein 1980, Mishra 1982, Johnsingh 1983, Barrette 1991). Facts on different population characteristics (e.g. grouping structure, densities, and age-sex ratios) have contributed significantly towards a better understanding of these ungulate species.

In its distributional range in Africa, Lions (*Panthera leo*) prefer large prey species within a weight range of 190 to 550 kg, irrespective of their availability. Yet they

predominately take prey sizably smaller than this reflecting their opportunistic hunting behavior. In the Serengeti, they preferred prey ranging from 170 to 250 kg (Sinclair et al.2003), which is lower but comparable to that found here for lions throughout their range. While the modal prey size of felids is usually less than their body weight (Packer 1986), lions are expected to take prey >45% of their body mass as they are larger than Carbone et al.'s (1999) 21.5 kg threshold, but the preferred weight range of lions is greater than 100% of their body mass.

This study focused on lions and its major prey species in the Gir protected area. The wild herbivores and the livestock are found as the chief prey base for lions in Gir protected area (Singh et al. 1999). as 78% of the lion scats contained livestock (Joslin 1973). The Gir habitat has improved since the declaration of the sanctuary and implementation of the Gir lion sanctuary project in 1972. This has also caused in increases in the herbivore population (chital, sambar, blue bull, chinkara, four horned antelope, and wildpig) from 9600 in 1974 to over 38,200 in 1995. Ungulate and their habitat in Gir have been studied by Berwick (1974) and Khan (1990). Sinha (1987) reported of 52% of the lion scats containing wild prey animals. In another study livestock remains were found in 32.74% of the scat and wild prey remains were found in 51.69%. The wild prey chiefly included Chital (*Cervus axis*), Sambar (*Rusa unicolor*) and Nilgai (*Boselaphus tragocamelus*). Chellam's study (1986-90) showed that the lions' food pattern had changed in favour of wild prey, with 65% of lion diet recorded as coming from wild animals and 35% from livestock kills. In another study Sambar represented 15% of the lion's diet while chital represented 43% (Chellam 1993).

The intensification in cover inside National park suits the habitat requirements of sambar but has been largely unfavorable for chital and nilgai, which prefer open areas with ample ecotones (Schaller 1967, Dinerstein 1980). Khan 1994 mentioned about cautious habitat management in National Park and Sanctuary east to increase the chital population. It was advised to create open grasslands and ecotones in valleys and flat areas of National Park through patch cutting, by promoting the growth of palatable *Acacia* and *Zizyphus* species and providing ample water in such areas. Similarly the dominance of a few species of shrubs is not beneficial for ungulates, and such areas require selective clearing.

The high grazing pressures in Sanctuary west and east from the livestock population of *Maldharis* and surrounding villages, weeds like *Cassia tora* and *Achyranthes aspera* started dominating the grass cover. The diverse and palatable shrub layer that contributes much to the maintenance of a high ungulate biomass might become affected by the spread of unpalatable *Lantana camara* as a consequence of draught induced plant mortality (Khan et al. 1994) and could lessen the food availability for ungulates. Among all ungulate species chital was common and widely distributed throughout Gir PA, and contributes 92.7 % of the total wild ungulate density. Sambar was the second most abundant species in Gir after chital (2.86 ± 0.81). Species that preferred moister vegetation communities like sambar were found in higher densities in the western and central parts of Gir (Dave 2008). The sambar density has increased steadily over last three decades (Berwick 1974, Khan et al. 1996, Goyal et al. 2004).

Notes on principal prey species

Blue bull or Nilgai (*Boselaphus tragocamelus*): Nilgai (*Boselaphus tragocamelus*) are the largest antelope in Asia, about the size of a horse, occur near human habitations and crop fields outside protected areas. They are found in a range of habitats, from level ground to undulating hills, in thin brush with scattered trees to cultivated plains, but not in dense forests and steep hills (Blanford 1888, Prater 1971). They are absent from the true arid zone where woody cover is inadequate to meet their requirements. Though nilgai were once common throughout India (Adams 1858, Blanford 1888), like most large mammalian fauna of India, they have declined significantly because of habitat destruction and over-hunting (Schaller 1967). North-central India, covering four states (Haryana, Madhya Pradesh, Rajasthan, and Uttar Pradesh) account for a large majority (about 60%) of the total Nilgai population in the country (Figure 5.5).

Nilgai are to a degree social in their habits (Roberts 1977), although congregations and large groups are rare. In Gir, mean group size of Nilgai (including “groups” of one) was described as 2.2, with high seasonal variability in group sizes (Khan et al. 1995). In Gir, down the years, the female-biased sex ratio of Nilgai seemed to be leaning more towards females: from 0.89:1 (Berwick and Jordan 1971) to 0.71 (Khan et al. 1995).

Studies on Nilgai food habits showed that they are browsers (Berwick 1974, Mirza and Khan 1975, Dinerstein 1979, 1980) or mixed feeders (Haque 1990, Sankar 1994). A study on ungulate food habits in Nepal (Dinerstein 1979) showed that sambar and Nilgai feed on the same browse species. Apart from this, there is very little information existing on the dietary overlap between nilgai and other wild ungulates. The large size of Nilgai meant they can exist on poorer quality food items, making them coarser browsers (Rodgers 1988). They are also found of raiding crops and are regarded as pests in agriculture fields.

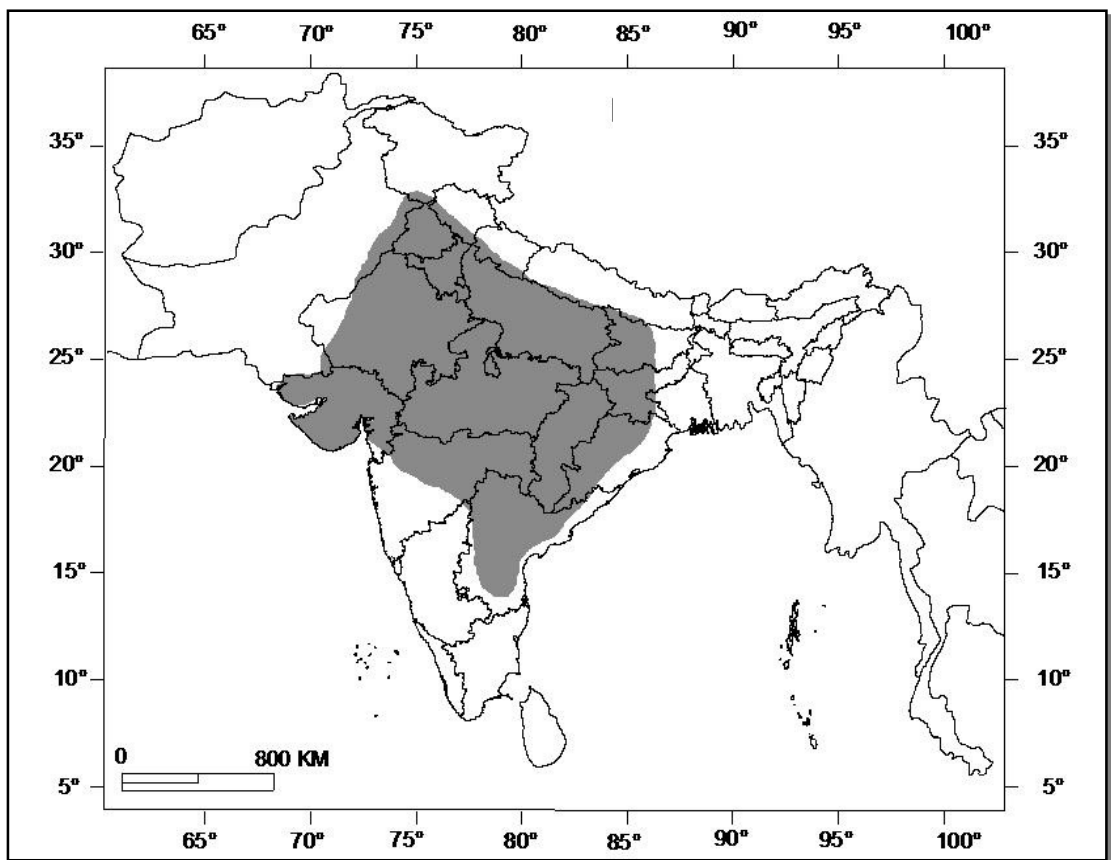


Figure 5.5: Distribution of Nilgai in India (Source: WII, GIS cell)

Nilgai are found in open areas with undulating or flat terrain (Berwick 1974); they avoid dense hilly forests and prefer scrublands with low tree and shrub densities (Chakroborty 1991, Sankar 1994, Khan 1996). A radio collared nilgai female had a mean seasonal home range of 3.6 km² in Sariska (Sankar 1994), and its annual home range was 7.3 km². In Texas, a nilgai female had a 0.6 km² home range and the mean home range of eight bulls was 4.7 km² (Sheffield et al. 1983).

Sambar (*Rusa unicolor*): Sambar (*Rusa unicolor*) is considered as the largest deer species native to South and South East Asia. Adult sambar stags weigh between 225 and 320 kg. Sambar hinds are smaller and weigh between 135 and 225 kg (Lydekker 1916, Crandall 1964, Downes 1983). Sambar is found in a broader diversity of forest types and environmental surroundings (Schaller 1967). Within India sambar occur in the thorn forests of Gujarat and Rajasthan, in the moist deciduous forests throughout peninsular India, in the pine and oak forests at the Himalayan foothills, and in the evergreen and semi evergreen forests of North-Eastern India (Figure 5.6).

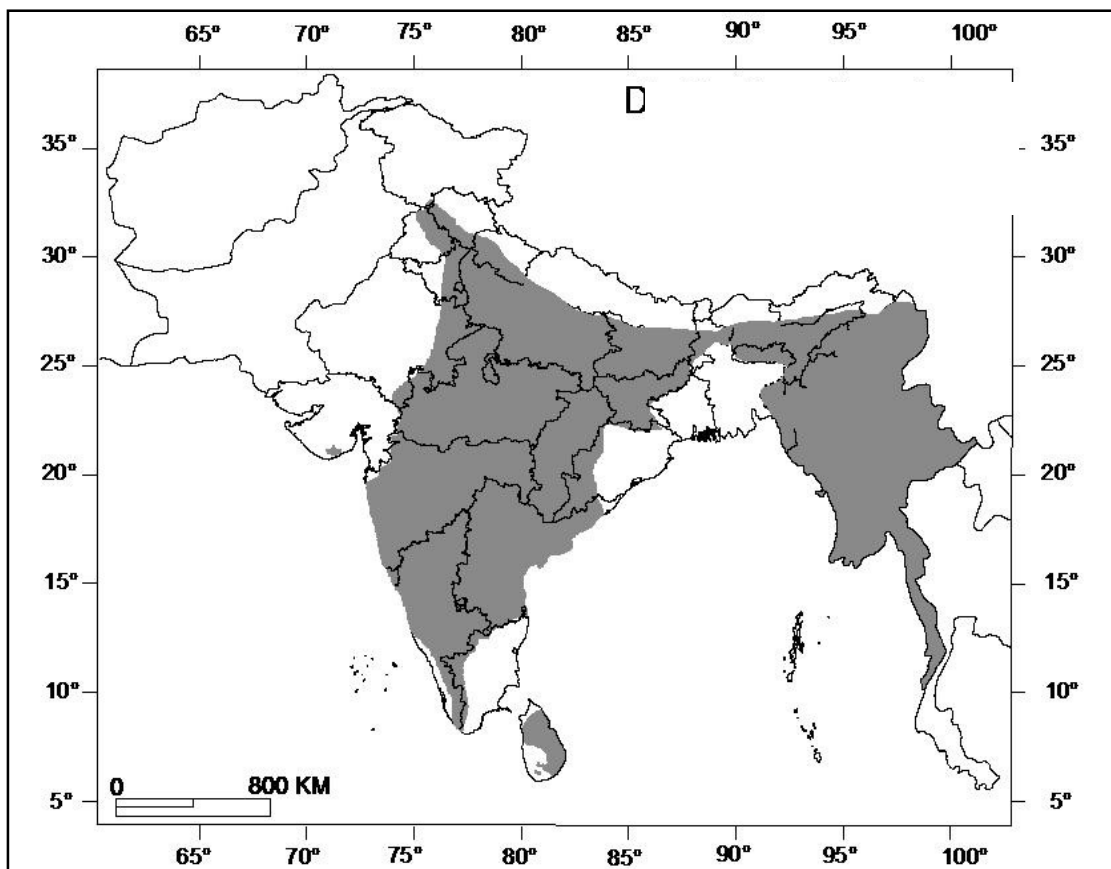


Figure 5.6: Sambar distribution in India(Source: WII, GIS cell)

Sambar has been recorded to occur in 208 protected areas of India found in group of small numbers, fewer than six individuals (Schaller 1967). The characteristic social unit is one hind and one fawn (Schaller 1967, Kelton 1981, Downes 1983). Average group size of sambar is reported to be 4 to 5 individuals (Jerdon 1874 and Prater 1971). Sambar group sizes ranged from one to five individuals in Gir (Khan et al. 1995). Sambar has been observed to feed on more than 139 species of plants (Schaller

1967, Johnsingh and Sankar 1991). The food requirements of sambar are less specialized than those of other deer (Schaller 1967). Young green grasses were found preferred forage of sambar in kanha but browse is often important during seasons when green grasses are scarce (Schaller 1967).

In Sariska, the mean home range of sambar stags was about 4 km² and for sambar hinds was 1.7 km² (Sankar 1994). The estimated annual home range of sambar stags was nearly 15 km², whereas that of hinds was approximately 3 km². The preference of sambar for heavy cover is recorded (Schaller 1967, Johnsingh 1983) and perhaps be the reason why sambar hinds, largely browsers, had smaller home ranges. Water is an important component of the sambar's home range, especially when the temperature is hot. Being an animal of hilly terrain, it is unusual for sambar to travel long distance to drink water. Sambar constitute one of the largest, and in turn, the most favoured prey species of large carnivores such as the tiger, leopard and dhole as reported from kanha (Schaller 1967), Bandipur (Johnsingh 1983) Rajaji National Park (Johnsingh et. al. 1993) and Sariska (Sankar 1994). Next only to chital, sambar are found as the second most important prey species of the large carnivores in India.

Spotted Deer or Chital (*Axis axis*): Chital or spotted deer (*Axis axis*) is the third largest deer inhabiting the plains and undulating terrain of India. Found in a variety of forest types in India viz. dry deciduous, moist deciduous, thorn and mangrove forests, dry deciduous habitats with scrub serve the preferred habitat for chital (Eisenberg and Seidensticker 1976). Declined drastically throughout their range, Chital are now only locally abundant within 123 protected areas of India and some forest tracts (Figure 5.7). Chital are mainly social animals, rarely seen as solitary individuals. The basic social unit of chital is consisting of an adult female, her offspring from the previous year and a fawn (Ables 1974). The usual herd is composed of two or more such family units and is often accompanied by individual deer or mixed sex and age classes.

Chital are known to feed on more than 160 species of plants (Schaller 1967, Johnsingh and Sankar 1991) and primarily considered as a grazer (Schaller 1967, Mishra 1982). On the basis of morpho-physiological ruminant feeding types, chital is categorized as an intermediate/ mixed feeder (Hoffman 1985) while with a diet

consisting of grasses, forbs, and leaves of woody plants, chital was categorized as a generalist feeder (Rodgers 1988). In Sariska, chital was a grazer as long as green grasses were available (monsoon and post monsoon seasons), but switched over to fallen leaves, flowers and fruits in winter (Sankar 1994).

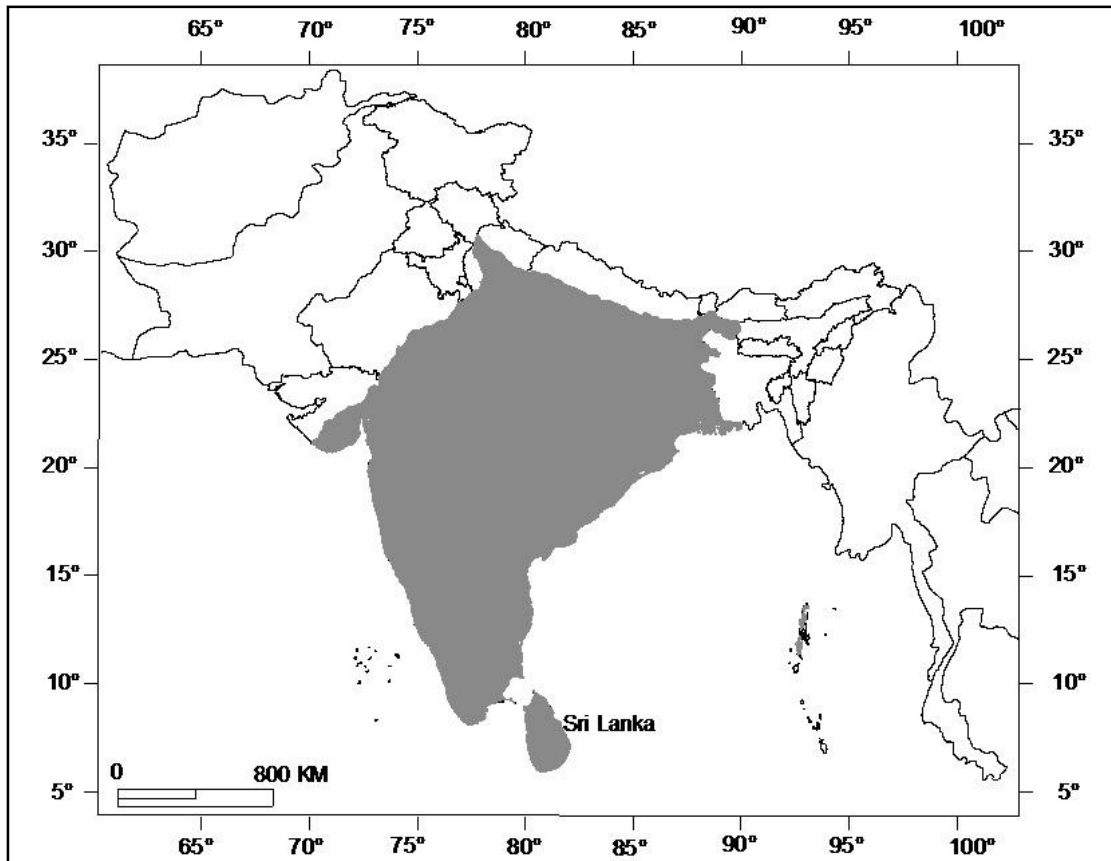


Figure 5.7: Cheetal Distribution in India (Source: WII, GIS cell)

In Sariska, the mean home range of male chital stag was found 3.5 km^2 , and that of a chital doe was around 2.5 km^2 while the estimated annual home range of a chital doe was around 16 km^2 (Sankar 1994). Annual mean home range of chital does in Karnali-Bardia was about 1.4 km^2 , and that of stags was about 2 km^2 (Moe and Wegge 1994).

Chital form one of the important prey of top carnivores in kanha (Schaller 1967), Bandipur (Johnsingh 1983), Rajaji National Park (Johnsingh et. al. 1993), Sariska (Sankar 1994), Pench (Biswas and Sankar 2002) and Ranthambore (Bagchi et al. 2003). Though the species has thrived well and is now locally abundant within protected areas, the remaining population is vulnerable to poaching, habitat

destructions, and livestock borne diseases and conservation and management of chital populations is of supreme importance in reducing large carnivore depredation of livestock and therefore, mitigate the increasing levels of human-wildlife conflict.

5.6 Species Distribution Modelling (SDM)

5.6.1 Introduction to Species Distribution Modelling

An explanation and a more detailed discussion of the technical details of all methods of SDM are out of the scope of this thesis but can be found in literature (e.g. Guisan and Zimmerman 2000, Elith et al. 2006, Elith and Leathwick 2009). SDM approaches differ from each other in terms of use of algorithm, and different names applied to them, such as: species distribution models (SDMs), habitat suitability (HS) models, ecological niche models, bioclimatic models, selection functions and correlative models (Elith and Leathwick 2009). These are used in delimiting population distributions, resource availability and habitat utilization (Elith and Leathwick 2009) and all relate species distribution data (presence or abundance at known locations) to the eco-geographical variables (EGVs) of these locations.

SDM is used to extrapolate the niche model or HS output to areas for which the spatial environmental data is available but no species distribution data is available due to the logistical difficulty or cost of carrying out adequate species sampling at all sites of interest. Thus, output of SDM can be used in designing studies in areas where previous record of the species is absent (Gibson et al. 2004) or identifying areas of high and low suitability for the focal species (e.g. Hirzel et al. 2001). Most of the earlier studies on SDMs was focused to gain ecological insight into determinants of species distributions (Mac Nally 2000), but the trend shifted towards SDM application for mapped products for conservation and land management (Elith and Leathwick 2009). Application of SDMs are now available in quantitative ecological studies, evolutionary biology, integrating SDMs with a phylogenetic approach to explore speciation (Leathwick and Austin 2001, Graham et al. 2004) etc. SDMs differ in the use of distribution data either presence-only (PO) or presence/absence (PA) data.

5.6.2 Presence/Absence data (PA data)

Techniques like logistic regression and classification and regression trees rely on the use of PA data (Guisan and Zimmerman 2000, Segurado and Araújo 2004). These methods relate PA data to a single (simple regression) or a combination (multiple regression) of EGVs (Guisan and Zimmerman 2000). Boosted regression trees (BRT; Elith et al. 2006) and multivariate adaptive regression splines (MARS; Leathwick et al. 2006; Elith and Leathwick 2007) are considered as the other regression techniques. Artificial Neural Networks (ANN) is a non-regression-based, PA technique implemented in the software SPECIES (Pearson et al. 2002). PA techniques have gained importance over time in better model performance over that achieved by PO techniques (Brotons et al. 2004), hence in absence of authentically collected absence records, authors recommended using “pseudo absence” data to make use of the PA framework (Osborne et al. 2001, Stockwell and Peterson 2002). Pseudo-absences data is generated in one of the three ways, to supplement PO approach: (i) randomly selecting points across the entire study site representing absences (e.g. Stockwell and Peters 1999); (ii) selecting random absence points as in (i) but weighting them in favour of those areas confirmed to contain ‘true’ absences (Zaniewski et al. 2002); (iii) including absence points identified from a circular buffer area around each presence point (Hirzel et al. 2001).

Sometimes PA approaches suffer from incorporating ‘false absences’, when absence data are unreliable and this is prone to bias regression models (Gu and Swihart 2004). The focal species may be considered absent from a location for one of three reasons: (i) no detection of the species at the time of surveying but it was present (Kéry 2002) (ii) the habitat is suitable yet the species is absent due to some unknown reasons including, but not limited to, colonization patterns and dispersal (Svenning and Skov 2004) (iii) because the habitat is truly unsuitable. Hence species absence data don't give any indication of habitat quality or suitability and could then impose several limitations to the effective use of PA approaches. This has demanded the need of sound ecological knowledge of the focal species in deciding of approach to utilize (Guisan and Zimmerman 2000, Austin 2002), largely with respects to the species' existence and tolerance of environmental change (Hirzel et al. 2001, Brotons et al.

2004). Therefore the practice of PO-based approaches is recommended in most of the cases (Hirzel et al. 2001, Pearce and Boyce 2006, Elith et al. 2006).

5.6.3 Presence-only data (PO data)

PO data can be used by two different approaches to produce a SDM. One depends only on presence records, without reference to other samples from the study area, while the second approach uses ‘background’ environmental data from the entire study area, focusing on how the environment at species locations differs from that available over the study area as a whole (the background’). Examples of the first approach include the Gower metric, implemented in the software DOMAIN (Carpenter et al. 1993). Data in this format has most commonly been analyzed using environmental envelope approaches, particularly using software such as BIOCLIM (Busby 1991) and HABITAT (Walker and Cocks 1991). The second approach includes Ecological Niche Factor Analysis (ENFA, implemented in the software BioMapper; Hirzel et al. 2002), Maximum Entropy (implemented in the software MAXENT; Phillips et al. 2006) and Principal Components Analysis (PCA, implemented in R and BioMapper; Legendre and Legendre 1998, Hirzel et al. 2002). Techniques such as GARP are often regarded as PO approaches; however, these were considered in PA approaches as they require the generation of pseudo-absences and so technically use a form of PA data.

There has been considerable debate among researchers and scientists, regarding the authenticity of both of these approaches (Brotons et al. 2004, Hirzel et al. 2001). and numerous studies tried to compare the results of multiple techniques applied to similar datasets. GLM and ENFA was compared using data from a ‘virtual’ species (Hirzel et al. 2001) and real data collected in the field (Brotons et al. 2004, Elith et al. 2006). PA approaches performed better than PO approaches where absence data are reliable (Brotons et al. 2004), however, as it is stated above, where absence data are unreliable PA approaches are at risk of incorporating ‘false absences’, which are likely to introduce considerable bias (Gu and Swihart 2004). For poorly known or cryptic species, unreliable absence data is a common occurrence in ecological studies (Hirzel et al. 2002), or species not at equilibrium with their environment (i.e. not occupying all suitable areas). On the other hand sources of PO data are widespread and comprise

of atlases, museum and herbarium records, incidental observation databases, radio-tracking studies and species lists (Pearce and Boyce 2006). Where the focal species is rare or highly mobile PO approach is most likely to follow. Therefore, PO approaches is more useful in a range of situations in which PA approaches are not applicable.

5.6.4 Applying SDMs: The importance of scale

SDM studies vary from continental or global scale when macroecological processes are being considered (Araújo and New 2007); or regional scale when detailed ecological insight is required (Ferrier et al. 2002). In studying the response of individuals to temporal and spatial resource heterogeneity, local extent and fine resolutions are used, while range shifts over larger geographical areas, assessing population distribution along broader environmental gradients, larger extents and coarser resolutions are useful (Wiens 2002, Guisan and Thuiller 2005). Many studies have stressed on species-habitat relationships at multiple scales (Mackey and Lindenmayer 2001, Whittingham et al. 2005, Qi et al. 2012). Analyzed in a hierarchical fashion; these studies established the effects of climate determining distribution at a global or national scale (largest extent, coarsest grain), however at scales of a few to hundreds of kilometers, topography and substrate type are paid attention (Mackey and Lindenmayer 2001) and unique habitat features are often quantified at scales of tens to hundreds to meters (Whittingham et al. 2005). In many cases, the grain (resolution) of the EGV and species distribution data matches, but depending on the type of collection it may differ. In most of the cases the resolution of the species distribution data and behavior of the species regulates which EGV data to be used; EGV data may then be aggregated (i.e. averaged to a larger grid cell size) to match the resolution of the species distribution data or defined at finer resolutions than they were collected in order to achieve consistency across all EGV layers collected in different ways with different resolutions. It is also important to consider EGVs and collected species locations to be matched at temporal scale, using data collected from the same time period where possible (van Beest et al. 2012). Combined with data collected at an appropriate spatial scale, this ensures that the species-habitat relationship is quantified as accurately as possible.

5.6.5 Applying SDMs: The importance of appropriate EGVs

In spite of its wide use, SDM faces criticism from different authors regarding the use of many variables which are thought to be irrelevant. Due to their easy availability and with the advances in sensory and GIS technologies, together with the extensive use of the internet, it is easy to find or distribute freely available data; therefore they are incorporated in developing model. In addition the measurement, acquisition and integration of these is much easier and cheaper, thus inclusion of these additional variables has become a practice, though many argue it as a bad practice and suggest use of sound ecological knowledge of the focal species in the choice of EGVs. They recommend the choice of only those variables which are likely to be ecologically relevant (Elith and Leathwick 2009).

However, it is not only the inclusion of these variables in initial models that has been criticized but also the procedures used to eliminate extraneous variables from the final model. As a general practice, all available data are included in the model and then analytical procedures are used to determine which are 'important' to describing the species distribution (Stauffer 2002). Additionally, in multivariate regression models variables are selected in a stepwise fashion, in a procedure that relies on statistical significance alone for otherwise seemingly 'arbitrary' inclusion of variables in the final model. The order of variable input might also affects the composition of the final model. Referred as 'statistical tinkering' (Mac Nally 2000) such stepwise methods are more widely criticized (Whittingham et al. 2006) and less in use. New information theory-based procedures such as Akaike's Information Criterion (AIC; Akaike 1974, Burnham and Anderson 2002, Richards 2008) avoid many of these criticisms as this approach allows variable selection to be based on more sound ecological knowledge of the focal species and their functional relationships with the environment. The final model gets selected based on the minimum AIC value, or weighting models based on the AIC value where there is some uncertainty in choosing between the final few model options (Burnham and Anderson 2002). It is accepted that fewer (and coarser scaled) variables give greater generality, whilst adding (finer resolution) EGVs will likely provide a model with greater predictive accuracy to the focal population, though the addition of extraneous EGVs, or a decrease in EGV resolution, may lead to different outcomes (Lowe et al. 2010, van Beest et al. 2012). The sampling of resource availability should therefore focus on ecologically relevant EGVs, matched at the appropriate scale.

5.7. Software

Here I used Maximum Entropy Modeling (MaxEnt) for species distribution modeling. Based on the principle of maximum entropy (Jaynes 1957), Maxent was developed for species distribution modelling by Phillips et al. (2004, 2006). MaxEnt estimates probability distribution by finding the probability distribution of maximum entropy (i.e, closest to uniform) (Phillips et al. 2006). The algorithm was preferred for use in this study because it (1) was considered best when compared with other novel methods (Elith et al. 2006, Gibson et al. 2007, Pearson et al. 2007, Hernandez et al. 2008), (2) does not need absence data, and (3) permits for the incorporation of categorical information (that is, landcover). All analyses were conducted using Maxent version 3.2.1, available at <http://www.cs.princeton.edu/~schapire/maxent/> (Phillips et al. 2004, 2006).

For validation of model performance, 100 replicates were chosen by randomly selecting a proportion of 75% occurrence points, keeping apart the remaining 25% as validation points in each replicate for a completely independent test of predictive accuracy. Different features (Linear, Linear Quadratic, Product, and Threshold) available in the MaxEnt software, as algorithms were used and the best AUC value was used to decide the final model out of all these. Percent of influence of each variables used in the model was presented in table format and Jackknife regularized training gain for the species was used to understand the effect of each variable in model building.

I used MaxEnt with the “auto-features” mode (Phillips and Dudik 2008). By default, MaxEnt chose uniformly and at random 10,000 background samples of pseudo absences from the study area and used them in place of absences during modeling to illustrate the environmental conditions in the region (Phillips et al. 2006, 2009). The logistic output format was selected because of its robustness and easier to interpret (Phillips and Dudik 2008). Grid cells with a small logistic value were predicted to be unsuitable or only marginally suitable for the species.

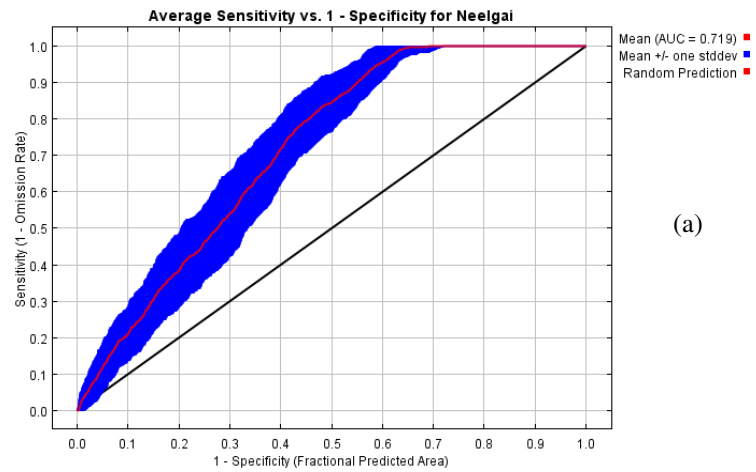
Receiver Operating Characteristic (ROC) curves (Phillips et al. 2004) was used assessing model performance. It is useful as the area under the ROC curve (AUC) offers a single measure of model performance, independent of any choice of threshold (Phillips et al. 2006). For each run, we calculated the AUC, the best classifier thus has

an AUC of 1, although the maximum AUC is less than one because of the use of presence only data (Wiley et al. 2003, Phillips et al. 2004). Usually, AUC values greater than 0.7 are considered to be potentially significant, while scores of 0.5 indicate a predictive discrimination that is no better than random (Elith et al. 2006).

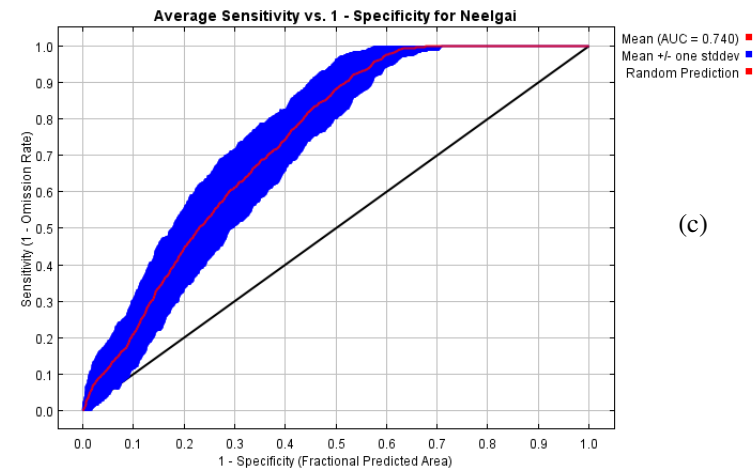
5.8. Results

Nilgai species distribution modeling

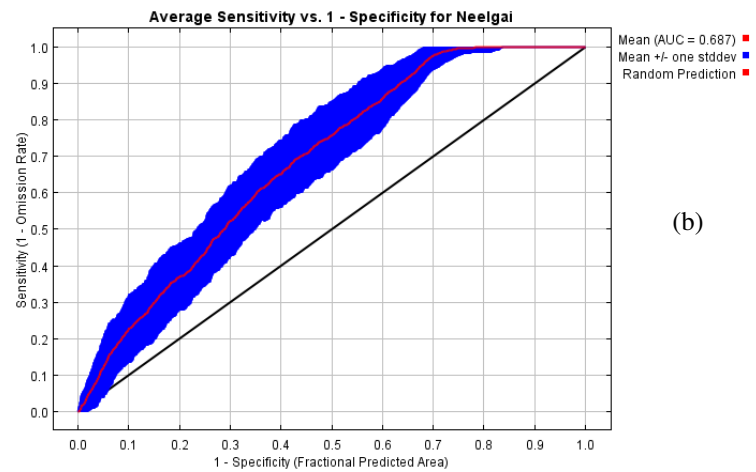
AUC values for models at all spatial scales were > 0.70 , implying a good model fit (Figure 5.8). Visualizations for each model are provided in Figure 5.9 and 5.10. Areas of high habitat suitability in each of the models (considered being areas with $> 50\%$ probability of presence), with the linear and linear quadratic feature type models indicated the largest area of suitable habitat. This was an arbitrary threshold to estimate the area of good habitat and provided a means of comparison between models. Eleven of the variables were common to each model, described below. However, contribution of each variable changed from one to another spatial scale and species requirement (Table 5.2). At larger than home range scale (5KmX5km), SDM was mostly contributed by euclidean distance to waterholes (42%), area of dense forest patches (39.2%), along with euclidean distance to Ness (11.8%). Open forest patches and digital elevation model also proved to be important variables in the prediction but with smaller individual contributions (5.3% and 1.7%). At home range scale (2.5KmX2.5Km), the most important explanatory variable was area of dense forest patches (50.6%), with area of open forest patches (15.6%), euclidean distance to ness (12%) altogether contributing 78.2% to the model output. Pre and post monsoon NDVI also proved to be important variables in the prediction but with smaller individual contribution (6.2% and 6%). At (1KmX1Km) scale, dense forest patches continued to be the most significant variable with contribution 52.6%, along with open forest patches (21.9%). Other important variables in the prediction were euclidean distance to ness (7.6%), Premonsoon NDVI (5.9%), Post monsoon NDVI and digital elevation model (3.6%). At local scale (500mX500m), the most important explanatory variable was area of dense forest patches (70%) along with euclidean distance to ness (7.8%), post-monsoon NDVI (4.7%), pre-monsoon NDVI (4.1%). Few other variables contributed very small amount of information, they are digital elevation model (2.9%), precipitation seasonality (2.9%), euclidean distance to road (2.7%), euclidean distance to drainage (2.4%), and euclidean distance to waterholes (1.6%).



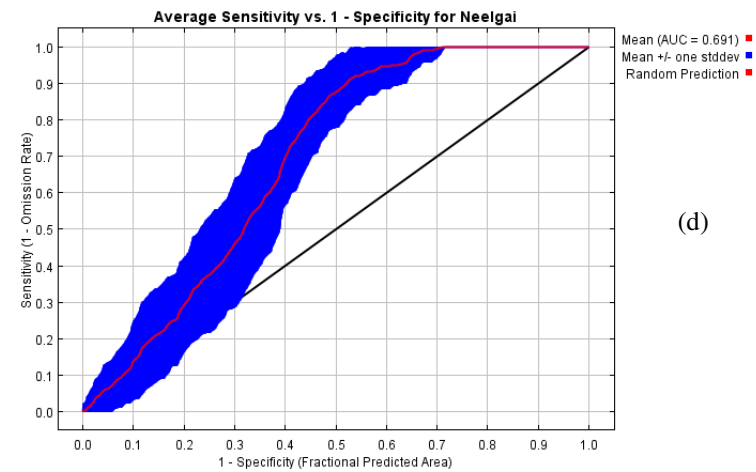
(a)



(c)



(b)



(d)

Figure 5.8: Receiver operating characteristic (ROC) curve for SDM of Nilgai in different spatial resolutions (a) at 500m X 500m scale (b) 1Km X 1Km scale, (c) 2.5Km X 2.5Km scale and (d) 5 Km X 5 Km scale

Table 5.2: Contribution of different variables in species distribution modeling of nilgai at different spatial scales

Spatial Scale	500 meter		1Km		2.5Km		5Km	
Variable	Percent contribution	Permutation importance	Percent contribution	Permutation importance	Percent contribution	Permutation importance	Percent contribution	Permutation importance
Dense Forest Patches	70	58.9	52.6	54.4	50.6	46.5	39.2	40.5
Open Forest Patches	0.9	3.2	21.9	15.3	15.6	17.8	5.3	4.2
Euclidean Distance to Ness	7.8	11.7	7.6	4.8	12	14	11.8	14.2
Premonsoon NDVI	4.1	2.9	5.9	9.7	6.2	2.7	-	-
Post monsoon NDVI	4.7	8.1	3.6	8.1	6	9.8	-	-
Euclidean Distance to Waterholes	1.6	3.9	1.5	2.3	5.3	4.7	42	35
Euclidean Distance to Road	2.7	2.7	-	-	2.2	0.8	-	-
Annual Mean Temperature	-	-	-	-	0.9	2.1	-	-
Precipitation Seasonality	2.9	3.7	0.7	0.6	0.7	1	-	-
Euclidean Distance to Drainage	2.4	1.1	2.2	3.6	0.6	0.6	-	-
Digital Elevation Model	2.9	3.8	3.6	1	-	-	1.7	6

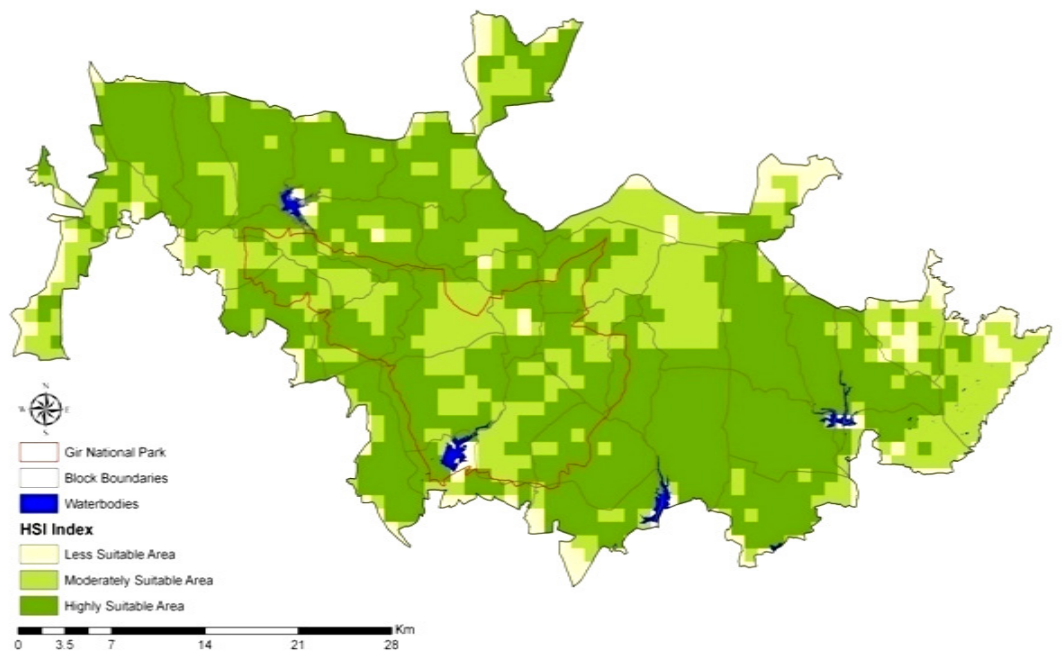
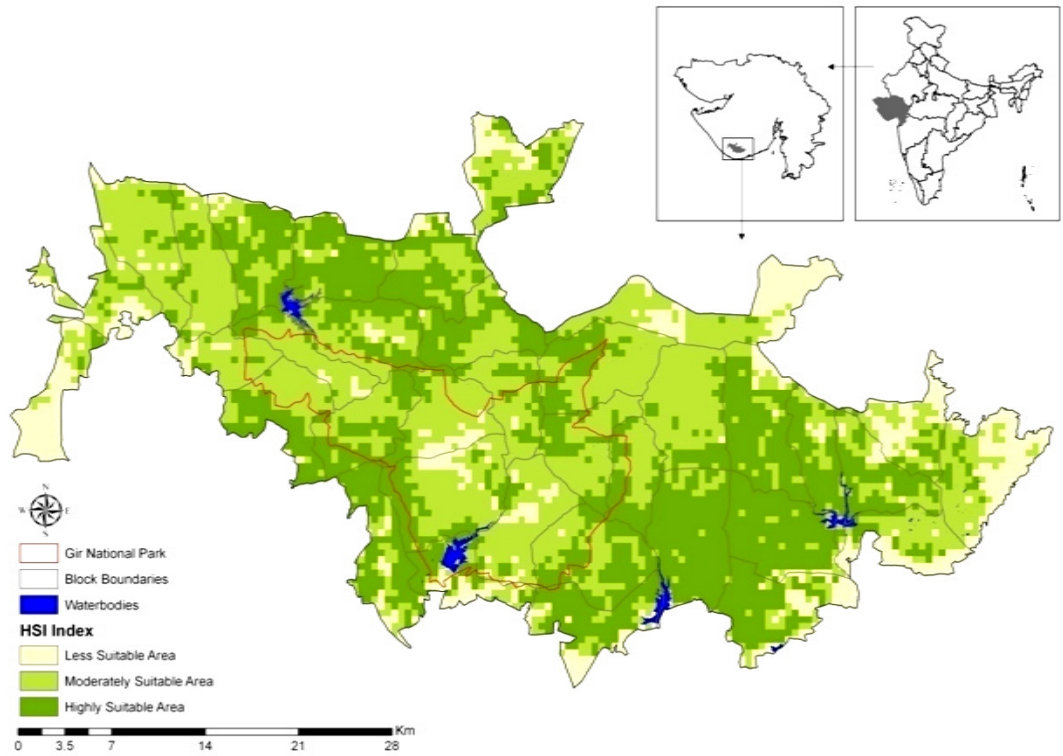


Figure 5.9: Species distribution model of Nilgai at 500mX500m, 1KmX1Km resolution

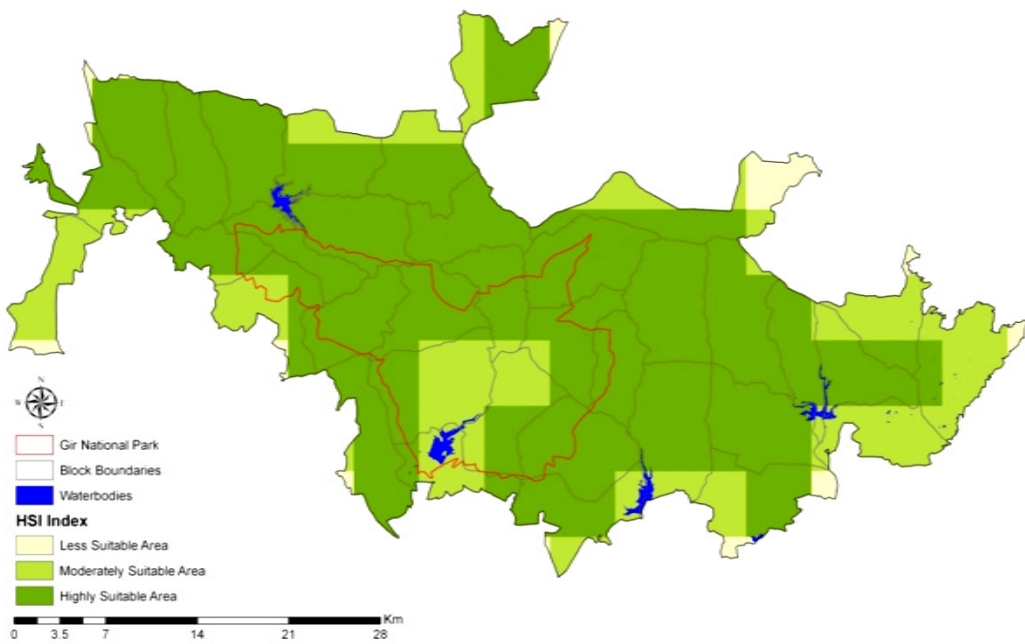
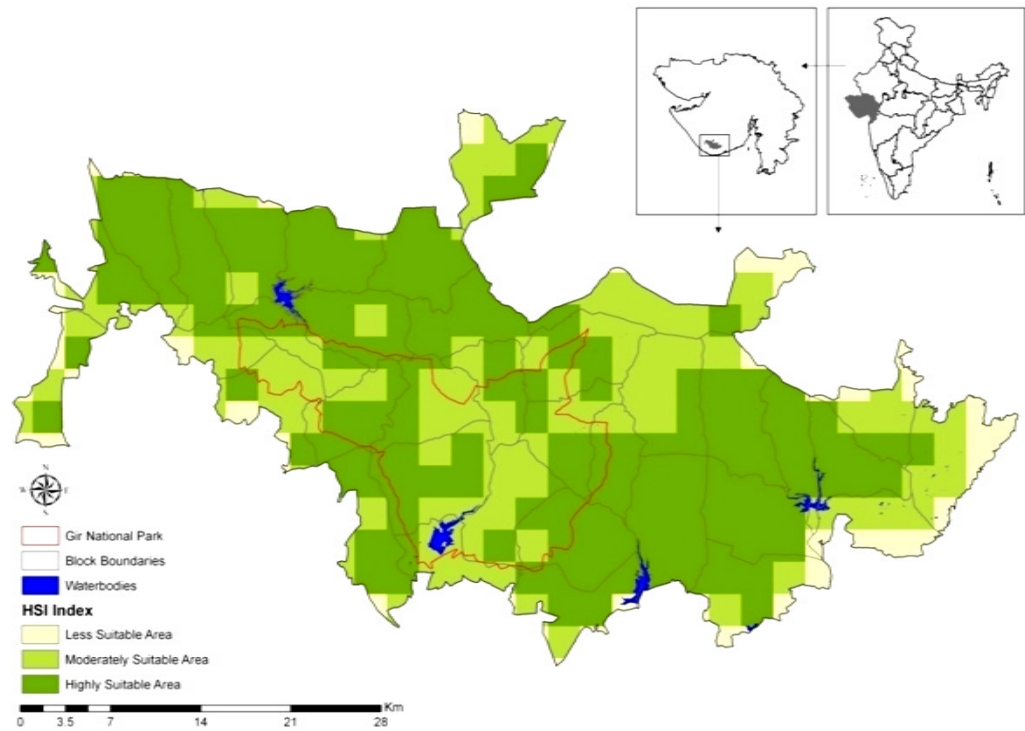
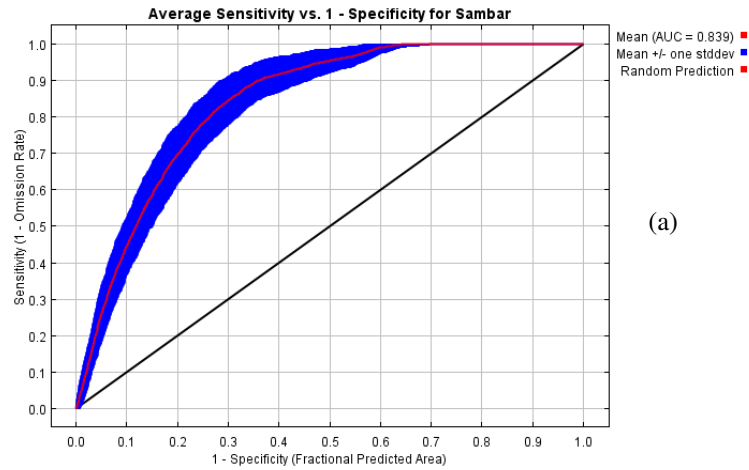


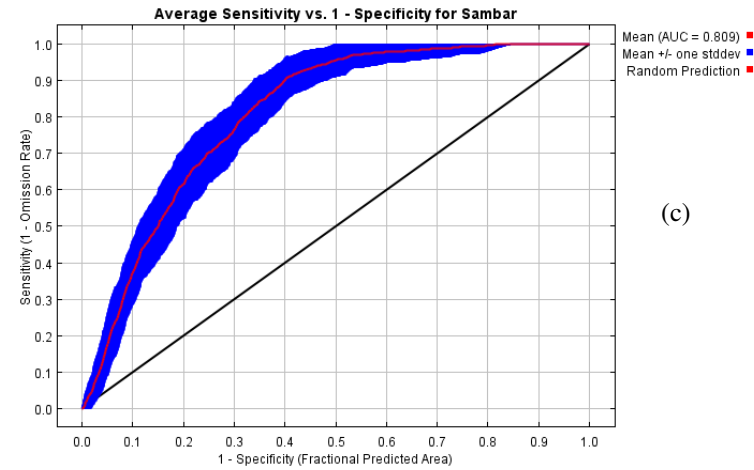
Figure 5.10: Species distribution model of Nilgai at 2.5kmX2.5K, 5KmX5Km resolution

Sambar species distribution modeling

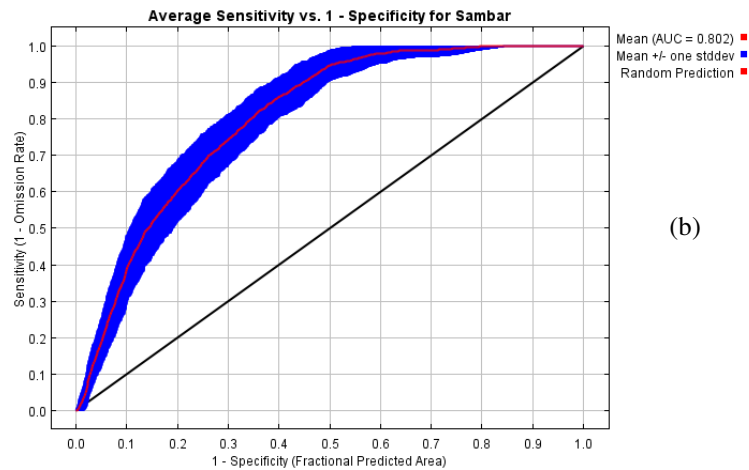
AUC values for all models were > 0.80 inferring good model fit (Figure 5.11). Visualizations for each model are provided in Figure 5.12 and 5.13. Areas of high habitat suitability in each of the models (considered being areas with $> 50\%$ probability of presence); with the linear and linear quadratic feature type models indicated the largest area of suitable habitat. This was an arbitrary threshold to estimate the area of good habitat and provided a means of comparison between models. Eleven variables were common to each model, though the percent of contribution varied from one scale to other (Table 5.3). At larger than home range scale (5KmX5km), SDM was mostly contributed by dense forest patches (92.5%) with very little contribution from euclidean distance to waterholes (6.9%). At home range scale (2.5KmX2.5Km), the most important explanatory variable was area of dense forest patches (84.3%), along with smaller individual contributions from other variables, precipitation seasonality and euclidean distance to ness (3.8%), open forest patches (2.6%), euclidean distance to waterholes (2.1%), pre-monsoon NDVI (1.3%), euclidean distance to road (1.2%). Annual mean temperature and euclidean distance to drainage also proved to be important variables in the prediction but with smaller individual contribution. At (1KmX1Km) scale, dense forest patches continued to be the most significant factor with contribution of 67.3% along with euclidean distance to nesses (13.8%). Other important factors in the prediction were precipitation seasonality (7.9%), open forest patches (5.4%), and euclidean distance to waterholes (2.5%). Post-monsoon NDVI, pre-monsoon NDVI, euclidean distance to drainage, digital elevation model also proved to be important variables in the prediction but with smaller individual contribution. At local scale (500mX500m), the most important explanatory variables were dense forest patches (57.6%), euclidean distance to ness (20.5%), and precipitation seasonality (9.9%). Other important factors were post-monsoon NDVI (4.3%), digital elevation model (2.7%), pre-monsoon NDVI (1.7%), euclidean distance to roads (1.5%). Euclidean distance to drainage, waterholes, and presence of open forest patches also proved to be important variables in the prediction but with smaller individual contribution.



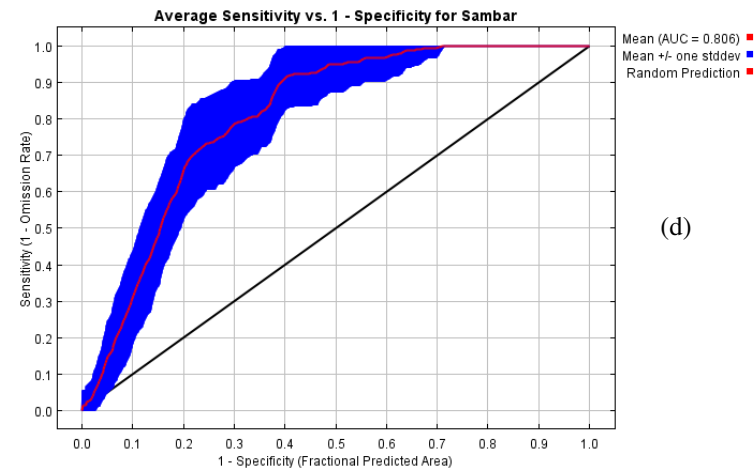
(a)



(c)



(b)



(d)

Figure 5.11: Receiver operating characteristic (ROC) curve for SDM of Sambar in different spatial resolutions (a) at 500m X 500m scale (b) 1Km X 1Km scale, (c) 2.5Km X 2.5Km scale and (d) 5 Km X 5 Km scale

Table 5.3: Contribution of different variables in species distribution modeling of sambar at different spatial scales

Spatial Scale	500m		1Km		2.5km		5Km	
Variables	Percent contribution	Permutation importance	Percent contribution	Permutation importance	Percent contribution	Permutation importance	Percent contribution	Permutation importance
Dense Forest Patches	57.6	56.9	67.3	60.2	84.3	81	92.5	87.6
Precipitation Seasonality	9.9	10.4	7.9	6.1	3.8	9	-	-
Euclidean Distance to Ness	20.5	17.5	13.8	20.7	3.8	2.2	-	-
Open Forest Patches	0.4	1.7	5.4	7.8	2.6	1.4	-	-
Euclidean Distance to Waterholes	0.6	1	2.5	1.3	2.1	2.2	6.9	11.7
Premonsoon NDVI	1.7	1.3	0.8	0.9	1.3	1.6	-	-
Euclidean Distance to Road	1.5	2.5			1.2	1.2	-	-
Annual Mean Temperature	-	-			0.4	0.2	-	-
Euclidean Distance to Drainage	0.8	0.7	0.5	0.4	0.3	1.1	-	-
Post monsoon NDVI	4.3	4.4	1.7	2.4	-	-	-	-
Digital Elevation Model	2.7	3.5	0.1	0.3	-	-	-	-

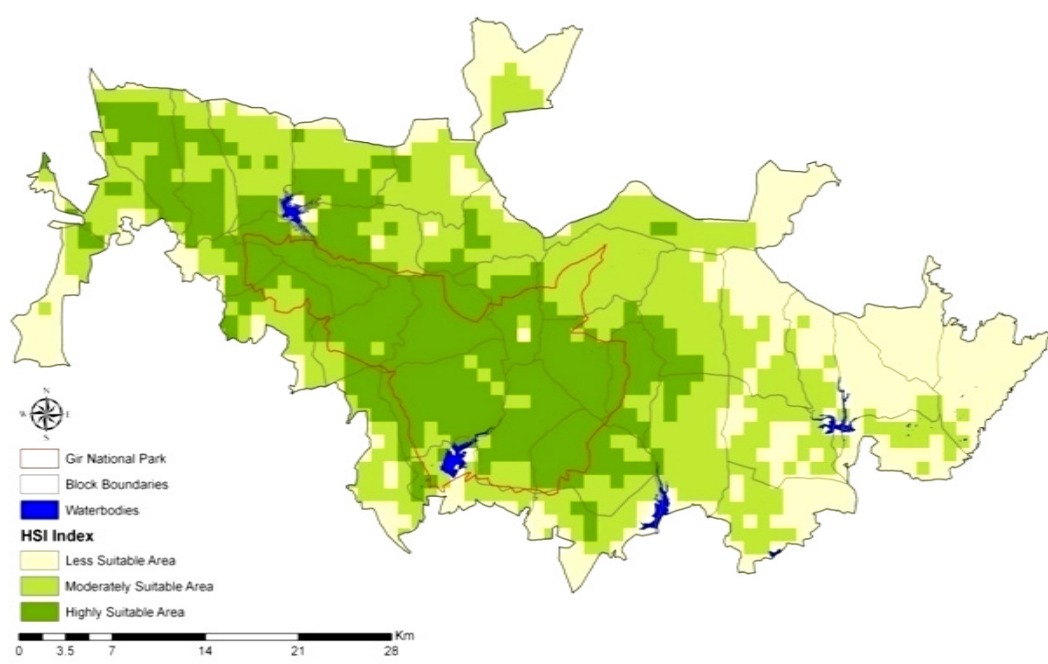
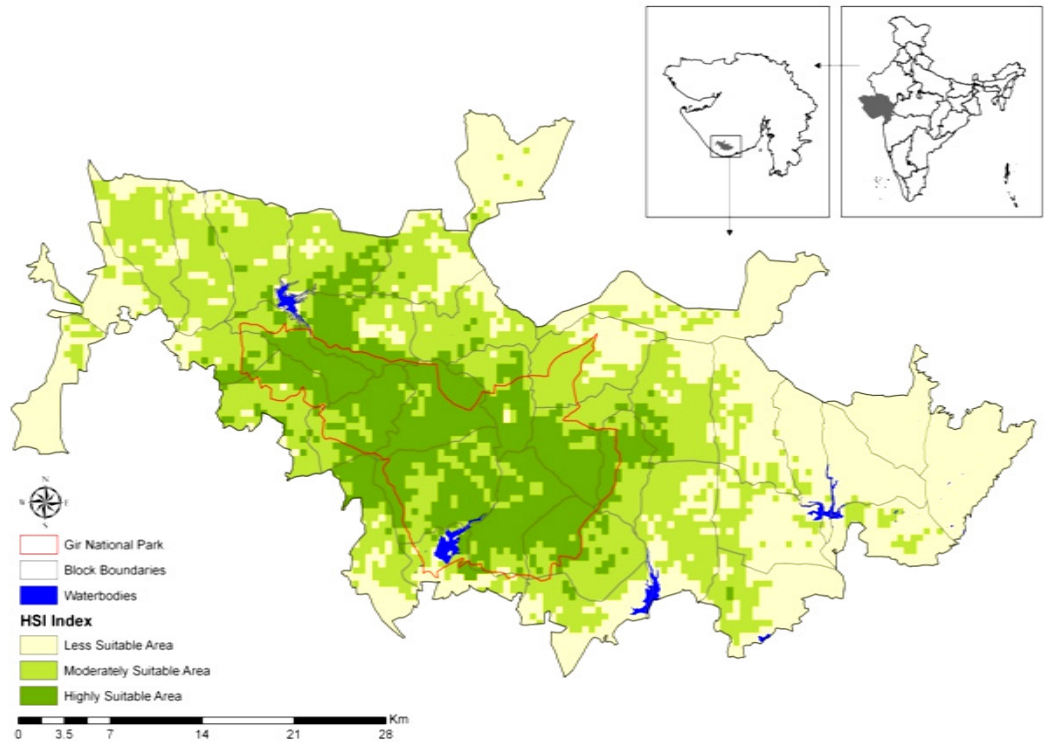


Figure 5.12: species distribution modeling of Sambar at 500mX500m, 1KmX1Km resolution

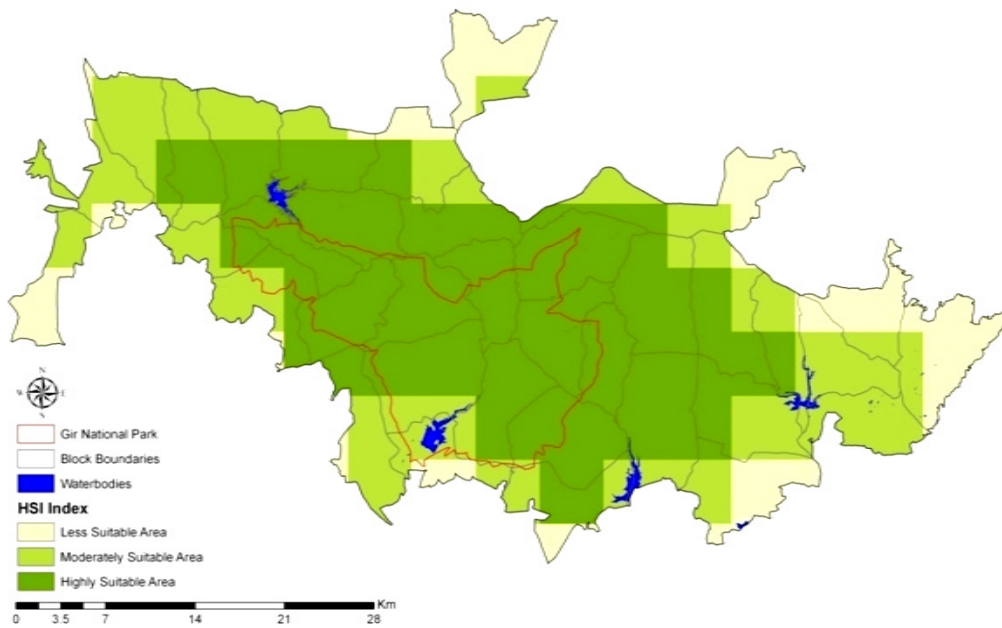
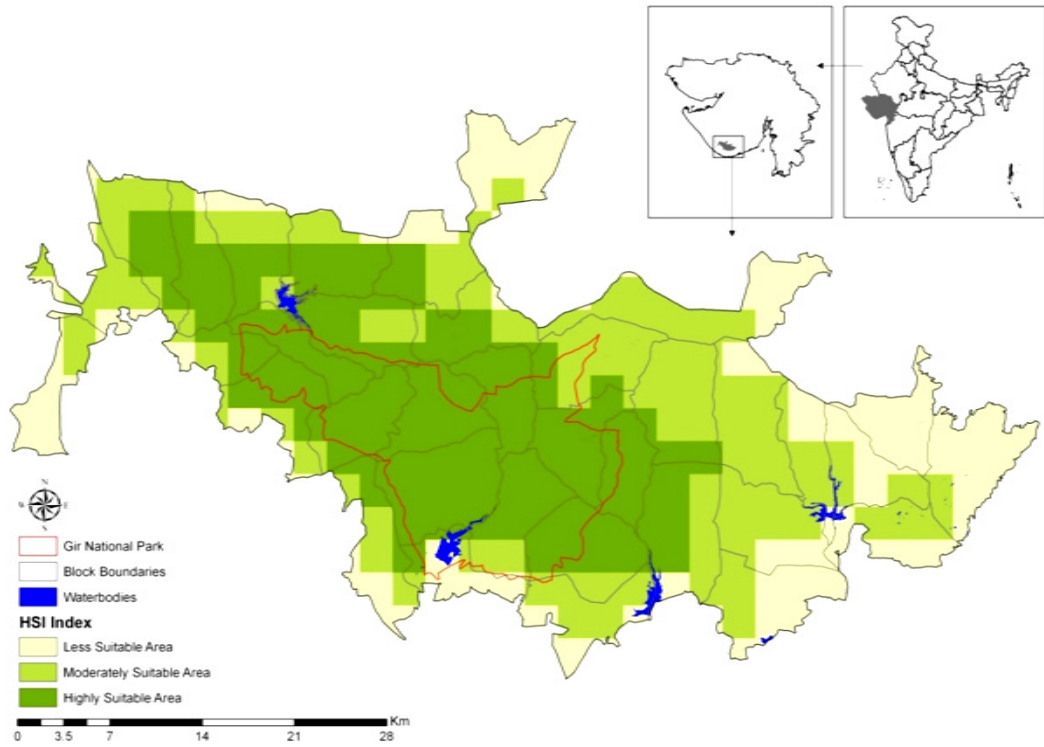
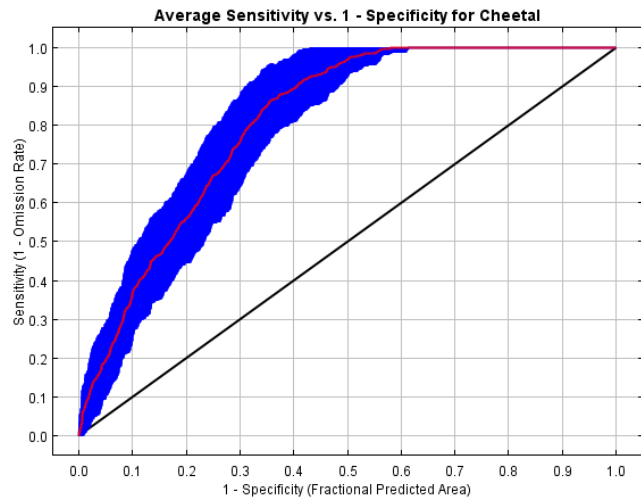


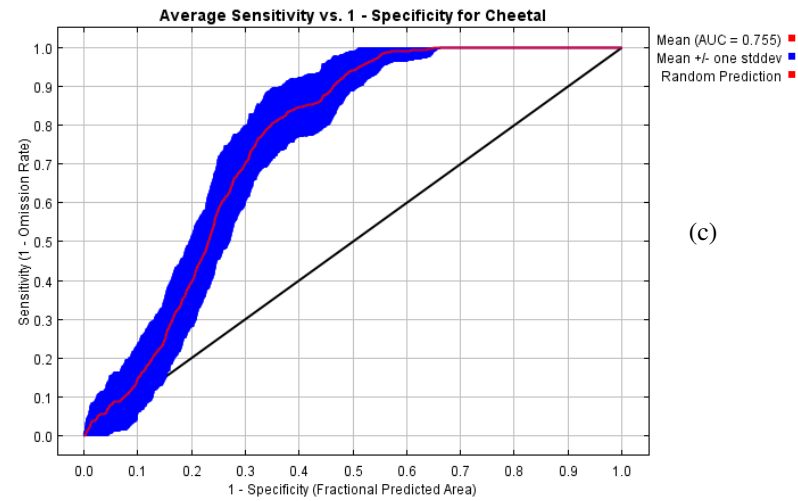
Figure 5.13: Species distribution modeling of Sambar at 2.5KmX2.5Km, 5KmX5Km resolution

Chital species distribution modeling

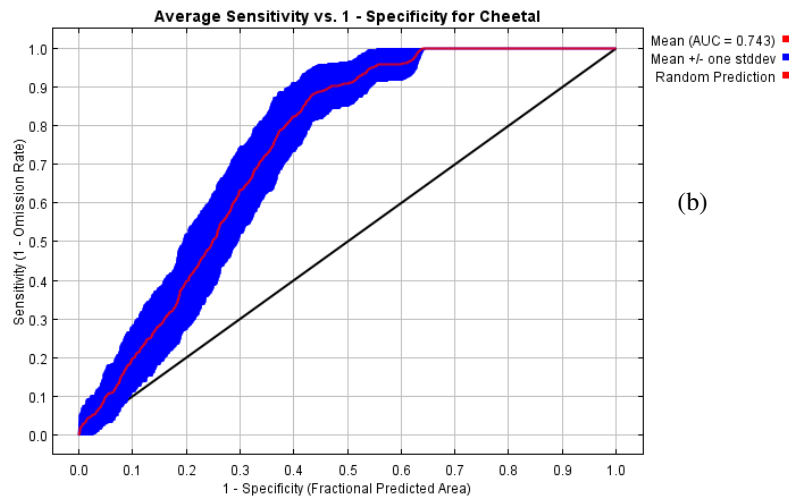
AUC values for all models were > 0.70 implying potentially significant result (Figure 5.14). Visualizations for each model are provided in Figure 5.15 and 5.16. Areas of high habitat suitability in each of the models (considered being areas with $> 50\%$ probability of presence), with the linear and linear quadratic feature type models indicated the largest area of suitable habitat. Nine variables were found common in all spatial scales (Table 5.4). At larger than home range scale (5KmX5km), species distribution model (SDM) was mostly contributed by dense forest patches (86.2%), along with other variables, euclidean distance to nesses (9.9%), euclidean distance to waterholes (2.1%) and euclidean distance to drainage (1.8%). At home range scale (2.5KmX2.5Km), the most important explanatory variable was area of dense forest patches (89.5%). Presence of open forest patches, euclidean distance to nesses, euclidean distance to artificial waterholes, pre and post monsoon NDVI, and euclidean distance to road also proved to be important variables in the prediction but with smaller individual contribution. At (1KmX1Km) scale, dense forest patches continued to be the most significant factor with contribution of 86.8% along with other variables such as euclidean distance to nesses (5.8%), open forest patches (4.8%) and pre-monsoon NDVI (2.6%). At local scale (500m X500m), the most important explanatory variables were presence of dense forest patches (74.7%), euclidean distance to waterholes (14.9%). Post-monsoon NDVI (4.2%), pre-monsoon NDVI (3.7%), digital elevation model (1.5%), open forest patches (0.8%) also proved to be important variables in the prediction but with smaller individual contribution.



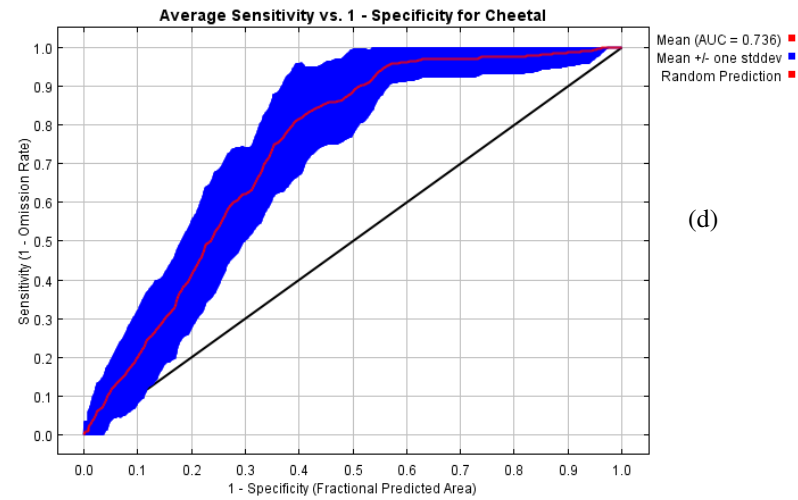
(a)



(c)



(b)



(d)

Figure 5.14: Receiver operating characteristic (ROC) curve for SDM of Cheetal in different spatial resolutions (a) at 500m X 500m scale (b) 1Km X 1Km scale, (c) 2.5Km X 2.5Km scale and (d) 5 Km X 5 Km scale

Table 5.4: Contribution of different variables in species distribution modeling of cheetal at different spatial scales

Spatial Scale	500meter		1Km		2.5Km		5Km	
Variables	Percent contribution	Permutation importance	Percent contribution	Permutation importance	Percent contribution	Permutation importance	Percent contribution	Permutation importance
Dense Forest Patches	74.7	59.7	86.8	88.7	89.5	87	86.2	91.8
Open Forest Patches	0.8	11.2	4.8	1	4.4	2.8	-	-
Euclidean Distance to Ness	-	-	5.8	6.9	3	2.8	9.9	4.7
Euclidean Distance to Waterholes	14.9	16.2	-	-	1.4	5.8	2.1	2.3
Post monsoon NDVI	4.2	4.6	-	-	0.6	0.1	-	-
Euclidean Distance to Road	-	-	-	-	0.6	0.9	-	-
Premonsoon NDVI	3.7	3.9	2.6	3.5	0.4	0.6	-	-
Digital Elevation Model	1.5	3.5	-	-	-	-	-	-
Euclidean Distance to Drainage	-	-	-	-	-	-	1.8	1.2

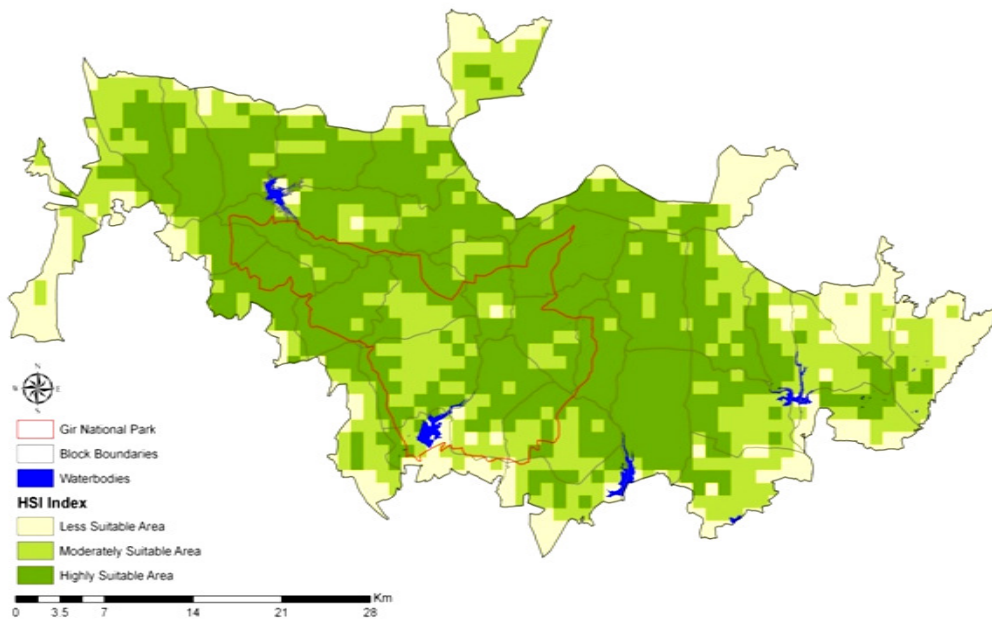
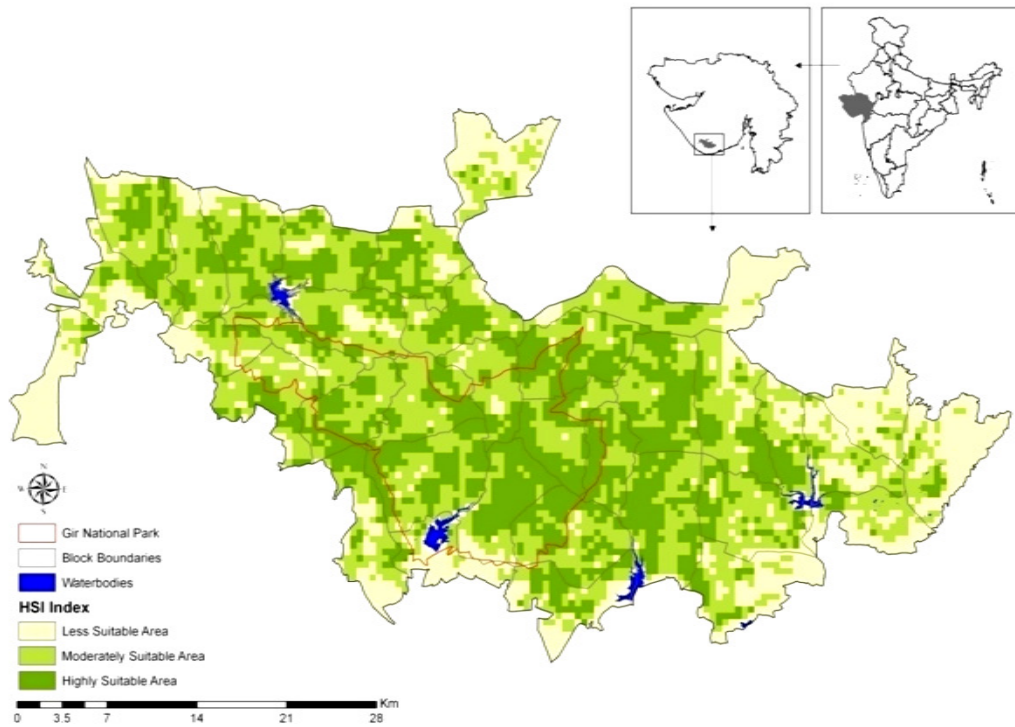


Figure 5.15: Species distribution modeling of Cheetal at 500mX500m, 1KmX1Km spatial resolution

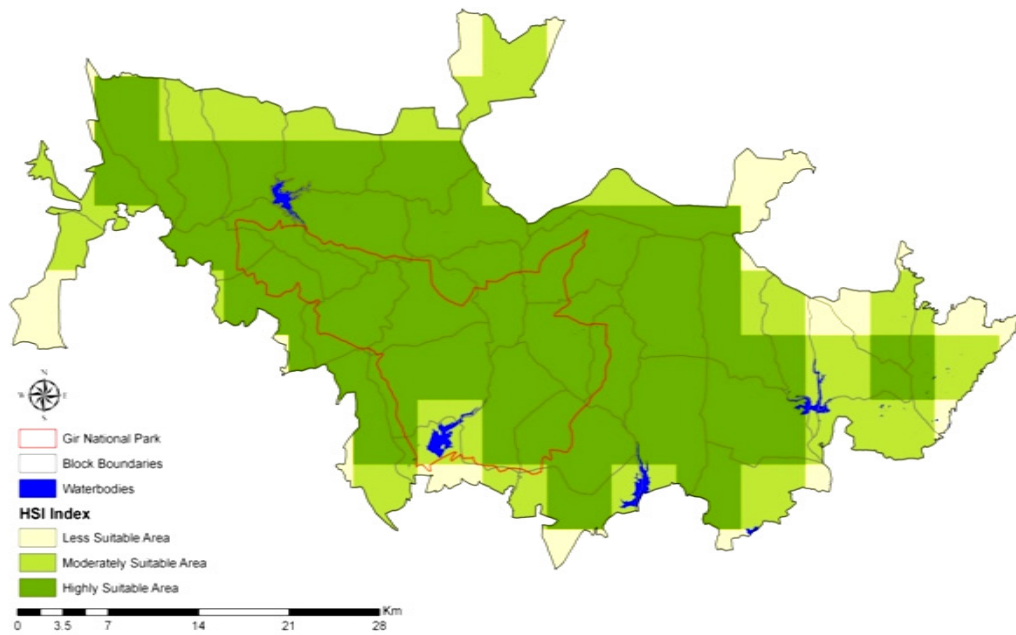
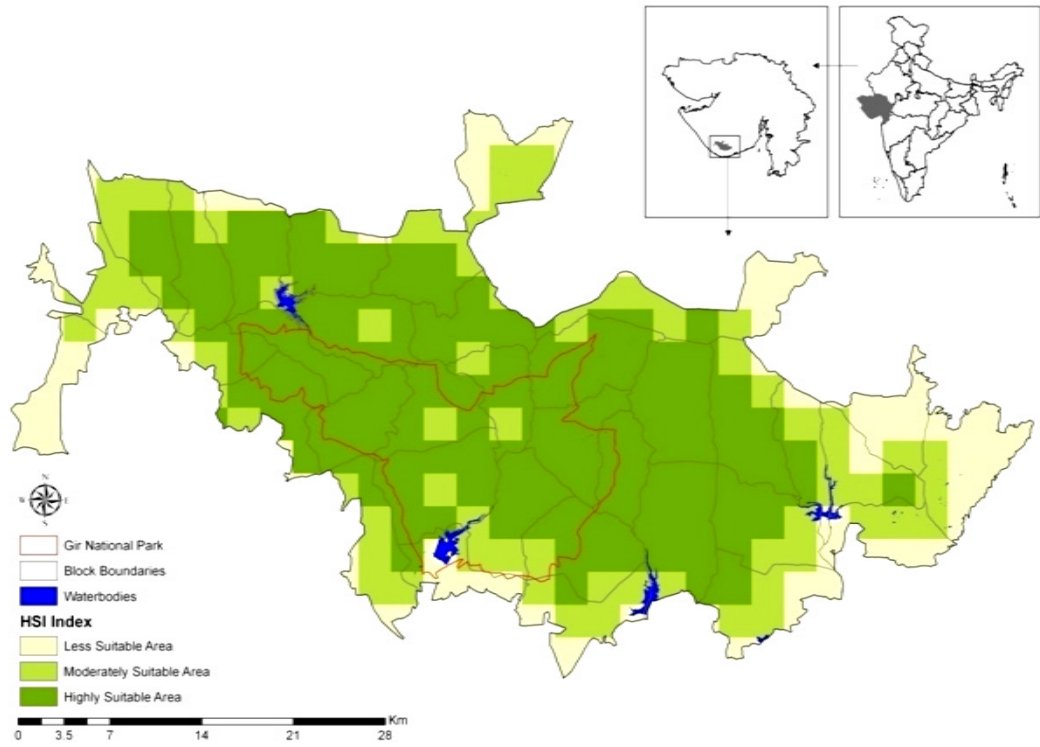


Figure 5.16: Species distribution modeling of Cheetal at 2.5KmX2.5Km, 5KmX5Km spatial resolution

Lion species distribution modeling:

The eastern Gir portion of the protected area with very few lions earlier showed a remarkable change. In an area, half the size of western portion it increased to 106 by 2001 (Divyabhanusinh 2005). It is believed that after the relocation of *maldharis* and resident livestock, the domestic ungulate population in east Gir has increased giving rise to lion population than that of west Gir. In comparison to teak dominated dense forest types of west Gir east Gir still supports thickets of dense forests along rivers and stream, and thorny and savannah type forests. With the effective protection measures coupled with conservation works, the availability of food and water for the lion has improved, especially in east Gir. In addition peripheral villages in west Gir have intensive cultivation and high human population, whereas in eastern Gir, open areas and hills with open thorny forests, facilitate the lion's movement outside the sanctuary, radiating upto Bhavnagar district (Singh 2007). The lions do not prefer the hills unless it is forced to occupy them (Anon. 1975). Rivers and riverside forests within Gir protected area are vital for wildlife survival. Based on the semi arid nature of the landscape, in late winter and in summer, the wildlife population is found concentrated along these rivers.

The lowest modeling scale (500mX500m) resembling lion's local range had eight important variables, of which two were topography and precipitation seasonality. Topography influences vegetation mosaic and grazing ecology because different plant communities occupy different topographic positions. In 1KmX1Km scale, there were seven variables, of which, area of dense forest patches contributed most (70.8%). The other variables contributing to the model output were, digital elevation model (9.8%), euclidean distance to drainage (5.6%), area of open forest patches (4.2%), euclidean distance to ness (3.3%), euclidean distance to waterholes (3.3%), and post-monsoon NDVI (3%) (Table 5.5). The influence of topographic/edaphic gradients and the distances resident herbivores travel to drink water can restrict their spatial distribution in the dry season. The distribution of large predators sometimes get affected by the patchy distribution of resident herbivores or must be overridden by the enormous herds of livestock in the dry season. In the wet season, the patchiness in the distribution of drinking water and food availability subsides, allowing resident herbivores to range more widely but topographic/edaphic gradients, acting through the medium of the degree of drainage, can still restrict the distribution of those

herbivores that avoid muddy or sticky soils, prevalent during the wet season. In addition, vegetation structure influences both lion prey availability and hunting success, the cost of territory maintenance and cub defense (Figure 5.18 and 5.19).

In the medium spatial scale (2.5kmX2.5km) nine important variables were found affecting the model output, but open forest patches contributed more to the model building at this scale than other scales and also distance to artificial water holes was a significant contributing factor to this scale. In other studies it was observed that in areas with high density of suitable prey and good hunting conditions, lions move less, than in areas with low density of suitable prey and poor hunting conditions (Ogutu and Dublin 2002, Hemson 2003). Seasonality in prey and water availability and vegetation cover can influence the stability of lion territories and the group sizes they support (MacDonald 1983), estrous in lionesses, the frequency of sexual activity, and hence birth peaks (Schaller 1972: pp. 182-3).

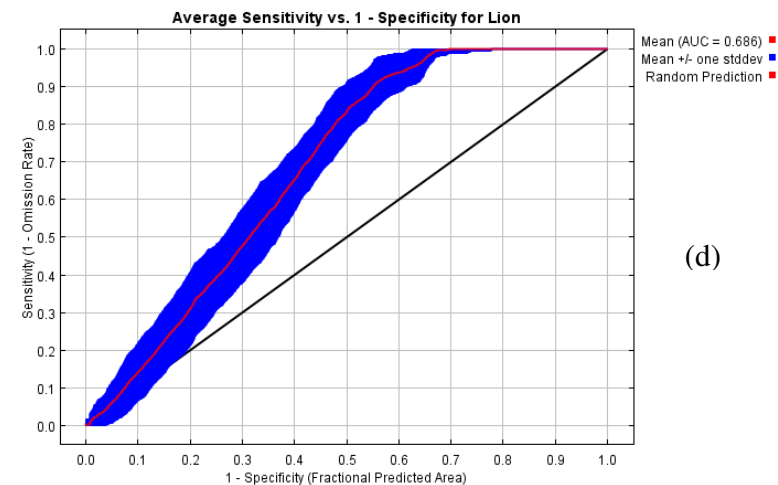
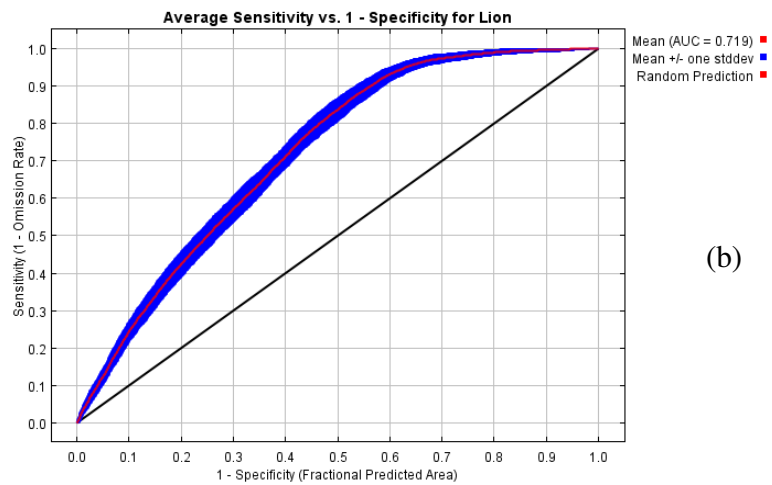
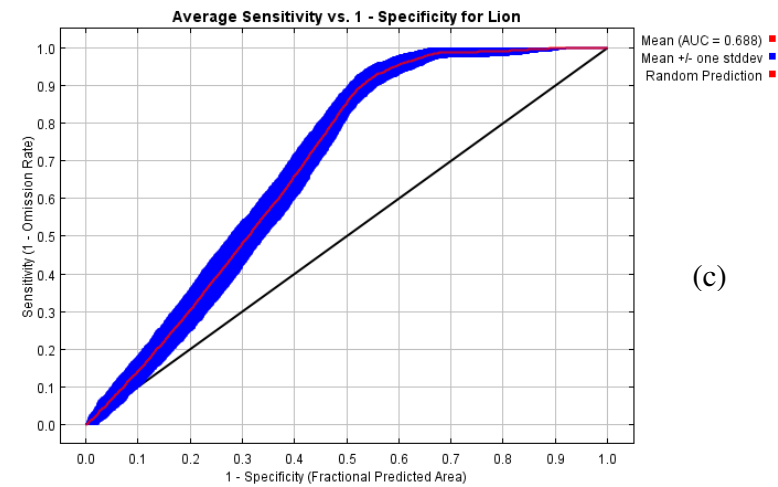
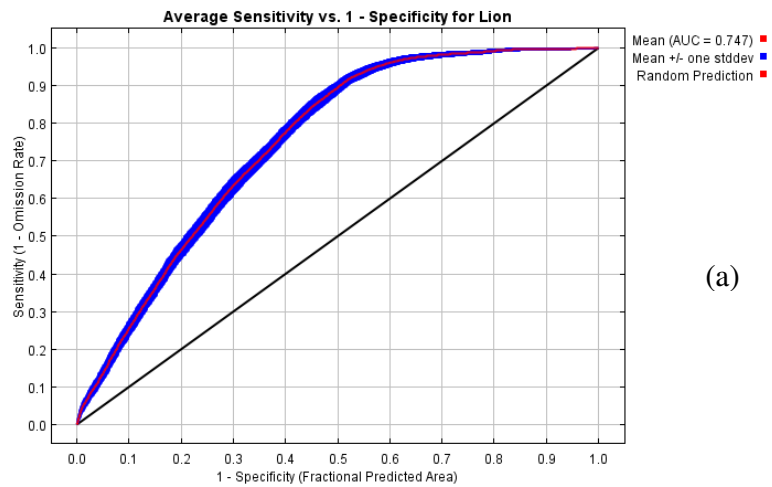


Figure 5.17: Receiver operating characteristic (ROC) curve for SDM of Lion in different spatial resolutions (a) at 500m X 500m scale (b) 1Km X 1Km scale, (c) 2.5Km X 2.5Km scale and (d) 5 Km X 5 Km scale

Table 5.5: Contribution of different variables in species distribution modeling of lions at different spatial scales

Variables	500mX500m		1kmX1km		2.5kmX2.5km		5kmX5km	
	Percent contribution	Permutation importance	Percent contribution	Permutation importance	Percent contribution	Permutation importance	Percent contribution	Permutation importance
Dense Forest Patches	72.7	64.1	70.8	61.5	68.1	62.5	50.9	65.4
Digital Elevation Model	10.4	17.2	9.8	18.2	-	-	-	-
Euclidean Distance to Ness	6.1	4	3.3	1.7	1.6	3.1	6.7	1.1
Euclidean Distance to Drainage	4.2	8.9	5.6	9.3	2.9	8.4	0.7	2.2
Euclidean Distance to Waterholes	3.4	1.4	3.3	1.1	3.2	7.1	19.9	3
Post monsoon NDVI	1.8	2.1	3	1.2	3.2	2.8	-	-
Precipitation Seasonality	1.1	2	-	-	0.2	1	-	-
Premonsoon NDVI	0.3	0.3	-	-	1	1.9	-	-
Open Forest Patches	-	-	4.2	6.9	19.1	9.5	21.8	28.3
Euclidean Distance to Road	-	-	-	-	0.6	3.8	-	-

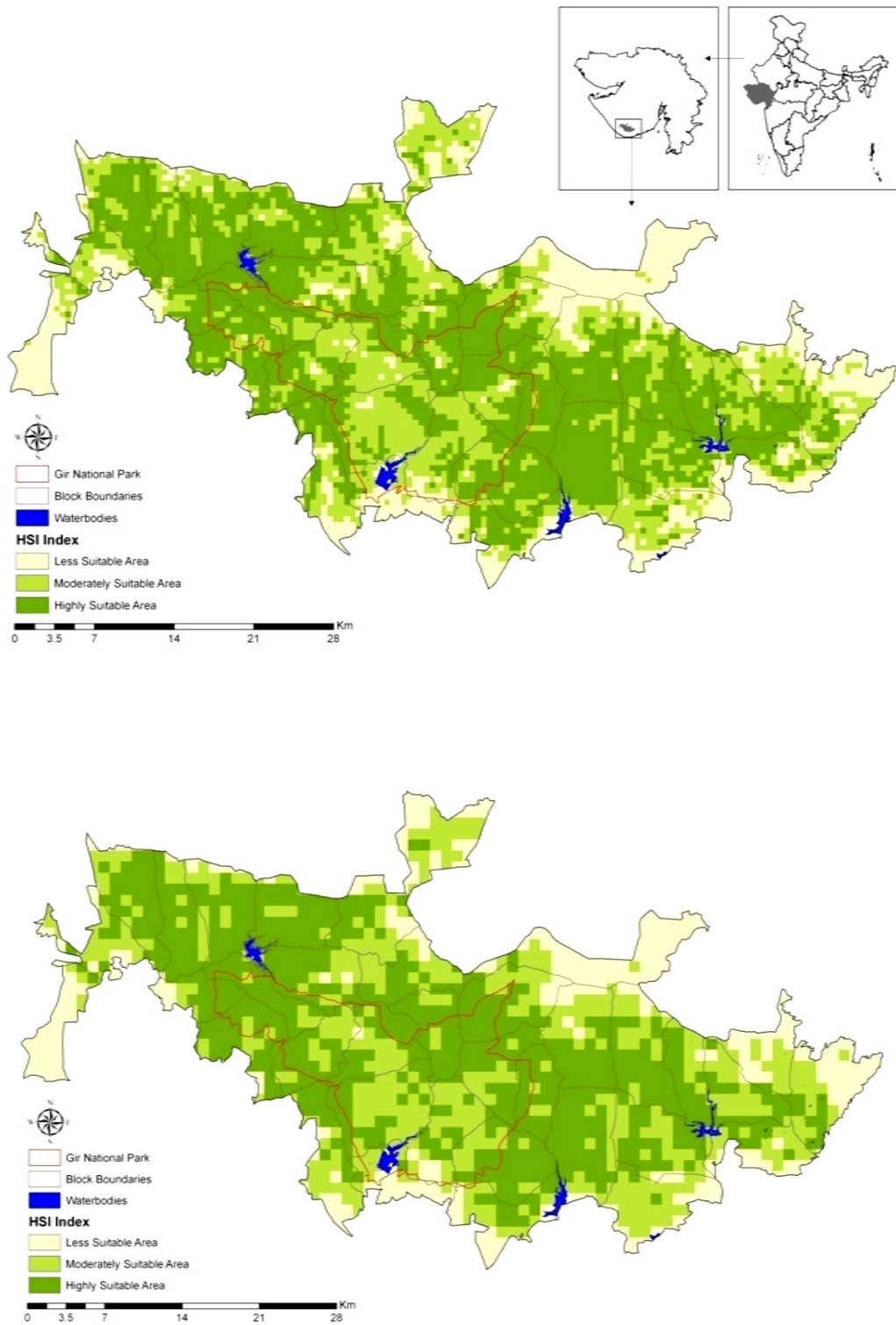


Figure 5.18: Species distribution modeling of Asiatic lions at 500mX500m, 1KmX1Km spatial resolution

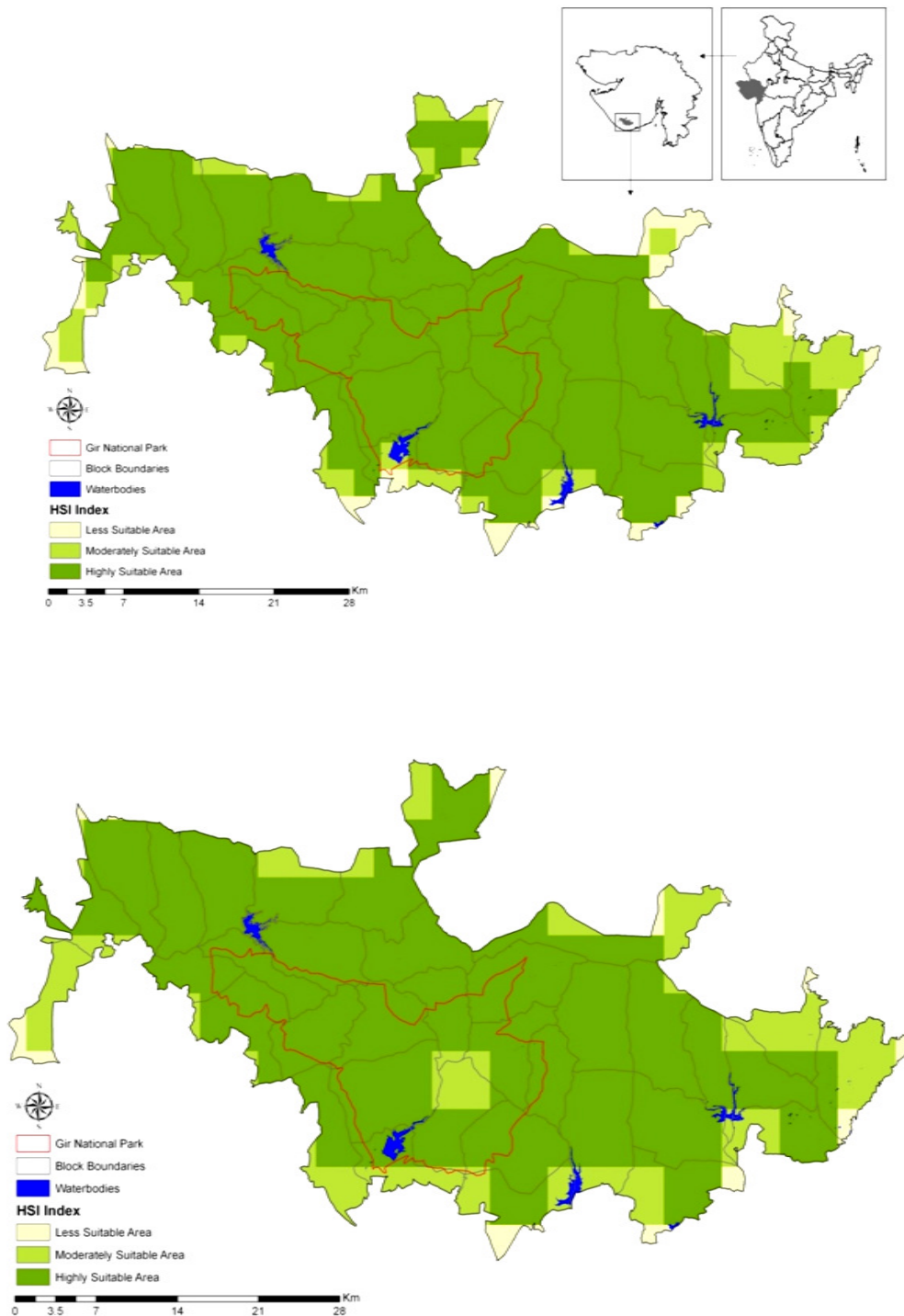


Figure 5.19: Species distribution modeling of Asiatic lions at 2.5 kmX2.5Km, 5KmX5Km spatial resolution

Research based on influence of waterholes on lion's movement concluded that waterholes were the key loci on the lions' route-maps (Valeix et al. 2009). Similar with this result, our study stated that waterholes were a key determinant of lion habitat selection both at home range (5kmX5km) and movement scale (2.5kmX2.5km). Because the distribution of herbivores in this ecosystems is largely influenced by the distribution of surface water, lions have a greater chance of encountering their prey in areas surrounding waterholes, this supports the idea developed from study done elsewhere (Valeix et. al 2009) that animals should spend more time in areas where resources are plentiful than in areas where they are scarce. Water sources have already been considered as crucial in lion habitat selection in the Serengeti (Mosser 2008). Through their influence on habitat selection, waterholes are likely to affect both lion home range size and configuration.

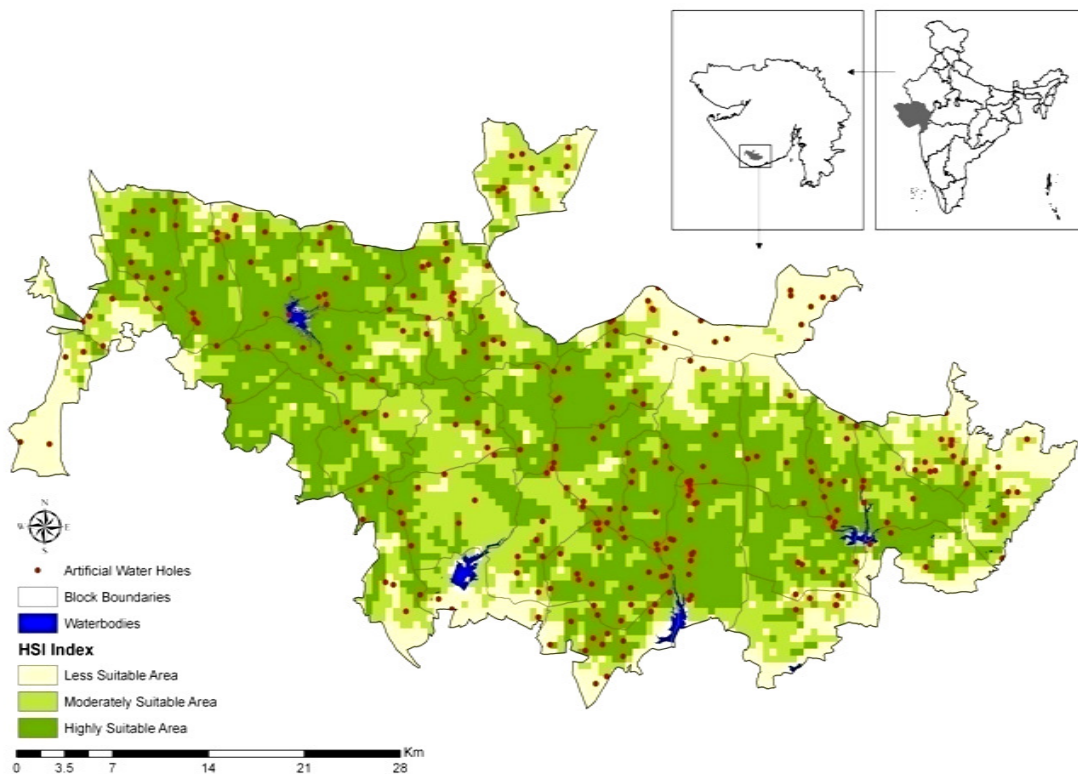


Figure 5.20: Relationship between lion's habitat selection and waterholes distribution

In arid and semi-arid savannas, large herbivores are more abundant in the vicinity of waterholes (Redfern et al. 2003, Valeix et al. 2009) as they need to drink regularly, and are accessible and vulnerable to predation by lions in the vegetation surrounding these water sources (Hopcraft et al. 2005). Hence, it is very likely that lions spend more time in the vicinity of waterholes and explore these areas thoroughly to maximize their encounter rate with potential prey in areas where prey are relatively more abundant and accessible. Our results further illustrated of how key habitat features (e.g. waterholes in this study), determined the dispersion of prey and might have a determining influence on predator spatial ecology and movement patterns. Suitable habitats were dictated by the distribution of these key habitat features (Figure 5.20).

In fine scale HSI modeling (500mX500m) and (1kmX1km), digital elevation model had a significant contribution and animals preferred areas between 50-210 meter in elevation, after which, habitat suitability values showed a declining trend. In a study in Masai Mara national reserve (Ogutu and Dublin 2002) high lion density was observed in areas having much steeper and irregular topography, better drainage, more natural springs and brushy vegetation. It is believed that these features supported more prey animals in the late wet and early dry seasons and resulted in a higher lion density. Similarly in areas with flat, low lying and poorly drained terrain, in receiving high rainfall, the area get waterlogged, which increase the productivity of the area but occurrence of tall grass in dry season increase the predation risk. Hence water logging in wet season and predation risk in dry season lower the cub survivability from starvation and diseases as it is evident from the study (Ogutu and Dublin 2002, Fosbrooke 1963). Unlike Africa's savannah plains, 46.1% of Gir's total area is occupied by plains to gentle slopes, 22.8% by moderate slopes and 31.5% by steep one. It is estimated that about 35% of Gir is not inhabited by the lions because of its hills and slopes (Patel 1992); only about 1,000 sq. kms. of the national park and sanctuary is used by lions. The northern part is more hilly than the southern.

The recent estimate of the overall density (\pm SE) of all wild ungulate species of Gir has been estimated at 48.3 (\pm 6.1) individual/ km² (Dave unpublished). Chital being the most common species with a density of 44.8(\pm 7.2) individual/ km² (Dave unpublished). This variation in

ungulate biomass across the Gir PA may appear to be the factors responsible for the lion social structure. The other important prey of lions, the livestock densities are highest in sanctuary east and probably influencing lions to form larger groups particularly in the vicinity of the cluster of nesses in this area (Meena 2009). Significant contribution of euclidean distance to ness at each scale, proved the above hypothesis. These findings supported one prediction of the resource dispersion hypothesis (Macdonald 1983, Kruuk and Parish 1982, Mills 1982, Kruuk and Macdonald 1985) that, if food is patchily distributed, then patch richness limits carnivore group sizes and patch dispersion, territory sizes. Thus, the dispersion and abundance of prey and vegetation cover appeared basic to the spacing and structuring of lion societies, as well as affecting territory sizes, perhaps, more than pride sizes. The spatial variation in distribution of suitable habitat patches at edges at minimum home range scale (5kmX5km) reflects resource dispersion (Figure 5.19). Suitable habitat is expected to encompass at least a minimum total area of key habitats, representing key prey patches, in absence of this, the habitat at the edges was rated moderately suitable by the algorithm. Probability of presence of lions decreased with the increase in distance from nesses. Lions use dry river channels for movement, hence the significant contribution of this variable at each scale, proved its significance in the spatial ecology of lions.

5.9. Discussion

Management and conservation measures are often proposed but implementation of which failed, because (i) of the problem in identifying indicator variables as they are often neither well defined nor measured adequately, then (ii) being too expensive to sample them over the whole area, and finally (iii) the chosen statistical methods do not allow identifying the most critical variables. Instead of its wide use SDM still face the criticism of including extraneous variables and SDM developed in an arbitrary scale failed to detect the indicator variables, whereas multiscale SDMs helps in identifying the contribution of important predictor variables in different spatial scales. Because SDMs are frequently used in studies using large grid sizes, understanding the effects related to spatial grid size is important in evaluating SDM results. Most future climate scenarios are often interpreted in several orders of

magnitude more generalized than the scale at which species experience the environment (Guisan and Thuiller 2005). Choosing a grain size for modeling is partly a technical issue. For example, it can be related to the grid cell size of available environmental data (Graham et al. 2004), characteristics of the species data (e.g. geographical accuracy, sample size, field survey constraints, or autocorrelation structure; Guisan and Hofer 2003, Gottschalk et al. 2005, Linke et al. 2005, Huettmann and Diamond 2006) or computer power (i.e. too many cells may require too demanding computer resources). Grain size is also a crucial ecological as well as management issue. Changing the grain size can influence the perception of a phenomenon, such as patterns of presence or abundance (Johnson et al. 2002, Tobalske, 2002, Wiens 2002, Graham and Hijmans 2006), or affect the relevance of the output for management applications (Araújo et al. 2005). Working at the wrong scale can be very inefficient.

The influence of grain size on the SDM-derived distribution area (i.e. the surface of the predicted distribution range) remained a poorly addressed question. The SDM-derived distribution areas of nine tree species could undergo up to a four-fold increase when increasing the grain size from 1X1 km² to 64 X 64 km² (Seo et al. 2009). Similarly, the potential distribution of a rare gazelle noticed a sharper increase in the predicted species distribution area (more than 15-fold increase) when increasing the grain size from 1X1 km² to 32X32 km² despite a slight decrease in model quality through upscaling (Hu and Jiang 2010). In our study, for the four species, grain size increase through upscaling gradually modified the shape of the environmental niche of the studied animals. At smaller grain sizes (500m X500m and 1kmX1km) the niche showed a dispersed pattern depicting the actual condition in the field while at higher grain sizes, it became compact. Most of the SDM derived distribution areas increased exponentially with grain size. This increase was primarily due to the increase of the observed distribution area caused by coarsening grain size, mostly at the edge of the distribution due to the presence of the species in cells adjacent to empty cells.

Increasing the operational grain size obviously induces a geometric increase of the observed species distribution area. If a species is present in only one out of four adjacent cells, merging

these four cells when upscaling will lead to a four-fold area increase, whereas merging the cells will not affect the area if the species is present in the four cells. The increase of the observed area with grain size will in turn affect the SDM derived species distribution area, it was found that the predicted species distribution area increase depended on the species range size (Seo et al. 2009). However, holding spatial extent constant by focusing on Britain, the aggregation of climate and species data at a 50 km resolution did not lead to overestimation of thermal tolerance compared with 10 km data (i.e. grain had no effect within Britain). Additionally grain size had no effect on model performance (Guisan et al. 2007). This is supported by other studies which showed that at large spatial extents and coarse grains, variations in DEM resolution did not result in very large changes in estimated surface area, implying limited effect of grain size on topographic and, hence, climatic heterogeneity (Nogue´s-Bravo and Araujo 2006). However, they showed that the effect was greater when using a fine resolution DEM in a high mountain region (Nogue´s-Bravo and Araujo 2006).

Considering model predictive accuracy, the slight decrease of most of the quality indices with grain size was consistent within the findings of previous studies (Guisan et al. 2007, Hu and Jiang 2010). However, sometimes the trend was found taxon-dependent (Guisan et al. 2007). Indeed, plant distribution was better predicted using small grain sizes, whereas model quality was improved by upscaling for some bird species. This improvement was mostly observed for a grain size increase from 0.1 to 1 km². Such an increase of model quality was also observed on green woodpecker *Picus viridis* distribution (Tobalske and Tobalske 1999) when increasing the grain size from 1 to 4 km². However an optimal grain size probably does not exist and that it has to be selected depending on the data quality, the ecology of the species and the goals of the study. Over the range of spatial scales we examined, ungulate and lions habitat selection appeared to be driven by multiple choices at a variety of spatial scales. Habitat selection was differentially influenced by spatial scale both between different habitat components (e.g., forest cover vs. increase in euclidean distance to water sources) and within components (e.g., increased selection at finer scales for some components). Distinctive scale-dependent patterns were evident. As evidenced for various ungulate species (Coulson et al. 2001; Simard et al. 2010), this study also demonstrated that environmental factors had a considerable influence on ungulate abundances.

The strongest relationships with environmental variables were observed at smaller spatial scale (500mX500m) coinciding with foraging scale. Similar conclusions were drawn for eight species of bats and capercaillie (Bellamy et al. 2013 and Graf et al. 2005). Small-scale ungulate–habitat relationships therefore reflected the decisions made by ungulates about where to feed, shelter, drink and avoid predation, anthropogenic disturbances within their home range. Relationships at larger scales reflected the occupancy, carrying capacity of the locality. Because different processes may be operating at different scales, it is not surprising that the relative importance of variables often changed with scale and thus multiscale SDMs performed better than single scale models and would therefore have greater practical value. The most important variables for most species related to presence of forest cover, digital elevation model, euclidean distance to water resources, and euclidean distance to nesses.

When Gir was spread across Junagadh, Rajkot, Amreli and Bhavnagar, the lions had a superior habitat in the plains, but with the progress of human habitation and cultivation, lions were compelled to confine themselves to the hilly environment of the Gir PA (Singh 2007). In Gir PA, the availability of wild prey is not uniform due to the nature of the terrain and the distribution of resources. Areas with high numbers of domestic animals are poor in wild ungulates. The plains of Gir, which are not used as pastures, have exceptionally high concentrations of chitals and sambars. The chhodawadi, Jamwala and Dedakadi ranges in western Gir have a high concentration of ungulates. The hadala, and Jasadhar ranges in eastern Gir and the Ankolwadi, Visavadar, Sasan and Devalia ranges in western Gir support a moderate density of chital, sambar, four horned antelope and wild pig, where the density varies from 30-50 ungulates for every sq km. There are also ranges like Khambha, Dalkhania and Babaria which have poor ungulate density (Singh 2007).

Habitat conditions in Gir seasonally changes from green lush in monsoon to green-yellowish, with golden grass in the post monsoon season and pre winter, to dry in summer, to the emerging fresh leaves in late summer and pre-monsoon. This seasonal variation has immense impact on animal spatial ecology. During the monsoons, the dense grasses, weeds and shrubs are not favored by lions and they try to adjust to more open areas, preferably forest tracks,

villages, open uplands and the peaks of low hills to avoid the flies and insects. The males generally prefer hilltops during the monsoon. In the dry season, the animals, especially the females, often use the riverine tracts. In the summer, the savannah type vegetation or open, thorny forests or leafless trees are not good for the animals as they need shade to escape the heat.

The big leafed *Butea* and the stunted teak in the boundaries of the Gir PA, which remains leafless in winter and summer, produce dense foliage during the monsoon; several short and thorny species like the *Zizyphus* and *Acacia* provide sparse cover so that plenty of grass can grow. Along the water bodies, and sometimes elsewhere, *Carissa carandus* grows into bouffant domes, with cave-like spaces between earth and foliage, which provide a cooler environment, are not only the lion's retreat during hot summer days, but are also shelters for the cubs. The *Carissa* fruit- tangy, blue, spheroid-like sweet berries- is a tasty food for ungulates and birds. Fruits of *Emblica officianalis*, *Diospyros melanoxylon*, *Ficus benghalensis* and *Zizyphus* and many other species are so nutritious, full of proteins, and vitamins that ungulates, and birds compete for them. Due to easy availability of rich food, wild ungulates do not prefer grass after October, when it turns yellow. The short *Zizyphus*, *Acacia* (hermo, khair and babool), *Emblica* and *Bauhinia* are good for browsers. In winter and summer, the fruits of *Ficus*, *Emblica*, *Zizyphus*, *Acacia*, *Carissa*, *Bauhinia* are so abundant that herbivores do not face any shortage of food. Most of the ungulates stop feeding on grass after October/November and switch to the nutritious leaves and fruits of *Acacias* and *Zizyphus*.

SDMs derived from this study can be used in opportunity mapping for planning both large- and small-scale habitat management for maximum effectiveness by identifying, for example, critical locations for corridors and networks, hot spots for diversity. Amongst all ungulates, found in Gir, probability of nilgai presence was high in places nearby *nesses*, showing the human tolerant nature of the species and competition for the same space amongst man and animal. As already mentioned, nilgai is inhabitant of open patches, so with the increase of area of open forest patches, probability of species presence increased, but importance of cover was also emphasized, when probability of species presence decreased with the

presence of very wide open forest patches. Presence of nesses and artificial water holes proved to be most significant at home range scale, rather than fine local scale. Precipitation seasonality and digital elevation model contributions were highest in local scale rather in home range scale. In each scale, analysis of SDM of nilgai revealed that, suitable habitat of nilgai was distributed in the range of 145-155 of precipitation seasonality, and after this range the probability of the species presence declined. The species, being inhabitant of flat terrain, with the increase of elevation, the probability of finding suitable habitat patches declined. Gir, being a part of the semi-arid ecosystem where wildlife is influenced by river channels and dams, distribution of nilgai, was found to be also affected by this very important factor, and with the increase in distance from the river channels, probability of presence of the species declined. Probability of presence of nilgai, found increasing upto the value of 0.5 of post-monsoon NDVI and the same trend was observed with the pre-monsoon NDVI. Due to its preference for open and undulating areas (Berwick 1974), the nilgai population was mostly restricted to eastern sanctuary and sanctuary border (Khan 1993). The nilgai density was relatively low but has increased from 0.37 per km⁻² in 1990 (Khan, 1997) to 1.16 ± 0.47 km⁻² (Dave 2008).

Mediated through edaphic and topographic gradients, rainfall determines habitat quality and structure through its influence on vegetation, (Bell 1982, McNaughton 1983, 1985) and can induce changes in habitat suitability capable of substantially modifying predator-prey relations (Smuts 1978, Whyte and Joubert 1988, Mills et al. 1995). Besides food and vegetation cover, rainfall also affects the distribution of drinking water, thereby modulating the spatio-temporal distributions of water-dependent herbivores and carnivores (Western 1975, Hanby et al. 1995, Owen-Smith 1996). In Gir, a pronounced gradient of increasing rainfall from east to west is superimposed on complex geological and geomorphic templates. Rainfall is positively linearly related with teak dominated communities and sambar habitat preferences. The higher probability of sambar suitable habitat in the higher rainfall areas inferred, rainfall shapes the spatial distribution of sambar. In the Gir forest, the sambar population found larger in the western and central zones that have dense forest cover and abundant water, and was low in the sparse and thorny forests of the northern and eastern zones (Johnsingh 1983, Khan et al. 1990, Singh 2007).

In each scale of SDM, probability of the presence of sambar, increased with the increase in precipitation and it was highest in the range of 145-153. Being an inhabitant of hilly terrain, probability of finding sambar was highest at approximately 200 meter in elevation. With the increase in value of post-monsoon NDVI, the probability of species presence increased, same trend was observed with the pre-monsoon NDVI. Being an elusive species and intolerant of human presence, with the increase in euclidean distance from nesses, the probability of species presence also increased. With the more availability of open forest patches probability of species presence declined whereas the reverse trend was observed with the increase in dense forest patches. Presence of water resources found significant in predicting spatial distribution modeling of the species and with the increase in distance from waterholes, probability of species presence declined, implying the importance of distribution of water holes in conservation and management of the species. In a study of the impact of management practices on lion and ungulate habitat of Gir, it was recommended a minimum distance of 8 kilometers between two nesses to leave disturbance free forest of sufficient size between them for sambar (Sharma and Johnsingh 1996) which was evident from the sambar species distribution modeling prediction at fine scale (500mX500m). It was observed that probability of sambar presence reached its highest value at or beyond 8 km distance from the nesses.

Chital prefer forests of the plains and undulating terrain; they avoid steep slopes and sighting them in the hills is very rare. They graze in the monsoon when grass is plentiful and green. In winter and summer, the fruits of *Ficus*, *Acacia*, *Emblica*, *Syzigium*, *Zizyphus*, *bauhinia*, and other species form their rich diet. During scarcity, they eat the fallen leaves of *Acacia*, *Zizyphus* and other fodder trees. Chital attains its highest densities in areas with flat topography, supporting *Tectona-Acacia-Zizyphus* woodland, thorn woodland and riverine woodland (Singh 2007, Khan et al. 1990). The construction of large reservoirs in Gir (e.g. Kamleshwar, Machundri) has also indirectly benefited the chital population by creating optimum habitat conditions. Open areas on the periphery of these reservoirs have been recolonized by *Acacia* and *Zizyphus* species as a consequence of secondary succession which intermix with *Tectona-Acacia-Zizyphus* woodland and riverine woodland. This provided a high diversity of habitats and food species with ample ecotones which chital is known to

prefer (Mishra 1982). These factors were possibly responsible for a greater increase in chital density in south west (SW) and parts of national park (NP) after the removal of domestic livestock. It is however, likely that the more hilly terrain, relative scarcity of water and subsequent increase in vegetation cover due to protection from livestock grazing and fire in the NP has limited the increase in chital density compared to the SW. The removal of domestic livestock from south east (SE) was marginal and it still remained overgrazed for most of the year. This coupled with low water availability in SE, has not allowed the chital population to grow at the same rate as in SW and NP.

Chital usually drink water once a day, and more frequently in summer. This has made them inhabitants of forest tracts with widely scattered but assured presence of water. Dense forest patches contributed most in all chital distribution models at different scales. Based on AUC value, the model developed at 500m scale was selected best with AUC 0.81 and the model incorporated all important resources for the species survival. The other models were also in acceptable range with AUC 0.74 (1kmX1km), 0.75 (2.5kmX2.5km) and 0.74 (5kmX5km). Importance of euclidean distance to water sources increased as we moved towards fine scale model supporting the behavior of the species. Pre and post monsoon NDVI both were vital in fine scale model than at home range scale, revealing the forest dwelling nature of the species.

The grazing distribution patterns of large herbivores are affected by abiotic factors such as slope and distance to water and by biotic factors such as forage quantity and quality (Bailey et al. 1996). The major abiotic factor determining the large-scale distribution patterns of large herbivores especially, chital was the water availability and terrain. In each scale of SDM, probability of the presence of species, increased with the increase in precipitation and the most suitable habitat patches were found in the range between 145-155. Chital distribution increased upto the elevation of 200 meter, and then the probability of presence of species started declining. With the increase in value of post-monsoon NDVI, probability of presence of the species increased whereas response of SDM with pre-monsoon NDVI was bimodal. With the increase in area of open forest patches, the probability of presence of species declined whereas the trend was reverse with the increase in area of dense forest patches.

Cheetal distribution found declining with the increase in euclidean distance to waterholes (Figure 5.15 and 5.16).

Chapter 6

**Modeling habitat suitability of a
species of conservation concern: Two case studies of
Asiatic lions emphasizing two major conservation issues**

Chapter 6 A

**Evaluating the potential for Asiatic lion (*Panthera leo persica*)
range expansion within Saurashtra, Gujarat.**

6A.1. Introduction

With the human population expansion and development, all of the world's large carnivores have experienced major shrinkages in their geographical ranges, often becoming restricted to areas (particularly protected areas) where human densities are low (Linnell et al. 2001, Woodroffe 2001). Human influences on large carnivore populations spread even into protected areas, mainly for wide-ranging species that often range beyond reserve boundaries. Such species are in risk from people in adjoining unprotected lands. The resulting edge effect may be severe enough to bring about the extinction of the protected populations (Woodroffe and Ginsberg 1998). Thus understanding a species' fundamental niche (Hutchinson 1957) and the threats to its survival are important aspects for the future conservation of the species.

In consequence of human tolerance, population increase, habitat shortage within protected area, lions, now live in a large part of Saurashtra, from where they were once extirpated. Besides Gir, lions permanently exist in the coastal forest patches of Kodinar, Girnar WLS and human dominated landscape of Amreli, Junagadh, Bhavnagar districts (Mitiyala WLS, Savarkundla along the river Shetrunji, Hipavadli, Palitana, Shihor). Lions being habitat generalists persist in some places of Saurashtra where adequate prey base is available and in absence of wild prey, their dependency on livestock intensifies the chance of human-animal conflict. Many areas recolonized by lions, are not wilderness but highly altered landscapes in vicinity to human

habitation. Further, many recolonizing populations are detached from source populations by distances and barriers that restrict the exchange of animals. Because management objectives often include supporting disjoint lion populations, it is imperative to examine what types of areas and conservation investments are obligatory for population survival. Furthermore predicting favorable lion habitat thus becomes a process of locating areas that contain refuge areas, sufficient prey and provide security from humans to minimize conflict.

The existence of a species in a location follows three limitations (Soberon and Peterson 2005, Soberon 2007): (i) the local environment permits the population to grow (Grinnellian niche), (ii) the interactions with other local species (predation, competition, mutualism, etc.) allow the species to persist (Eltonian niche), and (iii) the location is actually reachable, given the dispersal abilities of the species. These constraints govern the geographical distribution of the species. Understanding habitat relationships of endangered species is particularly important because habitat management is a critical component of conservation planning (Morrison et al. 1998). Studies of habitat selection patterns of animals provide information about areas and resources that influence the fitness of individuals and sustainability of populations (Fretwell and Lucas 1970, Powell et al. 1997). While protected areas play a vital role in large carnivore conservation, many carnivore species of concern are wide-ranging, and the existing reserve network is inadequate for their long-term conservation (Woodroffe and Ginsberg 1998, Linnell et al. 2001, Woodroffe 2001, Marker and Dickman 2004).

Growth of urban, residential, and agricultural land uses eliminates and fragments the habitat available for species to survive. In such situations, success of any conservation policy, targeted for a species threatened with habitat fragmentation, demands appropriate understanding of its underlying metapopulation framework and dynamics across the temporal scale. A metapopulation is defined as a "set of local populations which interact via individuals moving amongst populations (Hanski and Gilpin 1991). Metapopulation is defined as a "set of local populations within some larger area, where typically migration from one local population to at least some other patches is possible" (Hanski and Simberloff 1997). Each metapopulation is continuously being reformed by increases (births and immigrations) and decreases (deaths and

emigrations) of individuals, as well as by the rise and closure of local populations contained within it. As local populations of a given species vary in size, they become susceptible to extinction during times when their numbers are low and extinction of such local populations is common. The regional persistence of such species is reliant on the existence of a metapopulation and elimination of much of the metapopulation structure of the same species can increase the chance of its regional extinction.

Therefore present conservation strategies for mammalian carnivores contain the management of disjunct populations, which are detached from neighboring populations by barriers which limit but do not prevent the exchange of animals (Mech 1995). Managing disjoint populations is crucial because survival at a larger regional scale often depends on the growth and dispersal characteristics of local populations (Fahrig and Merriam 1994). Forest fragmentation and degradation have been assumed to be causing a decline in the Asiatic lions habitat, however little is known about the distribution of this endangered species at landscape level and no quantitative or systematic studies have attempted to address lions distribution in relation to composition and configuration of the landscape. Given the alarming situation of extreme habitat alterations due to human activities, it is important to accurately assess the distribution and conservation status of this threatened species.

In background of this reality, concept of management of extended Gir in new range of lions has become a necessity, (a) Assessing the population size and extent that the state can support (b) Specific areas that have preferred habitats for targeting conservation investments (c) Preparing areas and human attitudes to accommodate lions (d) Putting strategies in place to mitigate human-lion conflicts (e) Identifying habitat patches and corridors timely so as to put in place policy and management strategy to ensure their future conservation value is not compromised by unplanned developmental activity.

6A.2. Justification of the Study

The harmful effects of habitat fragmentation are believed to be exerted on species largely through their sensitivities to changes in habitat area and the degree of isolation of habitat patches (Abensperg- Traun and Smith 1996, Nupp and Swihart 2000). In

the agro-pastoral landscape of Saurashtra, where, developmental pressure is very high, the distribution of lion is determined by a combination of natural habitat and anthropogenic influences. Understanding the spatial effect of these two processes is vital for making spatially explicit decisions about conservation actions. The landscape within the distributional range of lions is not wilderness but a mosaic of forest, agricultural and developed land under a variety of private and public ownerships. Therefore, it is imperative to recognize areas that need to be managed intensively, providing suitable habitat to the increasing lion population. Moreover, this study provides a specific example of how spatial ecology can be used to design matrix conservation approaches. A better understanding of lion's habitat can be used to design habitat management and restoration strategies so that modern development and conservation go hand in hand with minimal conflict between these two important social and economic objectives of the state.

6A.3. Study Area

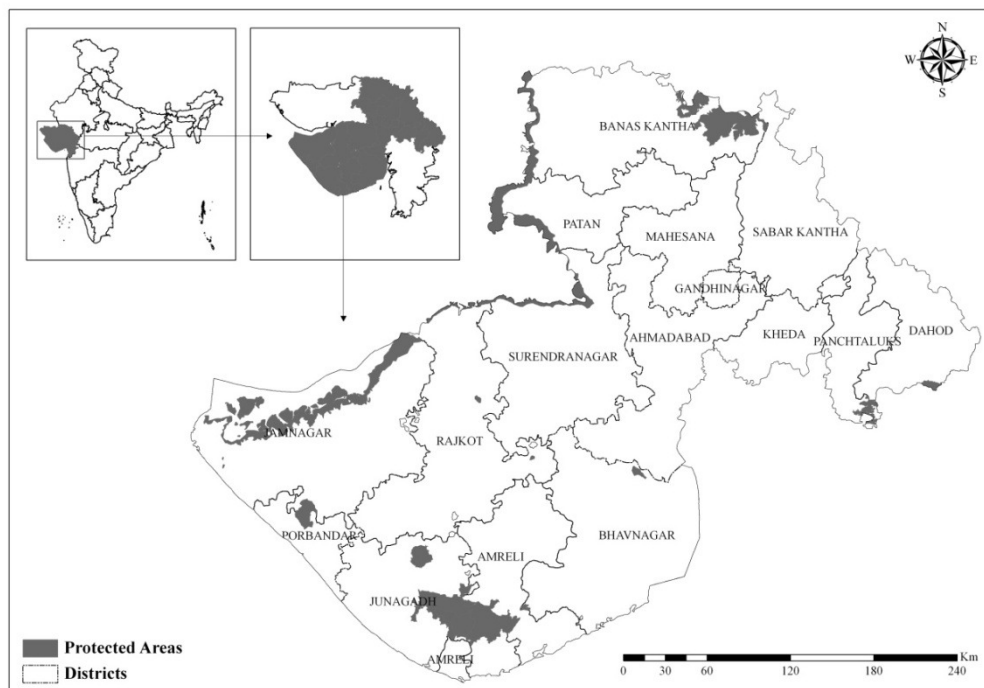


Figure 6A.1: Location of the study area within India and Gujarat

The area under study (Figure 6A.1) incorporates 17 districts of Saurashtra *viz.* Junagadh, Amreli, Porbandar, Jamnagar, Rajkot, Bhavnagar, where lions have already

spread and the following districts namely Surendranagar, Ahmadabad, Kheda, Panchtalukas, Gandhinagar, Mahesana, Patan, Banaskantha, Sabarkantha, Dahod where lions could possibly disperse within Saurashtra. The idea was to identify maximum available suitable habitat in the entire Saurashtra landscape where prevalent land use/cover pattern includes very dense forest, moderate dense forest, open forest, nonforest, scrub and waterbodies interspersed with human land uses like settlements, cities, townships, industry, mines, orchards and agriculture. Main rivers flowing in this area are Sani, Minsar, Ozat, Madhuwati, Megal, Saraswati, Shingoda, Matan, Malan, Bagod, Shetrunji, Kalubhar, Ghelo, Sukhbhadar, Bhogavo, Limdino, Sabarmati, Viswamitri, Dhadhar, Vatrak, Meshovo etc.

Crisscrossed by all these rivers, this area is very fertile, therefore under the influence of agricultural expansion. Surrounded by agriculture/urbanization from all sides, and distributed as island like features, several protected areas *viz.* Porbandar Wildlife Sanctuary (WLS), Khijadiya WLS, Hingolgarh Nature Reserve WLS, Blackbuck National Park (NP) Ratanmahal WLS, Jambugodha WLS, Barda WLS, Balaram-Ambaji WLS, RampuraVidi WLS, Paniya WLS, Mitiyala WLS, Jessor WLS, Gir WLS and NP, Gaga WLS, in this landscape are less viable in conserving species in absence of connectivity amongst them. The predicted north-eastward dispersal of lion's faces, bottleneck condition, at many places while moving from one protected area to another. Because of change in land use/ cover and infrastructure development in this area the movement of individuals between populations is reduced. This movement is essential to preserve the demographic and genetic vigor of the last free ranging Asiatic lion population.

6A.4. Objective

1. The objective of the study was to examine the distribution of suitable lion habitat in Saurashtra, especially in areas outside protected area to assess the potential of the landscape in supporting increasing lion population.

6A.5. Materials and Methods

The whole Saurashtra was chosen as reference area. The relative contribution of certain environmental and topographic variables to the geographic distribution of Asiatic lions was analyzed using Environmental Niche Factor Analysis (ENFA) (Hirzel 2002). Data from 19 radio collared lions within and outside of Gir PA along with lion sightings and lion scat locations were used as inputs in the models. ENFA is a presence only model that uses species presence locations along with other ecogeographical variables to compute the habitat suitability of the species. Information on recent land use/cover, digital elevation model, 19 bioclimatic variables (Hijmans et al. 2005), human disturbance, distance to roads and rivers etc. were used to generate the model (Table 6A.1). I did not include prey abundance into the models because prey density is not always a good predictor of suitable carnivore habitat (Mladenoff et al. 1995) and reliable regional datasets were unavailable. I used land cover as a surrogate for prey abundance. All variables were extracted to 1km X 1km grids and attributed to respective GIS raster coverages in Idrisi. Coverages were then exported to Biomapper and tested for correlation (with each other). Where two-or-more variables were strongly correlated ($r > 0.7$, and including negative values) were discarded.

Table 6A.1: Ecogeographical variables used in habitat suitability modeling

S. No	Type	Abbreviation	Variables	Reference
1	Habitat Composition	LULC	Land use/cover information	FSI,2009
2	Habitat Configuration		Landscape indices	FSI,2009
3	Topography	DEM	Digital elevation model	http://asterweb.jpl.nasa.gov/gdem.asp
4	Water Resource	EURVR	Euclidean distance to river channels	Digitized dataset
5	Anthropogenic Pressure	EURD	Euclidean distance to roads	Digitized dataset
6	Anthropogenic Pressure		Human footprint	http://sedac.ciesin.columbia.edu/data/set/wildareas-v2-human-footprint-geographic
7	Climatic Variables	BIO1	Annual Mean Temperature	Hijmans et al. (2005)
8	Climatic Variables	BIO2	Mean Diurnal Range (Mean of monthly (max temp - min temp))	Hijmans et al. (2005)
9	Climatic	BIO3	Isothermality (BIO2/BIO7) (*)	Hijmans et al. (2005)

S. No	Type	Abbreviation	Variables	Reference
	Variables		100)	
10	Climatic Variables	BIO4	Temperature Seasonality (standard deviation *100)	Hijmans et al. (2005)
11	Climatic Variables	BIO5	Max Temperature of Warmest Month	Hijmans et al. (2005)
12	Climatic Variables	BIO6	Min Temperature of Coldest Month	Hijmans et al. (2005)
13	Climatic Variables	BIO7	Temperature Annual Range (BIO5-BIO6)	Hijmans et al. (2005)
14	Climatic Variables	BIO8	Mean Temperature of Wettest Quarter	Hijmans et al. (2005)
15	Climatic Variables	BIO9	Mean Temperature of Driest Quarter	Hijmans et al. (2005)
16	Climatic Variables	BIO10	Mean Temperature of Warmest Quarter	Hijmans et al. (2005)
17	Climatic Variables	BIO11	Mean Temperature of Coldest Quarter	Hijmans et al. (2005)
18	Climatic Variables	BIO12	Annual Precipitation	Hijmans et al. (2005)
19	Climatic Variables	BIO13	Precipitation of Wettest Month	Hijmans et al. (2005)
20	Climatic Variables	BIO14	Precipitation of Driest Month	Hijmans et al. (2005)
21	Climatic Variables	BIO15	Precipitation Seasonality (Coefficient of Variation)	Hijmans et al. (2005)
22	Climatic Variables	BIO16	Precipitation of Wettest Quarter	Hijmans et al. (2005)
23	Climatic Variables	BIO17	Precipitation of Driest Quarter	Hijmans et al. (2005)
24	Climatic Variables	BIO18	Precipitation of Warmest Quarter	Hijmans et al. (2005)
25	Climatic Variables	BIO19	Precipitation of Coldest Quarter	Hijmans et al. (2005)

6A.6. Software

We choose Ecological-niche factor analysis (ENFA), developed by Hirzel et al. 2002 for this study. Based on Hutchinson's niche concept (Hutchinson 1957) ENFA identifies the variables regulating the niche of the species and differentiate between the available conditions and the habitat selected by the species. Therefore, ENFA is the single analysis based on the concept of ecological niche that defines precisely the specialization, in addition to the marginality. In the ecological space it searches for directions so that (i) the difference between the conditions used in average by the species and the conditions available on the study area (i.e., the marginality) is maximized, and (ii) the ratio between the variance of available conditions on the variance of conditions used by the species (specialization) is maximized. Whereas

MaxEnt (Phillips et al. 2006) assess species geographic distributions in relation to environmental variables with limited presence only data, and deals with spatial distribution not the habitat preferences as it is dealt by ENFA.

ENFA compares the distributions of the ecogeographical variables (EGVs) between the presence data set and the whole study area. Several EGVs are summarized in a few uncorrelated factors retaining most of the information. ENFA was developed using Biomapper 3.1 (Hirzel et al. 2004) following the procedures outlined by Hirzel et al. 2002. Similar to PCA, ENFA changes the original predictor variables into new, uncorrelated axes. Contrasting PCA, where the successive axes are designated to match the direction of maximum variance in the multi-dimensional space, the principal components of ENFA have ecological significance (Hirzel et al. 2002). The outputs of the ENFA incorporated eigenvalues and factor scores. The amount of variance explained by factors is explained by the eigenvalues. The first factor, called marginality, described the direction in which the species niche differs most from the available conditions in the study area (Hirzel et al. 2002). The coefficients of the score matrix related to the marginality factor indicated the correlation between each EGV and the factor. The greater the absolute value of the coefficient, the higher this EGV adds to the marginality. A low value (close to 0) specifies that the species inclines to live in average conditions throughout the study area, while a high value (close to 1) directs a tendency to live in extreme habitats. A positive value means that the species “prefers” the high values of this EGV, while a negative value means that the species “prefers” the low values. The following factors are termed as specialization factors, and are sorted by decreasing amounts of explained variance. These factors define how specialized the species is, by reference to the available range of habitats in the study area (Hirzel et al. 2002). Therefore, only a few of the first factors explain the major part of the whole information. Specialization ranges from 1 to infinity and thus is tough to interpret. For this reason, tolerance is used which measures the choosiness of the species about the available range of EGVs. Defined as the inverse of Specialization ($1/S$), tolerance ranges from 0 to 1, representing specialist species (stenoic) tending to live in a very narrow range of conditions or species that inhabit any of the conditions in the study area (eurioc). The distribution of the eigenvalue is compared to the distribution of Mac Arthur's broken-stick which is the expected

distribution when breaking a stick randomly. Therefore, the eigenvalues that are larger than what would have been obtained randomly may be considered "significant".

Based on the factor scores, a habitat-suitability map was computed, using the geometric mean algorithm. The number of factors included in the habitat-suitability map was decided according to the explained information of the model, i.e. the amount of variance of the data explained by marginality and specialization, and the explained specialization, i.e. the amount of variance of the data explained by the model (Hirzel et al. 2002). The habitat-suitability map was evaluated for predictive accuracy by a cross-validation procedure (Boyce et al. 2002). The species locations were randomly partitioned into k mutually exclusive but identically sized sets. Each k minus 1 partition was used to compute a habitat suitability model and the left-out partition was used to validate it on independent data. This process was repeated k times, each time by leaving out a different partition. This process resulted in k different habitat-suitability maps and the comparison of these maps and how they varied, provided an assessment of their predictive power. The number of partition used was four. Each map was reclassified into i bins, where each bin i covered some proportion of the total study area (A_i) and contained some proportion of the validation points (N_i) (validation points were the observations left out during the cross-validation process). The number of bins used was three. The area-adjusted frequency for each bin was computed as $F_i = N_i / A_i$. The expected F_i was 1 for all bins if the model was completely random. If the model was good, low values of habitat suitability should have a low F (below 1) and high values a high F (above 1) with a monotonic increase in between. The monotonicity of the curve was measured with a Spearman rank correlation on the F_i (Boyce et al. 2002, Hirzel et al. 2004). Three presence only evaluation measures were calculated, the Absolute Validation Index (AVI), the Contrast Validation Index (CVI) (Sattler et al. 2007) and the continuous Boyce Index (BI; Hirzel et al. 2006). AVI indicates how well the model discriminates high suitability from low suitability areas and varies from 0 to 1, while CVI indicates how much the AVI differs from what would have been obtained with a random model and varies from 0 to AVI. BI varies from -1 to 1, with 0 indicating a random model.

6A.7. Results

ENFA modeling indicated four geographically distinct prime lion habitat clusters in Gujarat (Figure 6A.2). 1. One cluster located at the coastal belt formed with Gir as a source patch with limited, fragile connectivity with other sink patches viz. Girnar, coastal patches at Kodinar, Rajula and *Prosopis* patches at the bank of Shetrunji river. 2. Second cluster around Barda and adjoining landscape. 3. The Third cluster is located at the north eastern side of the state formed of protected areas viz. Jessor wildlife sanctuary, Balaram-Ambaji wildlife sanctuaries. 4. The Fourth cluster located at the adjoining landscape of Jambugoda WLS and Ratanmahal WLS.

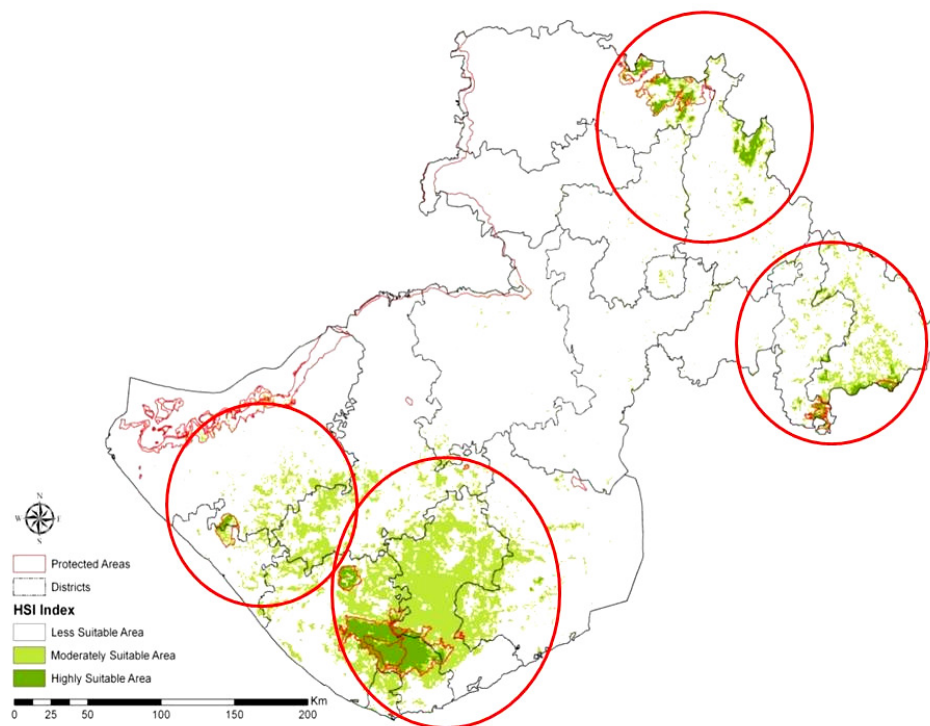


Fig 6A.2: Areas identified having potential to support range expansion of Asiatic lions (*Panthera leo persica*) in Gujarat.

From all the eco-geographical variables (EGVs), I used global and species correlation tree to keep uncorrelated variables. Eigen values and vectors were used in determining the contribution and correlation of EGVs and based on these two, marginality and specialization factors were computed. From the remaining EGVs (n=13), ENFA provided one marginality factor and 12 specialization factors totally

uncorrelated, each factor being a linear combination of the predictors (Table 6A.2). Among these factors I kept only those explaining a significant amount of total variance by comparison with a broken-stick distribution (always greater than 75%). The median algorithm predicted higher suitability values, resulting in wider core and marginal habitats. These areas are also fragmented, with unsuitable patches occurring inside globally good regions. By contrast, the geometric mean predicted fewer and more compact habitat patches. Many small marginal patches predicted by the median algorithm lack in the geometric mean map.

Table 6A.2: Score matrix computed from ENFA

Score matrix				
Eco-Geographical Variables	Marginality Factor (43%)	Specialization Factor1 (44%)	Specialization Factor2 (4%)	Specialization Factor3 (3%)
Precipitation Seasonality (Coefficient of Variation)	0.03	0.85	-0.20	0.36
Precipitation of Coldest Quarter	-0.21	-0.23	0.61	0.37
Temperature Annual Range (BIO5-BIO6)	-0.24	0.47	-0.34	-0.19
Digital Elevation Model	0.26	0.08	0.21	0.31
Euclidean Distance to Roads	0.01	-0.03	-0.08	-0.05
Euclidean Distance to Rivers	-0.03	0.03	-0.08	0.01
Human Disturbance	-0.1	-0.05	-0.01	0.08
Area of Moderate Dense Forest Patches	0.77	0.01	-0.11	-0.02
Post monsoon NDVI	0.28	0.02	0.35	0.01
Area of Open Forest Patches	0.37	-0.01	-0.09	-0.02
Area of Scrub Patches	0.04	0.03	0.03	0.07
Area of Very Dense Forest Patches	0.09	-0.01	-0.03	-0.02
Area of Water bodies	-0.03	-0.05	-0.53	0.77

Application of ENFA provided an overall marginality of $M = 2.4$, and an overall specialization value of $S = 3.5$, showing that the preferred habitat of lions differ from the mean condition available at this human dominated landscape. Two factor maps calculated for habitat suitability calculation accounted for 93% information and explained 86% specialization. The marginality factor alone accounted for 43% of this total specialization, meaning that lions display a very restricted range on those conditions for which they mostly differ from background Saurashtra conditions.

Marginality coefficients (Table 6A.2) showed that, distribution of lions is linked to area of moderate dense forest patches (0.77), open forest patches (0.369), post monsoon NDVI (0.27), elevation (0.26). By contrast lions tends to avoid increase in temperature annual range (-0.24), increase in precipitation of the coldest quarter (-0.21), increase in euclidean distance to drainage (-0.03), increase in human disturbance (-0.1). Area of scrub patches, very dense forest patches, waterbodies, and euclidean distance to roads had only marginal effects. The very large eigenvalue (62.0) attributed to this first factor explained that lions are sensitive to shifts from their optimal conditions on this axis. The next factors accounted for some more specialization, mostly regarding precipitation seasonality, temperature annual range (second factor) as well as precipitation of coldest quarter (third factor), area of waterbodies (fourth factor). A suitability map was computed based on two factor maps for the whole landscape, which is plotted above in figure 6A.2. The mean value of Boyce's index (0.926 ± 0.08) and the linear nature of the curve (Figure 6A.3) predicted accuracy of the model.

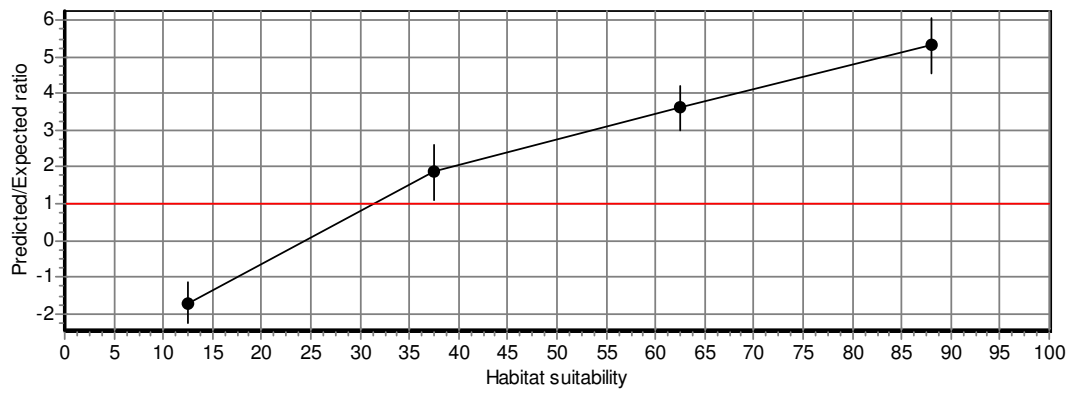


Fig 6A.3: Area-adjusted frequency distribution showing the mean and standard deviation of Boyce Index value, predicting ENFA model accuracy for lions

6A.8. Discussion

Lion density is found influenced by multiple factors, related positively to herbivore biomass, annual mean rainfall, soil nutrients and annual mean temperature, and with interactive effects between rainfall and soil nutrients (Schaller 1972, Smuts 1978, Van Orsdol et al. 1985, Hanby et al. 1995, Ogutu and Dublin 2002). Hierarchical partitioning (HP) (Mac Nally 1996, 2002) permitted assessment of independent

effects of each variable on lion ecology, taking into account all other variables. It established that aspects of lion ecology (lion density, females per pride, etc.) are most strongly associated with rainfall, temperature and landscape features, while herbivore biomass appears not to make important independent contributions, most likely because herbivore biomass is itself strongly dependent on rainfall and temperature.

The presence of lions was recorded long back in north Gujarat, especially along the banks of the Sabarmati, in Banaskantha and Mt. Abu, Rajasthan (Divyabhanusinh 2005). The existence of lions was confirmed from Palanpur (1814), Ahmedabad (1830, 1836), Baroda (1832), Anandara, on the border of Rajasthan (1972), Deesa (1878) and Mt. Abu (1867, 1881). Additional record referred to Rajkot (1832-33), Deesa (1967, 1878), Ahmedabad (1867), Barda (1868), Chotila (1868), Dhrangadhra (1868), Dank (1868), and Chorwad (1868). The climate, terrain and vegetation types across Gujarat, excluding south Gujarat, were suitable habitats; in the savannah-type of ecosystem like that of Saurashtra and the lion persisted in the landscape (Singh 2007).

The present distribution range of the Gir lion spreads in three districts - Junagadh, Amreli, and Bhavnagar. Nine Talukas of Junagadh (Una, Kodinar, Talala, Maliya, Mendarda, Junagadh, Visavadar, and parts of Keshod and Mungral) six Talukas of Amreli (Dhari, Liliya, Khambah, Jafrabad, Rajula, and Savarkundla) and part of three talukas in Bhavnagar (Mahuva, Palitana and Gariyadhar) form the extended home of the lion. These 18 talukas cover over 8,500 sq. km. area and if the areas recently visited by some lions, especially nomads are also included, this range extends to cover as much as 10,500 sq. km (Singh 2007). The distribution range includes of protected areas (1,470 sq kms) including Mitiyala, peripheral forests (403 sq kms), satellite area of Mitiyala, Girnar, and the coastal forests (290 sq kms), Government and private grasslands in and around Hippavadali (230 sq kms), and the villages within the range of these three districts (Singh 2007). None of these areas where the lions have moved within Saurashtra is large enough to sustain a healthy breeding population on a long term basis. On the other hand, if the lion population goes up in such pockets, they could become potential flashpoints of human-lion conflicts (Divyabhanusinh 2005).

Species distribution models represent differential use of habitats, which may lead to different pattern of occurrence or abundance in different parts of the range. At the

regional scale, I identified priority conservation areas in Gujarat using species distribution models to identify sites with the most suitable habitat for lions. The goal was to identify areas where to manage for the better sustenance of the increasing lion population. There are small scrub patches distributed all across the state, sustainability rate of which is very low and these patches need to be prioritized for conservation investments, since they act as stepping stones connecting big clusters. These small patches are mainly village range lands or *Prosopis* patches characteristic of this semi-arid landscape, *Vidis*, the grasslands of varying extents with thorny *Zizyphus* and *Carissa* bushes and scattered *Acacia* trees notified as reserve and non-reserve categories to meet fodder supplies during droughts.

It was found that at the broad spatial scale the distribution of Asiatic lions was primarily limited by availability of moderate dense forest patches, climatic conditions, while at the fine scale, within its potential range, lions appear to be limited by the availability of its preferred habitats, topographic conditions, and other necessary resources (Described in Chapter 5). Similar pattern was observed when climatic variables determined the potential regional range of a rare plant species, and at the local scale land use related variables had a stronger effect on distribution patterns (Lomba et al. 2010). This resembles to cross species patterns of rarity, in which climatic variables tend to be the most limiting factor over the entire range of the species, while at finer spatial scales responses to climate are often masked by responses to local environmental variables such as soils, terrain, and habitat type (Gaston 1994). From studies done on African lions, it was revealed that, there is complex connection between climate and pride structure and seasonal climate changes influence lion demography (Bertram 1973); older lions and cubs might die of starvation (Schaller 1972, Van Orsdol et al. 1985, Orford et al. 1988) and prides become smaller. The combination of climate and soil influences plant abundance, in turn affecting herbivore abundance and therefore carnivore abundance.

Understanding in what way organisms explore and exploit their surroundings is a key topic in ecology, and the assessment of factors affecting their habitat usage is of great importance for conservation. In the last ten years pockets of forests in Girnar, Babra *vidi*, Mitiyala, coastal Junagadh and Amreli, Dhari, Chatariya and Hippavadli in Bhavnagar, have been inhabited by the lion. The expansion has been more pronounced after 1990, as core area of Gir reached its saturation levels. Before 1995,

the eastern limit of the lion distribution range was restricted to Semardi-Virpur-Liliya and Mitiyala. It is not necessary to develop continuous forests in the corridor as it may not be possible in present context. Attempt should be made to develop government and common land as effective corridors. In corridors, patches of the dense forest should be raised and waterholes should be created so that, lions can find shelter and fulfill their basic requirements. The four clusters identified being suitable for lions, can be described as follows:

1. Gir-Girnar WLS and adjoining landscape at the bank of river Shatrunji

This landscape (Figure 6A.4) comprised of Gir, GirnarWLS, Mitiyala forests, Shatrunj river system and Hippavadli zone. The management of the Mitiyala forest and grasslands in Mahua and Savarkundla taluka in Bhavnagar was handed over to the Gir management authority to incorporate it with the Gir lion conservation (Singh 2007). Approximately 10 km. away from the nearest boundary of Gir, Mitiyala Forests is connected with Gir through degraded hills, barrenlands, private grasslands, locally called *vidis*, and cultivated lands in the form of corridors. In extending effective lion habitat from Gir to Shatrunji catchments and Hippavadli zone, management of the low degraded hills and barrenlands around mitiyala is recommended to be integrated with Mitiyala sanctuary (Singh 2007). Connected by small streams, Shatrunj river system is significant track for movement of the lion. Lions extensively use the patches of forests and scrublands from Gir sanctuary to Gadhiya-Krankach and *Prosopis* scrublands in Liliya via Sel river. The lions use these patches as daytime *refugia*. With spill of the population from Gir, these patches in Dhari, Liliya, Khambha, Sabarkundla, and Gariyadhar have become very important to accommodate lions seasonally (Singh 2007). Hippavadli Zone is considered as potential area for lion habitat. Isolated due to distance from Gir, still form an integral part of a large habitat with broken corridor patches. Six principal rivers of Gir viz. Hiran, Dataradi, Shingoda, Machhundri, Ghodavadi and Raval flow in south to Arabian Sea and provide opportunities for migration of lions. Lions have been using some 110 sq. km of coastal forests in Sutrapada, Dhamlej, Kodinar, Una, Jafrabad, and Rajula talukas as their shelter ground dominated by *Prosopis chilensis* and the planted *Casuarina equisetifolia*, and harbours nilgai and wild pig, form a part of the lion's diet, though livestock has remained the major food source for them here (Divyabhanusinh 2005).

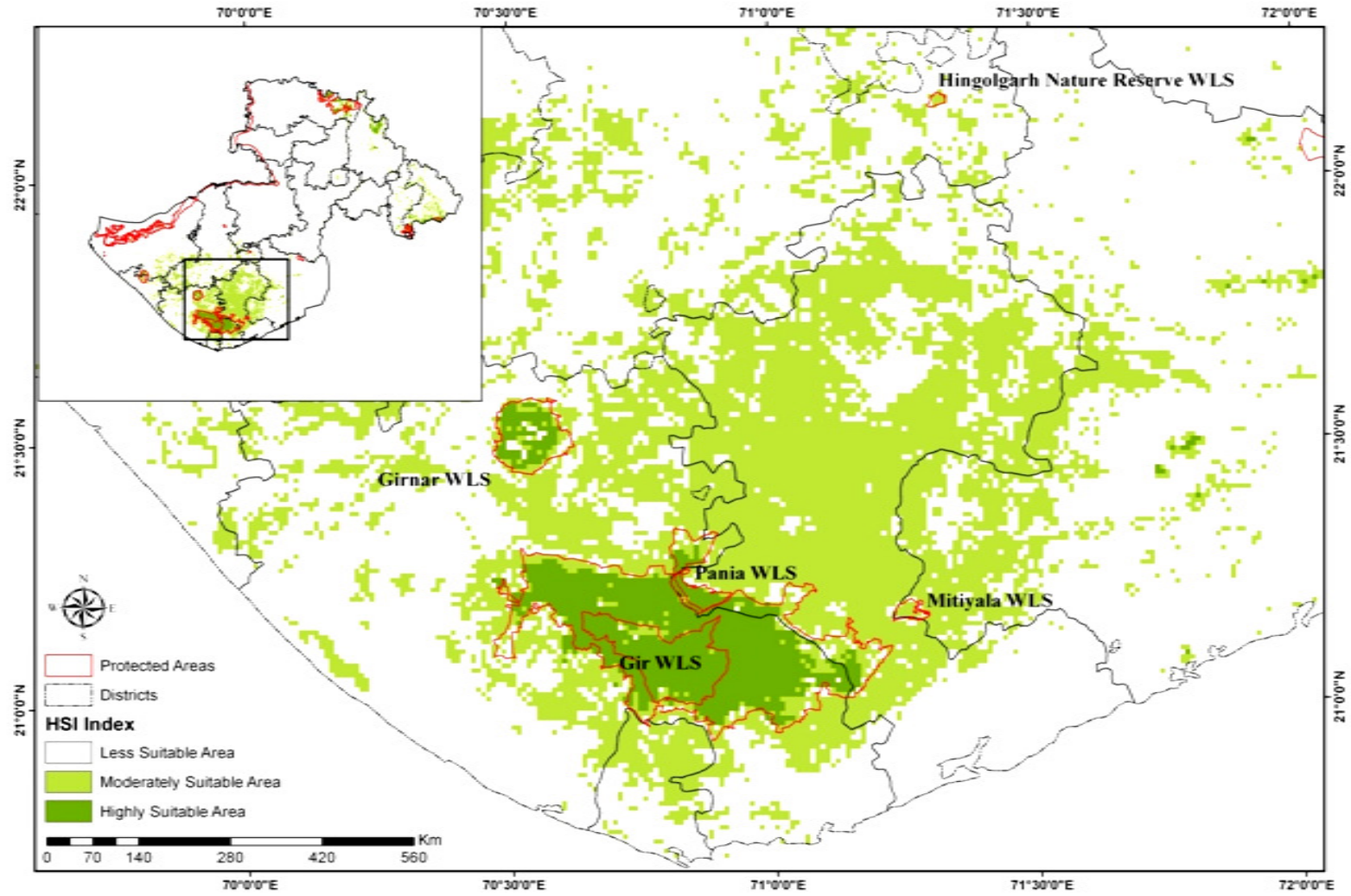


Figure 6A.4: Potential lion habitat at the Gir-Girnar WLS and adjoining landscape at the bank of river Shatrunji

2. Barda-Girnar WLS and adjoining Landscape

This landscape (Figure 6A.5) comprised of Barda Wildlife Sanctuary (192 sq kms) and its adjoining areas. Barda Wildlife Sanctuary has been talked about by the government of Gujarat as a potential area for lion reintroduction. Terrain of Barda is hilly and undulating with an altitude ranging from 79.2m to 617.8m above MSL. The two ephemeral rivers- Bileswary and Joghri drain out water of Barda. *Acacia senegal*, *A. nilotica*, *A. catechu*, *Euphorbia nivulia*, *Zizyphus spp.* are the main species in the area. As per satellite data 46-47% area has forest cover and rest of the land has open to degraded forests (Singh 2007). With the inclusion of adjoining grasslands, wastelands and forests in Porbandar and Jamnagar districts, this area has potential for supporting good lion population (Singh 2007). This area supported lion, sambar, chital and chinkara in the past but they are no more in the Barda Sanctuary. Blue bull and wild pig are main surviving species here. Recently an area of 300 sq.km. in the east of Barda and west of Alech has been identified as a support zone for Barda to develop an alternative home for the lion.

3. Jambughoda-Ratanmahal WLS and adjoining landscape

This landscape is comprised of Jambughoda and Ratanmahal WLS and its adjoining landscape (Figure 6A.6).

3.1 Jambughoda Wild Life Sanctuary: Declared as a sanctuary in 1990, Jambughoda wildlife sanctuary (130.38 sq. km.) located in the Panchmahal district of Central Gujarat. Mainly having forest of teak, bamboos and other miscellaneous species, the area has two water reservoirs- one at Kada and the other at Targol. Leopard is the top predator and the habitat is shared by other animals such as sloth bear, jackal, blue bull, wild boar and four horned antelopes. There are 25 villages (including 5 villages inside the Sanctuary) which are distributed among five forest blocks and two ranges. Local tribals collect minor forest produce from the area for their livelihood. Status of management of the PA is good, but needs to be improved by taking up habitat steps, so that animals can be provisioned with adequate food, water and shelter. Population of herbivores is scanty. Grazing and wood removed by local people and fire in the forest cause damage to the habitat.

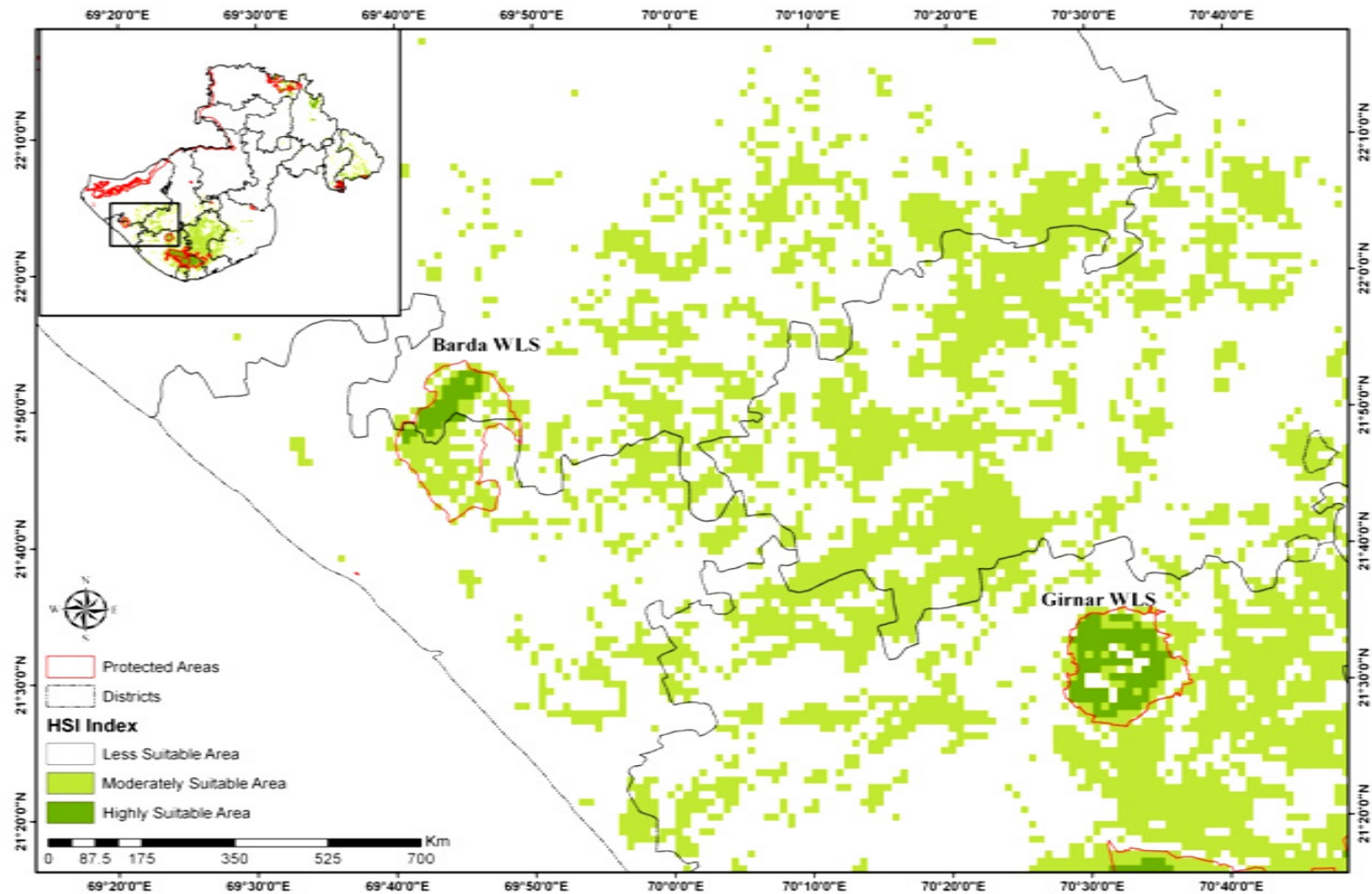


Figure 6A.5: Potential lion habitat at the Barda-Girnar WLS and adjoining Landscape

Ratanmahal Sloth Bear Sanctuary: Declared as sanctuary in 1982 and bordering the state of Rajasthan and Gujarat and, Ratanmahal harbors the maximum population of sloth bears in the entire state. The forests include dry teak forests at the foothills and mixed deciduous forests with dry bamboo brakes on the periphery. The forests of Ratanmahals form the catchment of river Panam- a major river of Central Gujarat. It drains through the districts of Dahod and Panchmahals. A total of 543 species of plants are recorded (119 species of trees, 40 species of shrubs, 238 species of herbs, 48 species of grasses, 87 species of climbers, 2 species of partial parasite and 9 species of orchids).

4. Jessore-Balaram Ambaji WLS and adjoining landscape (Figure 6A.7):

This landscape is comprised of Jessore Sloth Bear Sanctuary, Balaram-Ambaji WLS and adjoining landscape (Figure 6A.7).

Jessore Sloth Bear Sanctuary: Located in the Aravalli hills in North Gujarat, bordering Rajasthan, the forests belong to the southern dry mixed deciduous and desert thorn forests types. It is a home to the endangered sloth bear. Declared as a wildlife sanctuary in 1978, the area has a great ecological significance as it acts as a buffer between the desert eco-system and the dry deciduous type of forest ecosystem. The flagship species of the area is Sloth bear. The top carnivore, leopard cohabits the area with other vertebrates. *Prosopis* invasion occurring almost in pure composition in certain areas and economy of the people is mainly agrarian.

Balaram-Ambaji Wild Life Sanctuary: Declared as sanctuary in 1989, this area is characteristically rich in floral diversity, particularly medicinal plants. It has numerous floral and faunal species of global conservation significance. The rare animals include sloth bear, striped hyena, leopard, bluebull, porcupine, fox, small Indian civet, Indian pangolin and a number of reptiles including venomous and non-venomous snakes, monitor lizards, star tortoise etc. The area falls in the catchment of two rivers- Banas and Sabarmati. The forests of the sanctuary play an important role in conserving the depleting eco-system of Aravalis and in controlling the southward expansion of Thar desert.

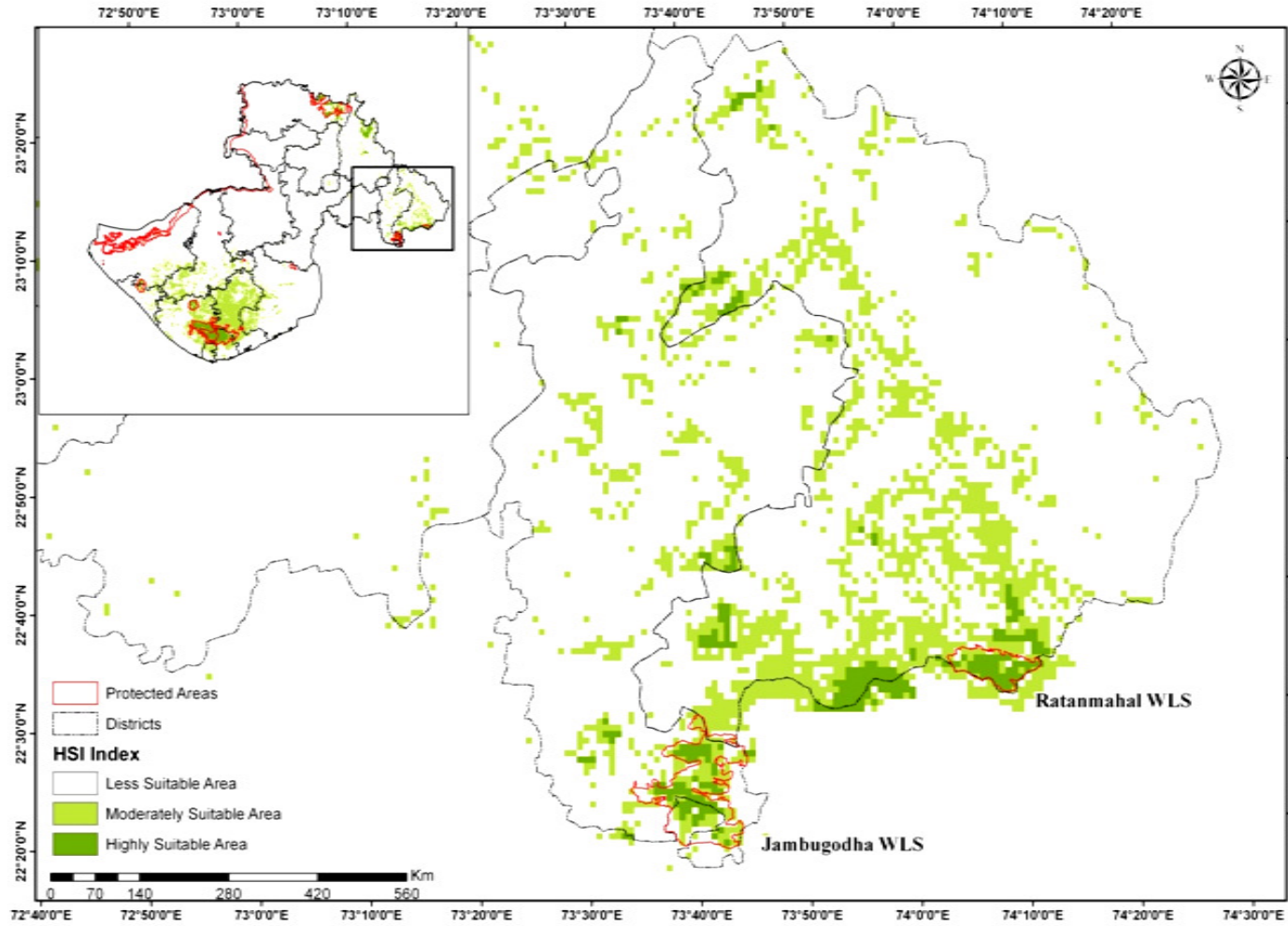


Figure 6A.6: Potential lion habitat at the Jambugodha-Ratanmahal WLS and adjoining landscape

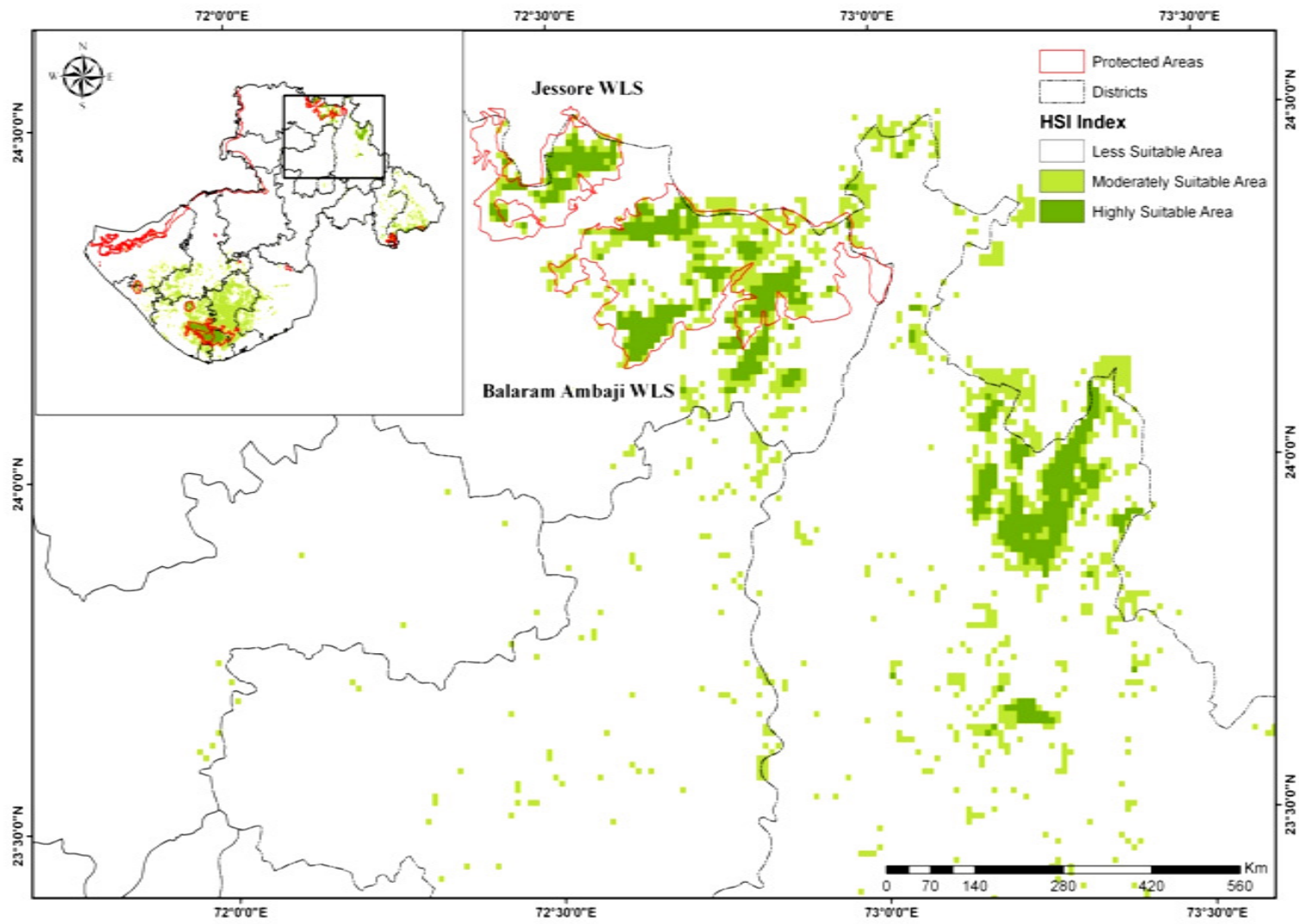


Figure 6A.7: Potential lion habitat at the Jessore-Balaram Ambaji WLS and adjoining landscape

This study provided the first detailed map of Asiatic lions habitat suitability covering major part of the landscape of Saurashtra, Gujarat. This landscape plays a crucial role in the long-term survival of the sole population of Asiatic lions throughout Asia. These data provided important information towards the future conservation of the species by identifying crucial habitat patches in a human dominated landscape. Overall the ENFA approach has permitted us to use the large amount of information collected over the years on lion's spatial distribution in the Saurashtra landscape to discover habitat use of this species, and the potential of this landscape in supporting the increasing lion population. The output of the model identified habitat areas that are highly fragmented, potential movement corridors; those are constrained by roads or other factors, and areas where particular management actions may benefit multiple species. They can therefore help prioritizing management interventions, as well as identifying areas where certain management actions should be avoided entirely (e.g., road-building, mining in a critical connectivity area).

Lions were principally associated to area of moderate dense forest patches and they showed tendency in avoiding zones of permanent human activities (settlements, intensive agricultural areas). However, distance to roads – which reveal the human impact in remote areas – did not come out as an important variable in the model, suggesting that lions tolerate human presence to some extent. This model allowed predicting the extension, distribution and fragmentation of suitable lion habitat. Although not suitable for all fine-scale site planning tasks, this result can be useful for site-planning purposes if used carefully in conjunction with field reconnaissance, expert consultation, and other decision-support maps and models. The immediate action that is required is the upgradation of the forests, thickets and grasslands now occupied by lions, and their protection by making sanctuaries of these areas. Equally significant is the protection of the "corridors" used by lions to migrate out. In these, particularly between Gir forest and Girnar - forest patches chosen by lions, their passage is often through open and cultivated lands.

There is relentless and increasing human pressure in this landscape making it difficult to leave them in the present state of development. Our modeling approach can assist in identifying conservation priorities in a spatial context integrating into a decision-theory framework. Successfully communicating our understanding of ecological

impacts to policy and decision makers is crucial in this process. The habitat models served as a valuable tool in predicting where large carnivore populations may ultimately exist in the study region if recolonization by lions becomes more widespread. This model was intended to project where viable breeding populations may establish in the future. Understanding where breeding populations of large carnivores may become reestablished through an assessment of habitat suitability will prepare wildlife managers for the inevitable impacts of these carnivores on the ecosystems in which they live. It will also allow managers to address potential human-carnivore conflicts. Last but not the least; public attitudes toward this large carnivore may ultimately determine the species' fate in this landscape.

Chapter 6 B

Managing the agro-pastoral landscape for Asiatic Lions (*Panthera leo persica*): Habitat suitability modeling and corridor identification between Gir and Girnar wildlife sanctuary

6B.1. Introduction

Habitat fragmentation is widely recognized as one of the primary threats to biological diversity (Wilcox 1980, Wilcove et al. 1986, Meffe and Carroll 1997). Urbanization is the main cause of habitat fragmentation, threatening carnivore persistence around the world (Ferrerias et al. 1992). The degree of anthropogenic alteration fragmented the landscape within the present distribution range of Asiatic lion (*Panthera leo persica*). The area of Gir forest, spread over an area of 5000 km², two centuries ago, gradually shrank with steady deforestation. The earlier existent corridors connecting hills, forests and the lion's habitat in the form of protected and un-classed forests have almost been transformed to the agriculture and human settlements (Johnsingh et al. 1998). Highways and expressways have been widened and are now filled by speeding vehicles, making it difficult for animals to safely cross them. A large track of grazing pastures and wastelands, which were used by both wildlife and livestock, have been transformed to other uses.

Girnar WLS, one of few small breeding units, recolonized by dispersing lions from Gir protected area in mid 1980s (Singh 2007), is bordered on three sides by townships of Junagadh, Bilkha and Bhesan. The small lion population of Girnar exists as a separate but connected with the lion population of Gir Protected area. Occasional exchange of lions between the two populations is crucial for sustaining long-term demographic and genetic viability of the Girnar lion population. Functional connectivity, defined as the degree to which inter-fragment landscape facilitates or impedes the movement of individuals between patches, encapsulate the combined effects of matrix structure and its influence on the movement of a particular species (Taylor et al. 1993, Tischendorf and Fahrig 2000). Understanding how this spatial

heterogeneity impacts lion's dispersal, and hence persistence of populations in this fragmented landscape, aid in the identification and subsequent preservation of critical habitat.

The increasing lion population of Girnar (Banerjee et al. 2010) uses definite areas within the sanctuary avoiding of rugged terrain of Girnar (unpublished radio telemetry data). In addition the sacred nature of this forest, dotted with many hindu and jain temples and visited by hundreds of thousands of pilgrims each year, reduce the amount of available habitat for this increasing lion population. The approximate carrying capacity of Girnar as 25 individuals (a conservative estimate based on lion densities in Gir Protected Area; Jhala et al. 1999, 2004), natural barriers, anthropogenic disturbances within the sanctuary, the traditional land use changes outside protected area in this region (Banerjee et al. 2010), claimed a study identifying functional corridors joining Gir and Girnar protected area and recommending management interventions, safeguarding the corridor.

With the advent of Global Positioning System (GPS) technology it was possible to gain new insights into lion dispersal and connectivity among disjoint lion populations. Analysis of detailed GPS revealed dispersal paths provided information on how animal perceives and moves through this landscape. We used data from radio collared lions (n=9) to develop habitat suitability model (Hirzel et al. 2003). Output of habitat suitability was then used in PATHMATRIX (Ray 2005) to delineate the least cost pathway between Gir PA and Girnar to spatially define the corridor habitat between the two lion populations.

6B.2. Study Area

This landscape includes the revenue lands of approximately 90 villages (figure 6B.1) , consists of broken topography, small drainage systems, Government and privately owned wastelands, fallow lands and is predominantly agricultural with seasonal crops and fruit (mango) orchards (Figure 6B.1). This area is exhibiting rapid growth, yet there are still many natural areas interspersed among human development. Climate is generally dry with two main seasons' viz. summer and winter. The monsoon which

on some terrestrial vertebrates (Dettki et al. 2003, Reutter et al. 2003, Brotons et al. 2004). How ENFA works, is described in earlier chapter 6A, section 6A.6. We superimposed a 100m X 100m grid on the study area. The environmental variables in GIS format all pertain to the same geographic area, were extracted to grids to be useful in modeling method to predict environmental suitability for the species as a function of the given environmental variables. The grid size was considered suitable to obtain and understand habitat choice by lions and being sufficiently large to enable us to use freely available data sets on environmental variables without compromising on spatial resolution. We extracted data of different eco-geographical variables such as Digital Elevation Model (DEM), Normalized Differential Vegetation Index (NDVI), Euclidean distance to available water sources, considering both the major rivers and dry river channels and Euclidean distance to major roadways as a disturbance factor to animal movement to model lion habitat use pattern and identifying crucial areas for corridor.

The output of ENFA was then used to calculate the resistance matrix (1-Habitat Suitability value) and PATHMATRIX (Ray 2005) was used to delineate the most feasible passageway that lions use for movement in between two protected areas and could be designated as functional corridor for long term sustainability of biological linkage between these two protected areas. This algorithm computes a least-cost path between a source population and a target population by using a friction (or resistance) layer. The friction layer is a raster map where each cell (landscape unit) expresses the relative difficulty (or cost) of moving through that cell for a given species. A least-cost path minimizes the sum of frictions of all cells along the path, and this sum is the least-cost distance.

6B.4. Results

Results of ENFA suggest that, lions have exhibited preference to dry river channels as their passage ways (Figure 6B.2). The areas with higher elevation were not preferred by lions. The marginality factor was explained by euclidean distance to rivers and dry river channels (0.61) and the specialization factor 1 (9%) was explained by DEM (0.71), specialization factor 2 (5%) was explained by euclidean distance to major

6B.5. Discussion

Conservation biology theory recommends that building of linkage structures for wildlife between isolated habitat patches may increase or at least maintain levels of interpatch dispersal, thus sustaining gene flow and supporting population viability of target species (Kozakiewicz 1993, Forman and Alexander 1998). Gir protected area is at its carrying capacity for lions (Singh 1995), constructive changes in the environment may improve this figure slightly, but for the sustenance of increasing lion population, habitat upgradation in new areas naturally colonized by the lions and preserving the linkages in between these areas are essential (Singh 2007). Variation in the landscape can influence the movement of individuals through the matrix and therefore the prospect of habitat patch occupancy (Wiens 1997, Ricketts 2001). The valley portion and foot hills of Girnar are used by lion as major part of Girnar is not suitable for them (unpublished radio collared data). The animals use peripheral villages which are integral part of the habitat as the lions prey on livestock at night in these bordering villages (Singh 2007).

The human dominated landscape between Gir and Girnar is used by lions primarily for movement purpose. Distance of Girnar from the nearest boundary of the Gir sanctuary (W) Division is about 22 km. Lions use streams, barren land, *gauchar*, agricultural field and scrubland for movement and also for shelter when it becomes necessary. Though few patches exist, where there is a possibility of finding resident animals (unpublished data) but very small such patches, surrounded by human settlements and timely occupancy by animals, don't show potential as strong lion hold in near future, until and unless they get protection.

The provisioning of landscape corridors has often been recommended as mitigation for the isolation effect of fragmentation (Wilson and Wills 1975, Saunders and Hobbs 1991). Several studies supported that the presence of a corridor improves population density relative to isolated patches (MacClintock et al. 1977, Fahrig and Merriam 1985, La Polla and Barrett 1993, Dunning et al. 1995, Haddad and Baum 1999). An understanding of how species move through fragmented landscapes, by means of elements such as corridors and stepping stones, is vital for species management at a landscape-scale (Brooker and Brooker 2002, Bowne and Bowers 2004).

The behaviour of species moving through the corridor within agricultural landscapes is expected to be influenced by the nature of the matrix, the type and spatial distribution of adjacent habitats, season, and interaction between same and other species. Wooded and shrubby vegetation together with the corridor functioned as day *refugia* as well as passage ways for lions, signifying their importance to lion's movement in this landscape. Regular use of mango orchards and *gauchar* (pasture land) indicated that such patches are important for movement in fragmented landscape and serve as a link to other forested patches. Agricultural intensification, and the associated loss due to explosion of urbanization, has resulted in riverine forest patches acting an important contributor to biodiversity conservation in this agricultural landscape (Jhala et al. 2012). These biological corridors may alleviate negative impacts of habitat fragmentation by permitting movement between large areas of suitable habitat for the species. Use of the same riverine patches by animals; for it's to and fro movement between two protected areas confirmed the importance of these patches as functional corridor. Some of the cultivated lands which are not suitable for agriculture may be reverted back under pasture or thorn forest for multiple uses. Barren lands, riverbeds, and some of the private lands may be acquired or purchased and the corridor area should be afforested by suitable species to allow free movement of lions between the two areas.

Analyses of land use and cover change processes within the Gir-Girnar corridors suggested that riverine patches are vanishing at an alarming rate and the remaining patches are likely to be converted to agriculture. Sustainable management of these forest fragments huddled along the riparian corridors of tributaries of the main river Ozat and other small rivers is urgent with a focus on biodiversity. The Fragmentation Hypothesis (1975) however, further predicts that as existing habitat decreases, the spatial arrangement of the remaining habitat become gradually important due to an exponential rise in the distance between patches (Saunders et al. 1991). Individual patch size, the degree of isolation, the impact of edge effects and interactions between patch size and isolation therefore play an increasing role, and so population sizes decline more than that predicted by the Random Sample Hypothesis (Andrén 1994).

The gradually fragmenting forest cover in this agro-pastoral landscape may have substantial impacts on the functionality of the remaining patches as they are ever more exposed to influences arising from outside their boundaries. The possible consequences of this modification in landscape structure comprise, but are not restricted to, a decline of biodiversity (Fahrig 2003), an increased exposure to the dispersal of non-native species (Brothers and Spingarn 1992), alterations in plant–animal interactions (Cadenasso and Pickett 2000), changes to physical factors (Cadenasso et al. 1997), exposure to increased atmospheric deposition (Beier and Gundersen 1989), and degradation of water bodies biotic conditions (Shandas and Alberti 2009).

Within the area, identified as corridors, there are four state highways cutting the corridor at several points, detrimental to the animal's use of the corridor. Widely accepted as a source of habitat fragmentation (Reed et al. 1996, Fleury and Brown 1997, Forman and Alexander 1998), roads have a broad range of ecological effects on wildlife populations, both direct and indirect (Forman and Alexander 1998, Trombulak and Frissel 2000, Iuell et al. 2003). They facilitate animal–vehicle collisions; decrease reproductive success (e.g., litter failure; Bjurlin and Cypher 2003); cause disturbance and pollution (Iuell et al. 2003); constrain movement and distribution of species (Clarke et al. 1998, Alexander and Waters 2000, Lod'e 2000, Yale Conrey and Mills 2001, Proctor 2003, McDonald and St.Clair 2004, Shepard et al. 2008); decrease colonization rate; increase extinction rate (Mader 1984); and change population density (Bjurlin and Cypher 2003), biodiversity (Gutzwiller and Barrow 2003, Chen and Roberts 2008), and prey availability (Bjurlin and Cypher 2003). The indication that roads cause detrimental effects on habitat persistence and animal life-history traits, ranging movements, and density variations appears overwhelming. Though, no study is yet conceptualized on assessing the effects of these highways on the movement of lions, however, few nodes (n=12) have been identified, serving as bottlenecks (passageways across highways) and incorporation of crossing structures for wildlife is recommended in road-construction and -improvement projects.

As a sensible attempt, the corridor area is declared as “ECOSENSITIVE” (under The Environment (Protection) Act 1986, Government of India by Gujarat Govt. with restrictions on high impact industry and mining activities. Infrastructure development needs to be kept at minimal and when essential should be mitigated with appropriate investments e.g. highways, roads, and railways need appropriately designed underpasses for wildlife, with funneling fences, sound and light barriers at strategic passage ways. Private property needs to remain permeable to wildlife movement, thus high walls, electric fences need to be restricted or designed for permeability of target species. Urban sprawl is difficult to regulate and can reduce the viability of the corridor significantly. The least cost corridor habitat currently passes through areas that have the least resistance to lion movement, declaring this as ecosensitive and by providing appropriate mitigation and incentives to local communities should assist in maintaining its permeability for future generations ensuring the metapopulation structure of lions in this landscape.

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