Tigers, prey loss and deforestation patterns in Sumatra

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Abstract

A fundamental requirement for conserving a large carnivore such as the tiger is understanding its response to the principal threats of habitat loss and poaching. This thesis investigates the influence of these threats on one of the largest tiger populations on Sumatra, located in the Kerinci Seblat (KS) region, Indonesia.

Interview surveys with a pioneer farming community living adjacent to KS National Park (NP) showed that most farmers had positive attitudes towards tigers and their conservation. Farmers thought that wildlife crop raiding was the greatest limitation to agricultural success and that deforestation would adversely affect tigers, tiger prey and themselves. An analysis of deforestation (forest converted to agriculture) in the KS region between 1995 and 2001 showed a mean deforestation rate of 0.96%/yr. Deforestation was correlated with lower elevations, closer proximity to settlements and public roads, flatter terrain and being outside of KSNP. To mitigate this deforestation, KSNP became the focus of an Integrated Conservation and Development Project (ICDP), but a further analysis showed there was no difference in deforestation rates between ICDP and non-ICDP villages. In villages bordering KSNP, higher rates of conversion occurred in villages with greater occupancy by a logging concession (HPH) and in flatter areas. This suggests that addressing land insecurity created by the designation of customary forest as a HPH was more relevant to lessening deforestation.

In farmland bordering KSNP, most farmers (80.2%) claimed that wild boar were the most destructive crop pests, but this did not corroborate with actual results because although wild boar raided most frequently (76.4%) pig-tailed macaque caused the most damage (73.1%). Investigating the factors that determined tiger prey distribution in the KS region showed a negative association with roads. However, snare trap location was more likely to be found close to logging roads and in richer villages, thereby challenging the rationale of the KS-ICDP that sought link biodiversity conservation with village development. Tiger distribution was also found to be negatively associated with distance to roads. Using this factor to construct a habitat suitability model identified three subpopulations of 98, 20 and 15 tigers in KSNP. A population viability analysis supported law enforcement activities that kept poaching below 3 tigers/yr in the smaller areas and maintained connectivity with the larger subpopulation.

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The greatest pride and pleasure is undoubtedly reserved for my parents: by knowing they were always there proved to be the most supportive force throughout my Ph.D.

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GENERAL INTRODUCTION

Sumatran tiger on a lowland forest ridge trail inside KSNP

1.1 INTRODUCTION

Protected areas (PAs) have two main roles: they should represent the biodiversity of a region and should remove this biodiversity from the factors that threaten it (Margules and Pressey 2000). In reality many PAs are under-funded and this makes their protection difficult. So when wildlife, which does not recognize the boundaries of a PA, ventures outside of a reserve its protection is even less certain. In the developing countries the situation is compounded further because PAs are being isolated by human settlements, agricultural development and the active elimination of wildlife on these lands (Newmark 1996). In this thesis I explore how deforestation patterns, and more pertinently edge effects, caused by agricultural expansion impact upon the tiger and tiger prey species in Sumatra. The main interest of this thesis is identifying the locations of tiger populations in Kerinci Seblat National Park (KSNP), west-central Sumatra, determining their viability under different management scenarios, and then exploring the possible ways to mitigate forest habitat loss.

1.2 LARGE SPATIAL SCALE EDGE EFFECTS

Tropical rainforests are being converted and degraded at an increasing rate. This has a direct impact upon their biodiversity (Whitmore and Sayer 1992). A topic that is of considerable concern in tropical conservation is large spatial scale edge effects (Laurance 2000), known as LSSEE. The process of LSSEE initially causes abiotic change which has direct biological effects which in turn has indirect biological effects (Murcia 1995). The abiotic effects may be a change in the physical forest environment. The direct biological effects may be a change in forest fauna and flora abundance, diversity and biomass. Indirect biological effects may be a change in predator-prey interactions, herbivory and seed dispersal. Large carnivores are particularly susceptible to LSSEE because of their trophic status as top predators.

1.3 LARGE CARNIVORES IN HUMAN-ALTERED LANDSCAPES

Large carnivores occur at naturally low densities and have large ranges. This makes them vulnerable to direct threats such as poaching and habitat loss (Lande 1988, Caughley 1994). Large carnivores are also vulnerable to indirect threats, such as the poaching of their prey, because their abundance is related to the abundance of their prey (Carbone and Gittleman 2002). This makes large carnivores sensitive to edge effects and human activity (Woodroffe and Ginsberg, 1998, Purvis et al. 2000, Woodroffe 2000, Brashares et al. 2001, Crooks 2002). Most large carnivore populations now live in close proximity to, or are embedded in, human-altered landscapes. This leads to competition over resources, such as space and food, and typically causes conflict. This does not bode well for the continued existence of large carnivores, which have disappeared over vast areas or have been reduced to remnant populations in their former ranges as a result of anthropogenic threats (Ginsberg and MacDonald 1990, Nowell and Jackson 1996, Ginsberg 2001). In circumstances where populations diminish in size, they become more susceptible to stochastic events such as disease, inbreeding, that further drives these populations towards extinction (Soulé 1980, Caughley 1994). If populations of large carnivore are to survive in the long term future, then they will require conservation intervention and prudent management.

Large carnivores are focal species that can complement ecosystem-level conservation planning by revealing thresholds in habitat area and landscape connectivity because their distributional patterns are associated with regional-scale population processes (Carroll et al. 2001, Schadt et al. 2002). This makes a large carnivore such as the tiger a suitable focal species. Nearly all tigers inhabit human-altered landscapes but human-related threats have already caused the extirpation of the Balinese tiger in the 1940s and the Javan tiger in the 1980s (Seidensticker et al. 1987). The continued existence of the tiger therefore presents a curious paradox because these species will only survive in these landscapes in the future.

1.4 TIGERS

The tiger, *Panthera tigris*, once had one of the widest distributions of all the felids. This testified to an ability to adapt and survive over a wide range of climates, habitats, and prey assemblages (Mazák 1981, 1996). At the turn of the 20th century there were probably more than 100,000 tigers globally, ranging from Turkey to Bali (Figure 1.1). Yet at the turn of the 21st century the global tiger population had declined by about 95%. Expanding human populations put increased pressure on tiger habitat, tiger prey, and tigers themselves. The acceleration of these pressures over the past 25 years highlights the need to protect tiger populations and for sound scientific research to guide tiger conservation.

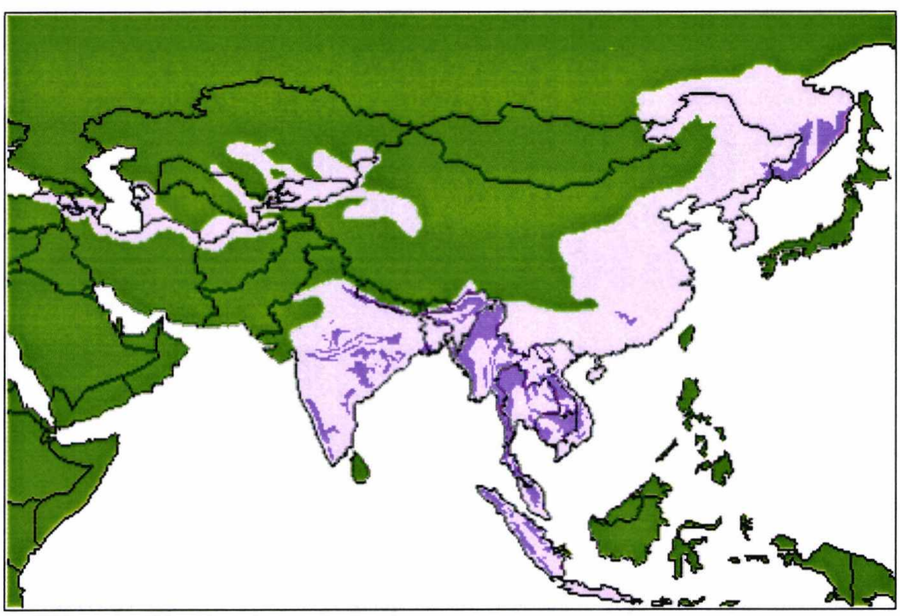


Figure 1.1: Distribution of tigers across Asia: 1900 and present (after Mazák 1979 and Wikramanayake et al. 1998)

1.4.1 Tiger research

Tiger research in the 1980s and 1990s focussed on their ecological requirements (Sunquist 1981, Smith and McDougal 1991, Smith 1993, Karanth 1995, Karanth and Sunquist 1995, Karanth and Nichols 1998, Smith et al. 1998). This provided valuable insights into the resilience of tigers and important biological information that would be later used for managing wild tiger populations. Much of this field research was conducted in India and Nepal, countries with strong links between tigers and religion

(Jackson 1999). Unfortunately rigorous scientific tiger population census techniques were not followed during this period because they were not widely known, or understood, or were subordinate to more rapid survey methods. As a result, the Indian tiger population censuses were carried out inconsistently. This may have hindered tiger conservation. The large amounts of money and time spent on fieldwork gave the impression that substantial investment of conservation resources was synonymous with adequate protection. In reality, field conservation efforts were poorly focussed and had little effect in even monitoring tiger population trends (Karanth 1999). Since then, unambiguous and well-designed protocols for studying tiger population dynamics have been developed (Karanth and Nichols 2000) and are widely practiced across the tiger range states (Franklin et al. 1999, Karanth and Nichols 1998, Kawanishi 2002).

These techniques are now being used to develop a better understanding of how tigers are affected by human activities. Contemporary research now focuses on tiger ecology in spatio-temporal human-altered landscapes (Smith et al. 1998, Miquelle et al. 1999[a]). This new *modus operandi* has led to the development of an ecology-based framework for priority setting in tiger conservation (Wikramanayake et al. 1998).

1.4.2 Tiger Conservation Units

Tiger Conservation Units (TCUs) were developed to encompass all areas containing tigers or thought to contain tigers. The TCUs were categorized according to their bioregion and predominant vegetation type. Each TCU was scored and ranked on the basis of three salient characteristics for wild tigers: habitat integrity, poaching pressure, and population status. The sum of these three factors formed a hierarchical structure and determined their priority status:

Level I – offering the highest probability of persistence of tiger populations over the long term because of their large blocks of suitable habitat for tiger and prey, and low to moderate poaching pressure on tiger and prey.

Level II – offering a medium probability of persistence of tiger populations over the long term. These units have moderate-large blocks of suitable habitat, moderate to high poaching pressure on tiger and prey, but with potential for anti-poaching measures.

Level III – offering a low probability of persistence of tiger populations over the long term because of their small size, isolation, and fragmented habitat. With intensive management they may harbour small tiger populations, but they suffer from high poaching that endangers conservation efforts.

S – requiring an immediate survey due to lack of data.

This research framework had the salient outcome of allowing objective identification of key tiger conservation areas and resource allocation. In the southeast Asia bioregion, from south of the Isthmus of Kra to the southern tip of Sumatra, four TCUs were identified as Level I, six as Level II, 17 as Level III, and four requiring an immediate survey. This framework highlighted the important position of Indonesia in tiger conservation: it has three out of five Level I TCUs in the southeast Asia bioregion. These include Gunung Leuser NP and KSNP, which represent two of the largest PAs in Asia and are strongholds for the Sumatran tiger, *P. t. sumatrae*.

1.5 SUMATRAN TIGERS

1.5.1 Important tiger areas in Sumatra

Sumatra has 29 protected areas, 26 of which showed definite evidence of tiger presence in the early 1990s. These 26 PAs cover 45,641 km^2 or 9.63% of the island (Ramono and Santiapillai 1994). A Sumatran tiger Population and Habitat Viability Assessment (PHVA) carried out by the Indonesian government and the IUCN/SSC Conservation Breeding Specialist Group estimated that around 400 tigers are dispersed between five core PAs. These were the National Parks of Berbak, Bukit Barisan Selatan, Gunung Leuser, KS, and Way Kambas, with 100 tigers occurring outside of these core areas (Tilson et al. 1994) (Table 1.1, Figure 1.2).

Table 1.1: Key tiger habitats in Sumatra

Protected Area	IUCN 1997 Classification[a]	TCU Level	Province	Size (km^2)	Altitude (m)	Estimated Tigers[b]
Barisan Selatan	II	I	Lampung	3650	0-1965	68
Berbak	II	II	Jambi	1627	0-20	50
Rimbang	Ia	I[c]	Riau	1460	200-1090	42
Gunung Leuser	II	I	Aceh	7927	0-3420	110
Kerinci Seblat [c]	II	I	Jambi	13680	125-3800	76
Kerumutan	IV	II	Riau	1200	0-0	30
Way Kambas	II	II	Lampung	1300	0-50	20
Outside PAs						~ 100
TOTAL						~ 496

[a] Ia = nature reserve, II = national park, IV = game reserve,

[b] Figures from Sumatran Tiger PHVA (Tilson et al. 1994)

[c] Rimbo Panti included with KSNP for TCU classification

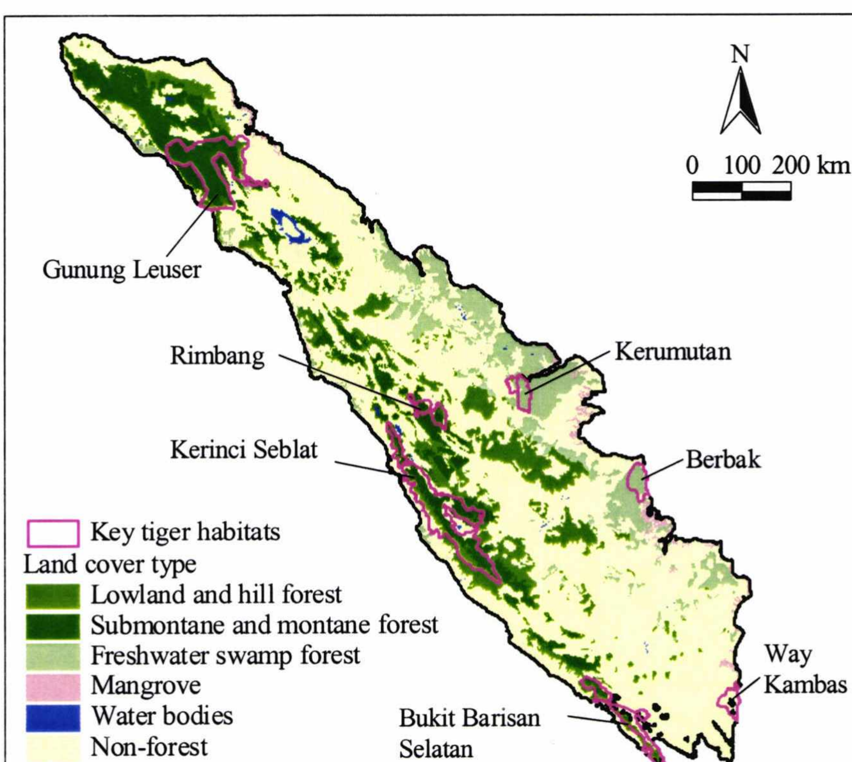


Figure 1.2: Locations of important tiger areas on Sumatra (1 km^2 land cover data adapted from WCMC)

The Sumatran tiger PHVA drew attention to the decline in this subspecies, from a conservative estimate of 1000 tigers during the late 1970s (Borner 1978, P. Jackson pers. comm.) to 800 tigers in 1985 (Santiapillai and Widodo 1985) to a highly fragmented population of around 500 tigers in 1994 (Faust and Tilson 1994), and now considered critically endangered (IUCN 2002). Interestingly, open interview surveys conducted in villages bordering Way Kambas NP recorded many farmers responding that they believed tigers to be more abundant on Sumatra today than 20 years ago (Nyhus et al. 1999). This is unlikely given the imminent threats facing the Sumatran tiger but it cannot be easily disproved as information on tiger population trends and their distribution across Sumatra is limited.

The data compiled for the Sumatran tiger PHVA were inferred from the number of tigers that could be theoretically supported, given the available habitat remaining and using mean tiger home range size data from Gunung Leuser NP and Bengal tigers from Nepal (Sunquist 1981, Faust and Tilson 1994, Griffith 1994). Apart from highlighting the decline of Sumatran tigers and the need for conservation intervention, the PHVA emphasized the paucity of reliable data and the need for more rigorous scientific research, such as monitoring population trends and obtaining basic information on tiger distribution. This led to the creation of the Indonesian Sumatran Tiger Conservation Strategy.

1.5.2 The Indonesian Sumatran Tiger Conservation Strategy

The strategy aimed to 'develop and sustain a conservation programme in Indonesia that will ensure the long-term viability of wild Sumatran tigers in major protected areas of Sumatra, to develop a captive management programme for Sumatran tigers, and to link these *in situ* and *ex situ* conservation activities for the reinforcement and recovery of wild populations'. Through priority setting of conservation effort the Indonesian Sumatran Tiger Conservation Strategy made four hierarchical recommendations:

- Priority 1: secure and protect all remaining tiger populations and their habitat;

- Priority 2: develop conservation management goals and intervention strategies for the remaining wild tiger populations, including demographic and genetic support for most populations;

- Priority 3: develop a captive management programme for the reinforcement and recovery of wild populations; and

- Priority 4: establish a communication and infrastructure network that is responsible for the survival of Sumatran tigers in Indonesia, accountable to PHPA, national and international conservation agencies, NGOs, and the Indonesian public.

Unfortunately the strategy failed to establish or develop these recommendations because of political constraints. Research on Sumatran tigers has only recently begun to address these points, although not through any collective efforts.

1.5.3 Sumatran tiger research

The most important and recent scientific research that has been conducted on the Sumatran tiger are an ecological study (Franklin et al. 1999), and two edge effect studies (Kinnaird et al. 2003, O'Brien et al. 2003).

Franklin et al. (1999) studied tigers in dense secondary forest, mixed forest, and grasslands in Way Kambas NP. They identified 21 tigers over 16 months and recorded a tiger density of 4.3 tigers/100 km^2 and the home ranges of three tigers, a male (116 km^2) and two females (70 and 49 km^2). They concluded that Way Kambas could support 36 adult tigers, which was 80% more than the previous PHVA prediction. The figures from Way Kambas NP are probably higher than those that would be expected from intact or slightly degraded hill and submontane forest, the predominant tiger habitat types on Sumatra. Additional research is therefore required in habitats at higher altitudes.

Kinnaird et al. (2003) calculated that forest cover in Bukit Barisan Selatan NP (BBSNP) had declined by 28% between 1985 and 2000. From this they estimated that

BBSNP could probably support only 40-43 tigers, instead of the 68 tigers predicted by the PHVA. Tigers were found to avoid forest edges as far as 2km into the NP, and a deforestation model that investigated two physical factors predicted that only 20% of core tiger habitat would remain in BBSNP by 2010. Kinnaird et al. (2003) concluded that this could quite possibly lead to the extirpation of tigers from BBSNP. Previous deforestation studies have tended to either focus on assessing the physical factors that explain deforestation, such as proximity to roads or elevation (Cropper et al. 2001, Sader and Joyce 1988, Dirzo and Garcia 1992) or the social factors, such as poverty and capital markets (Barbier 1997).

Again from BBSNP, O'Brien et al. (2003) confirmed that a relationship existed between relative tiger abundance and absolute density (Carbone et al. 2001). Although the study did not consider physical landscape factors it was the first to test the impact of human activity on tiger and prey abundance, for which there was a significant and negative relationship.

Continuing on from these tiger studies in Sumatra, future research should focus on accurately mapping tiger distribution and tiger prey distribution in order to monitor these populations over large areas (Karanth et al. 2003). A comprehensive tiger distribution map does not exist for Sumatra. Research should focus on hill and submontane forest because these are the main tiger habitat types. Research into tiger habitat loss should investigate the interactions of both physical and socio-economic factors with deforestation. Such studies are lacking but necessary because they can provide a better understanding of the deforestation process (Lambin 1997). In the Brazilian Amazon, an analysis including physical and socio-economic factors found that forest closer to highways and in areas with higher rural human population densities was more likely to be cleared (Laurance et al. 2002). This study is particularly relevant to KSNP, because a network of roads and villages that have various socio-economic statuses surrounds it.

1.6 AIMS OF THE STUDY

To help promote sound conservation management of the Sumatran tiger this study has four main aims: to map forest habitat change, to present new information on the factors that influence tiger and tiger prey distribution, to determine how future forest habitat change will impact on tiger prey, and to analyse the possibilities of different strategies to protecting tigers, their prey, and their habitat.

Hence this study specifically seeks to answer the following research questions in the KS region:

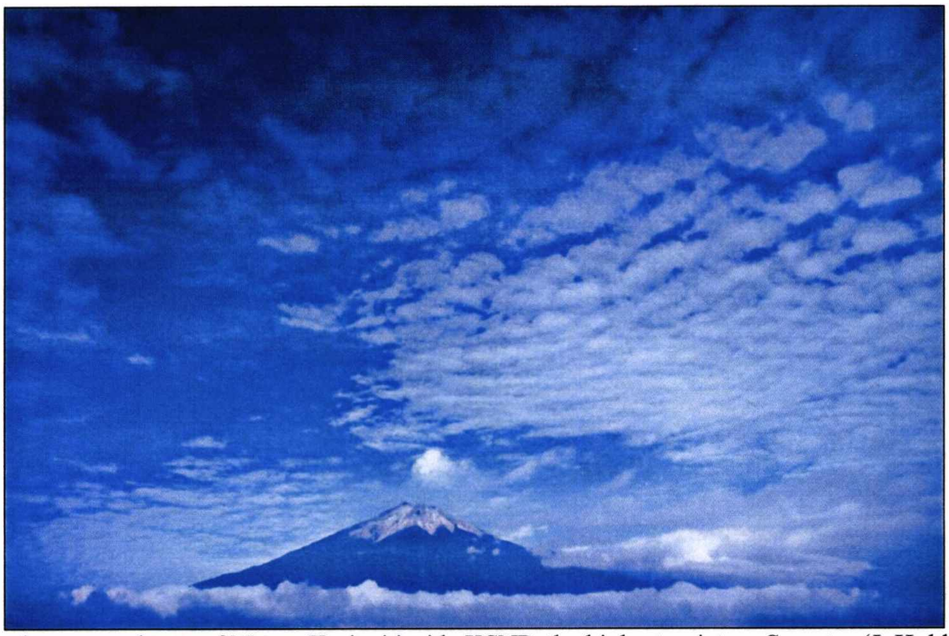
- How is forest distributed between the different forest sectors?
- What factors determine forest loss?
- Where will forest loss occur in the future?
- Did the KS-ICDP prove to be an effective strategy to mitigate forest loss?
- What role did land insecurity play in explaining forest loss?
- Which wildlife species are perceived as the worst crop pest?
- Which wildlife species are observed as the worst pest?
- Which factors determine crop raiding patterns?
- What factors determine tiger prey distribution?
- What factors determine snare trap location?
- What factors determine tiger distribution?
- Where are the areas of core tiger habitat?
- How large are the tiger subpopulations in KSNP?
- How viable are the tiger subpopulations in KSNP?

These questions are covered in sequence in the chapters, and have the overall objective of determining if secure land tenure rights are a viable alternative to protecting tiger and tiger prey habitat, thereby conserving these species in the KS region.

1.7 STRUCTURE OF THE THESIS

Chapter 1 has introduced the main themes of the thesis: the problems with large carnivores living in human-altered landscapes. It has then presented the Sumatran tiger as an ideal candidate for a case study of these issues. **Chapter 2** describes the KS region study site and its history and importance for Sumatran tigers. **Chapter 3** uses household interview surveys to determine the demography and socio-economy of a farming community adjacent to KSNP and whether these factors influence a farmer's attitudes and perceptions towards farming, KSNP, wildlife and conservation. These responses are used to guide the themes of the subsequent chapters. **Chapter 4** gives an introduction to geographic information systems (GIS) and remote sensing (RS) and then describes the methods used to construct the GIS and RS datasets for the KS region. **Chapter 5** uses the spatio-temporal GIS and RS information to map forest distribution and forest change in the various forest sectors in the KS region. **Chapter 6** uses these forest cover data to investigate the factors that determine agricultural conversion of forestland and then uses these factors to predict future forest loss patterns. It then investigates the feasibility of different approaches to mitigate deforestation in the KS region. **Chapter 7** identifies the guarding strategies employed by farmer against crop pests and which crop pests they perceive to be the worst. From the monitoring of actual crop raiding in the farmland this chapter then presents the crop pests that raid most frequently, that cause the most damage and the factors that explain their crop raiding patterns. **Chapter 8** describes how tiger prey base, and snare trap distribution maps were constructed and then used to investigate the factors that determine their abundance. The factors explaining tiger prey base distribution are then used to construct a habitat preference map, from which the impact of edge effects on tiger prey are determined. **Chapter 9** identifies where core tiger habitat is located, how many adult tigers may be resident in each core area, how viable these populations are and how different management scenarios can be used to protect these populations. **Chapter 10** concludes by presenting an overview of the major findings in this thesis and discusses their relevance to tiger conservation.

Chapter 2
A DESCRIPTION OF THE KERINCI SEBLAT REGION



The dormant volcano of Mount Kerinci inside KSNP, the highest point on Sumatra (J. Holden)

2.1 INTRODUCTION

The KSNP runs along the length of the Barisan volcanic mountain chain in the west-central part of the Indonesian island of Sumatra. The national park is found between 1°07'-3°45'S and 100°58'-102°85'E (Figure 2.1). At *c.* 13,300 km^2, KSNP is the largest national park in Sumatra and the second largest on Indonesia, after the *c.* 25,050 km^2 Lorentz NP in Irian Jaya. When originally declared a PA in 1986, KSNP covered some 14,850 km^2. It was later downsized to remove areas of lowland and hill forest containing valuable timber trees. Formally gazetted in 1999, KSNP is still the only officially recognized PA for the whole of Indonesia. In July 2004, KSNP was nominated as a UNESCO World Heritage Site: Tropical Rainforest Heritage of Sumatra site, which also comprised the Gunung Leuser and Bukit Barisan Selatan NPs.

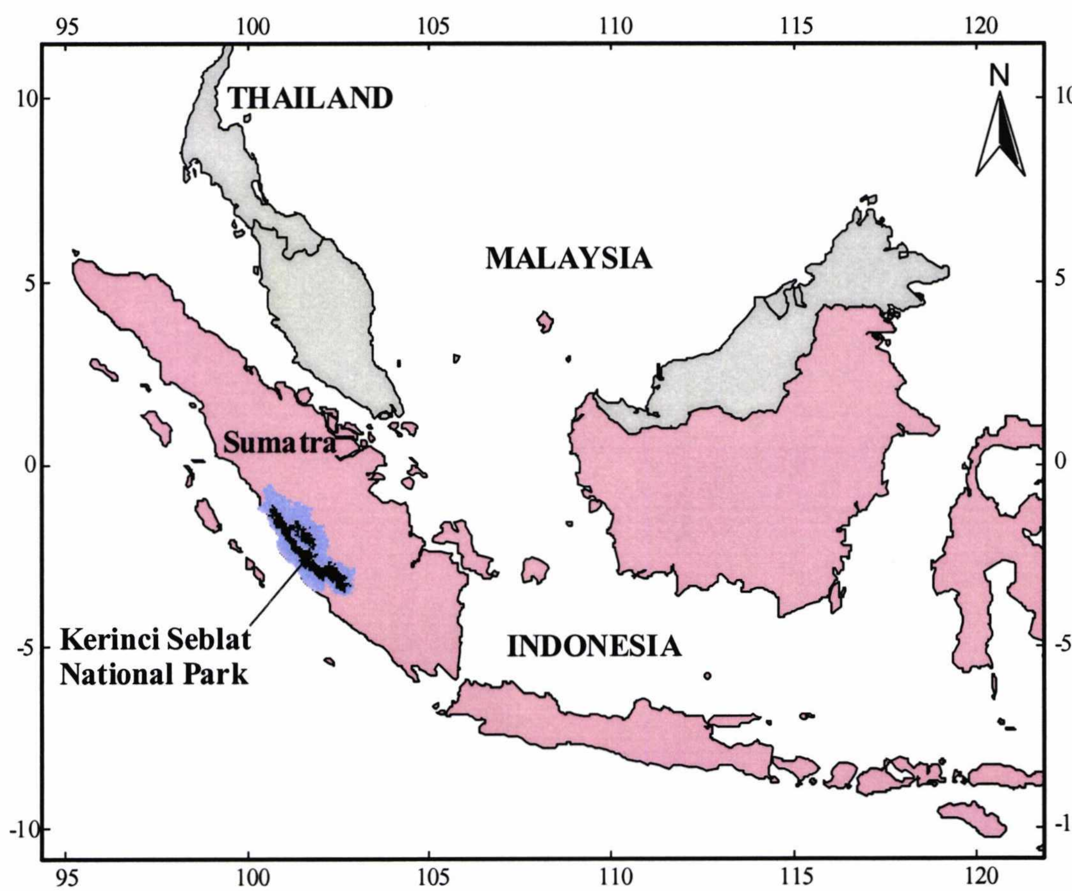


Figure 2.1: The only gazetted NP in Indonesia: the *c.* 13,300 km^2 KSNP, shown in the KS region

Along its 345 km length, KSNP spans 10 districts, which defines the study area and is referred to as the KS region hereafter (Figure 2.2). In the centre of the mountainous park is a populated valley known as the Kerinci enclave. Asphalt trade roads running from the main market town of Sungai Penuh, Jambi, to Tapan, West Sumatra and from Sungai Penuh to Muara Labuh, West Sumatra, split KSNP into three sections. Only the eastern section of the park has been completely isolated though.

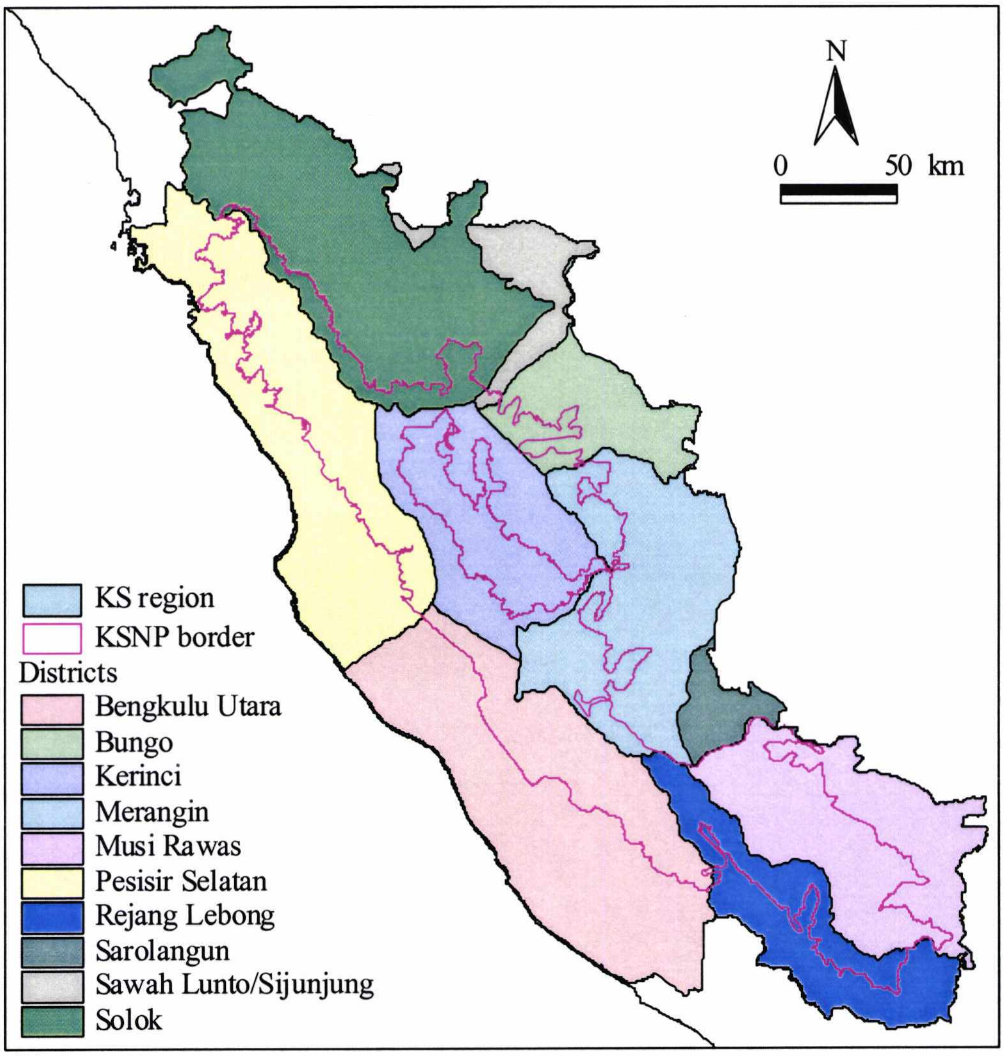


Figure 2.2: The 10 districts of the KS region containing KSNP, west-central Sumatra

This chapter describes the main factors that make KSNP an area of international conservation importance, including for tigers and their prey species. Section 2.2 describes the topography, geology, climate, and hydrology. Section 2.3 describes the rich biota that has resulted in part from these physical conditions. Section 2.4

describes the human involvement in and around KSNP, including culture, and demography. Section 2.5 describes the economic activities and the human-related threats facing KSNP. Section 2.6 describes the conservation value and conservation management in place to reduce these threats.

2.2 PHYSICAL CHARACTERISTICS

The Barisan mountain range is part of the great volcanic arc that extends along the length of Sumatra, Java, and the Lesser Sunda Islands. This range dominates the topography of KSNP and in turn determines the geology, climate, flora, and fauna of the park. A rift valley running from north to south Sumatra divides KSNP into two parallel ranges. The western and eastern ranges have their own distinct physical characteristics and biodiversity (Laumonier 1994). The raised topography in the western range leads towards the west coast of Sumatra and the Indian Ocean, whereas the eastern range gives way to the central plains of Sumatra, in Jambi province. Settled between these two ranges is the Kerinci valley, an area of *c.* 1,450 km^2.

2.2.1 Topography

The mountain ranges consist of undulating terrain oriented from east to west along the spine of Sumatra. These ridges descend to create the numerous rivers for this area. Inside the park the elevation starts at 200 m asl and reaches 3,805 m at the summit of Mount Kerinci, an active volcano and the highest point on Sumatra. The other active volcano in the park is Mount Seblat at 2400 m.

2.2.2 Geology

The geology of the region is very varied (Figure 2.3). The western section of the KS region is predominantly volcanic, whereas the eastern section is largely composed of metamorphic rocks, karst limestone and large granite massifs. Soils of the Kerinci valley mainly comprise fertile alluvial soils. Following the US system of soil taxonomy (USDA 1978) the dominant soil type is the dystropepts (62%). Dystropepts fall under the order of Inceptisols (L. *inceptum* = beginning). The major features of these soils are their lack of characteristics, being embryonic soils with few diagnostic features. The horizons form quickly and mainly from the alteration of parent material.

The formation of the Inceptisols under continuously warm conditions, as in the tropics, gives rise to tropepts. In the KS region the majority of these soils are dendritic and have a moderate to high drainage, given their low base saturation, and are termed dystropepts (*dys* being the formative element of this great group). The nature of Inceptisol productivity is quite variable. They can appear as very fertile soils and have an agricultural importance, due to their moderate to high drainage properties, low drainage usually being a limiting factor in high productivity.

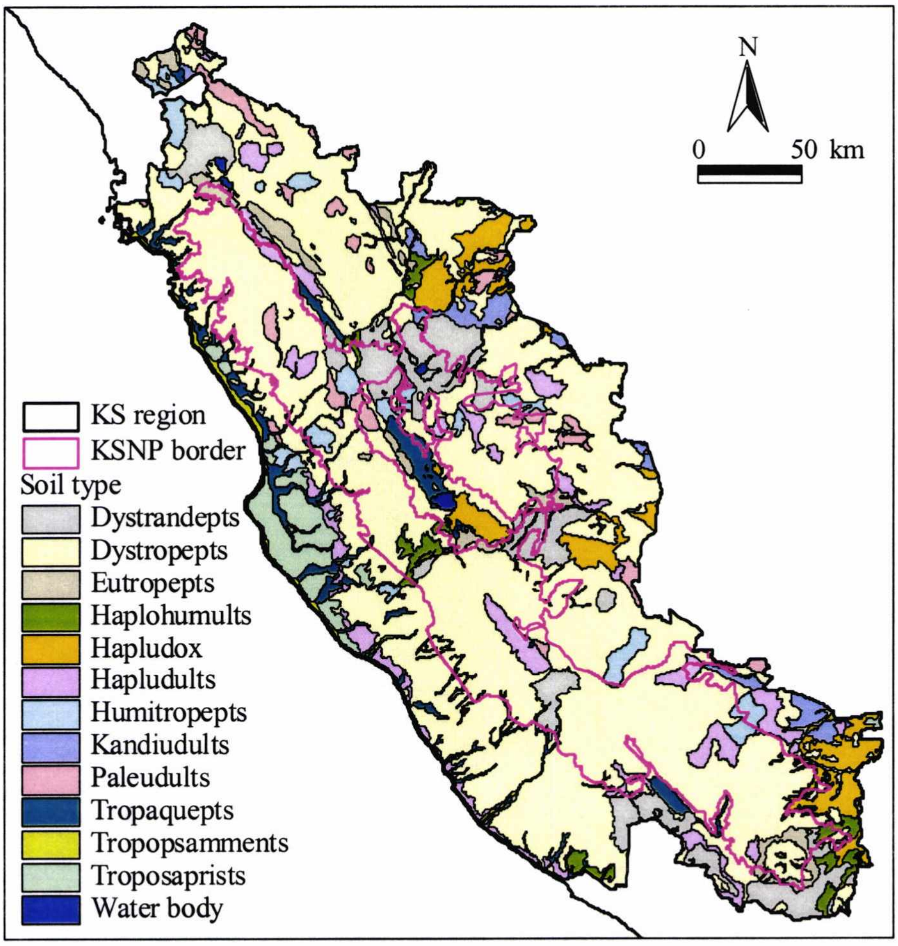


Figure 2.3: Detailed soil map based on U.S. Taxonomy system of soil classification

2.2.3 Climate

The KS region lies within a warm perhumid bioclimate (Whitmore 1984). It is characterized by a variable climatic cycle arising from its physiography and geography. The lowland areas are the hottest, with a mean annual temperature of 30ºC, the Kerinci enclave 23ºC, and then the temperature decreases by about 0.6ºC with every 100 m increase in elevation (De Wulf et al. 1981). In general, KSNP has a dry hot period, from July to October, when average temperatures are 24-30ºC, with daily fluctuations of 2ºC. The temperature falls after October when the rainy period begins, typically from November through to May. The number of rainy days varies from 180 to 220 each year. The eastern slopes and most of the western foothills receive more than 3,000 mm of rain each year, whereas the western coastal areas receive more than 4,000 mm each year. The Kerinci enclave has much less rainfall because the surrounding mountains shelter it. Average annual rainfall in the enclave is 2,300 mm, although the time, onset, duration, and volume of rain are variable (Departemen Kehutanan 1995).

2.2.4 Rivers, lakes and wetlands

The peaks and troughs of the mountainous terrain in the KS region have given rise to a dendritic network of rivers and streams. The rivers originating from the mountains in the western range drain off into the Indian Ocean to the west. In the eastern section, three major catchments drain KSNP, the most important being the easterly, and southeasterly draining streams that eventually form the Sungai Batanghari and Sungai Musi, the largest river in Sumatra (De Wulf et al. 1981).

There are 15 lakes in the KS region, although some have been converted to rice fields. To the north of Mount Kerinci lies the volcanic crater lake of Tujuh (9 km^2) while to the southwest lies Lake Kerinci (42 km^2) in the Kerinci enclave. The volcanic action on Sumatra has created numerous saltlicks within KSNP. These are areas where minerals seep to the surface and they act as an important mineral supplement for many large forest mammals.

2.3 THE FLORA AND FAUNA

Sumatra forms part of the Sundaland region that has been identified as having an incredibly rich biological diversity (Myers et al. 2000).

2.3.1 Flora

In terms of flora the island of Sumatra is one of the most species-rich countries in the world, with 202 out of the 395 known families of seed plant (Williams et al. 1997). Sumatra has over 10,000 types of vascular plants species of which 12 % are endemic (Whitten et al. 1984). There are 13 endemic genera (van Steenis 1987). Within KSNP there are an estimated 2000-3000 vascular plant species, but the flora is still poorly known (WWF and IUCN 1994-1995). Plant diversity in the KS region is extremely high and is on a par with the Brazilian Amazon, often cited as the floral pinnacle of species richness (Gillison et al. 1996). Laumonier (1994) divided the forest types for KSNP according to elevation and aspect because these types had fairly distinct floral composition, which can be consolidated into four broad forest types: lowland forest; hill forest; submontane forest; and montane forest (Table 2.1).

Table 2.1: Details of the elevation and aspect bands of the different forest types in the KS region

Forest type detailed	Elevation (m)	Broad forest type
Eastern lowlands	0-200	Lowland
Western lowlands	0-300	
Eastern lower hills	200-450	Hill
Western hills	300-800	
Eastern upper hills	450-800	
Submontane	800-1400	Submontane
Lower montane	1400-1900	Montane
Montane	1900-2400	
Upper montane	2400-2900	
Tropical subalpine	2900+	

2.3.1.1 Lowland forest

This habitat type occurs from 150 m to 300 m asl in Bengkulu province. This habitat type contains some of the most species rich forests in Sumatra and is also the most threatened habitat type on Indonesia (Holmes 2001). In KSNP, lowland these forests are characterized by an abundance of dipterocarps, including the export quality timber tree of *Shorea atrinervosa* ('Meranti') and *Dipterocarpus* spp. This forest includes useful plants such as *Mangifera torquenda* (wild mango fruit), *Parashorea lucida* (medicinal plant) and *Calamus leoli* (rattan). Rattan has been over-exploited in KSNP, particularly the giant rattan manau (*C. manna*), that also occurs in hill forest (Siebert 1989)

2.3.1.2 Hill forest

This habitat type occurs from 300 m to 800 m and covers 40% of KSNP. Dipterocarps are less abundant than in the lowland forest. The dominant emergent species is from the genus *Hopea*. The good timber quality *Shorea platyclados* is typically found above 500 m. Some of the more charismatic flowers of KSNP occur here: the giant aroid *Amorphophallus titanum*, and the parasitic flowers *Raffleisa hasseltii, R. arnoldi* and *Rhizanthes zipellii*. Hill forest includes the medicinal *Lansium domesticum* and *Aglaia argentea*, and the riverine *Harpullia arborea* ('kayu pacet'), sought after for its unusual veined sapwood. In addition, the aloewood, *Aquilaria* spp. ('gaharu') can still be found in lowland and hill forest, but overharvesting jeopardizes the survival of this species (Soehartono and Newton 2000).

2.3.1.3 Submontane forest

This habitat type occurs from 800 m to 1,400 m. The dominant plant species in submontane forest are from the families Myrtaceae, Clusiaceae, Euphorbiaceae, Fagaceae (beech), Myrtaceae, Clusiaceae, and Moraceae. Like the hill forests, emergent trees can reach up to 50 m here. The good quality export timber trees include *Shorea platyclados* and *Altingia excelsa*. Associated with the emergent trees are numerous hemi-epiphytic and epiphytic figs (*Ficus cf. binnendykii*, *F. disticha* and *F. elastica*). The understorey is notable for its palms such as *Livingstona altissima* and *Areca catechu*, its orchids such as *Asplenium* spp., *Bulbophyllum* spp., *Dendrobium* spp., and *Eria* spp., and its pitcher plants *Nepenthes* spp.

2.3.1.4 Montane forest

This habitat type occurs from 1,400 m to 3,600 m. However, montane forests contain four fairly distinct forest formations: lower montane, montane, upper montane, and tropical subalpine. In lower montane forest (1,400-1,900 m), the canopy is between 25 and 30 m high, decreasing to 15-25 m in montane forest between 1,900 and 2,500 m. Fagaceae is one of the dominant canopy tree families, particularly from the genus *Quercus* and *Lithocarpus*. A remarkable fern belt (*Gleichenia* and *Dicranopteris* sp.), 3-4 m, tall occurs on Mount Kerinci between 2,400 and 2,700 m. From 2,800 m slopes become very steep and many of the moss-covered trees initially grow horizontally before upwards. Above 3000 m is a dense thicket, comprised mainly of Ericaceae and Symplocaceae.

2.3.2 Fauna

The KS region supports a rich and varied fauna. Located to the south of the Lake Toba zoogeographic boundary, the fauna in KSNP differs markedly from that of the similarly sized Gunung Leuser National Park (GLNP) to the north. Unlike GLNP, orangutans are absent from KSNP. However, noteworthy species occurring in KSNP but not present in GLNP include Asian tapirs *Tapirus indicus*, Sumatran rabbits *Nesolagus netscheri*, and western tarsiers *Tarsius bancanus*.

The diversity in KSNP includes over 370 bird species (including 17 of the 20 Sumatran endemics), 85 mammals (including five Sumatran endemics) and over 40 species of anurids (Holden 2002[a,b]; Holden unpublished data). Excepting the bird list, the mammalian and anuridian diversity is probably much higher because this information is derived from preliminary biodiversity surveys. KSNP has been recognized as the last stronghold for Asian tapir on Sumatra (Holden et al. 2003) and previously for its importance for the Sumatran rhino *Dicerohinus sumatrensis* (van Strien 1985). Seven cat species have been recorded in KSNP: marbled cat *Pardofelis marmorata*; flat headed cat *Prionailurus planiceps*; leopard cat *Prionailurus bengalensis*; fishing cat *Felis viverrina*; golden cat *Catopuma temminckii*; clouded leopard *Neofelis nebulosa*; and, Sumatran tiger (Holden 2001, Linkie unpublished data).

2.4 HUMAN INVOLVEMENT

The earliest signs of human presence in the KS region appear to be in the south of the Kerinci Valley *c.* 4,000 years ago. A large migration to the interior only occurred in the 18^{th} century. The population in the Kerinci enclave is estimated at 270,000 people with a rapid annual growth rate of 2.2%/yr (Aumeeruddy 1992). Most of the population is concentrated in the market town of Sungai Penuh.

2.4.1 Culture

Sumatra has a remarkable diversity of cultures, even for Indonesia. In the KS region the main ethnic groups are the Batin, Muko-Muko, Pekal, Redjang, Lembak, Kubu, Minangkabau, Kerinci and Pisang (Lebar 1972). The dominant religion practiced by these groups is Islam (comprising 98.5% in Bengkulu, 98.4% in Jambi, 98% in West Sumatra, and 96% in South Sumatra). Christians make up about 1-1.5% of the population, with the remaining being Hindu or Buddhist.

2.4.2 Human population trends

The population of Sumatra, including the four provinces of West Sumatra, South Sumatra, Bengkulu, and Jambi that span the KS region, has become more urbanized (Figure 2.4). The urbanization process during 1980 and 2000 has been greatest in Bengkulu (20% increase), West Sumatra (16.3%), and Jambi (15.6%), which are all higher than the regional average for Sumatra (14.4%). The trend towards urbanization may reflect the poor economic performance of the agricultural sector and the shift from rural to urban work with its higher employment (Circle-Indonesia 2002). There may also have been faster population growth in the urban areas than in the rural areas.

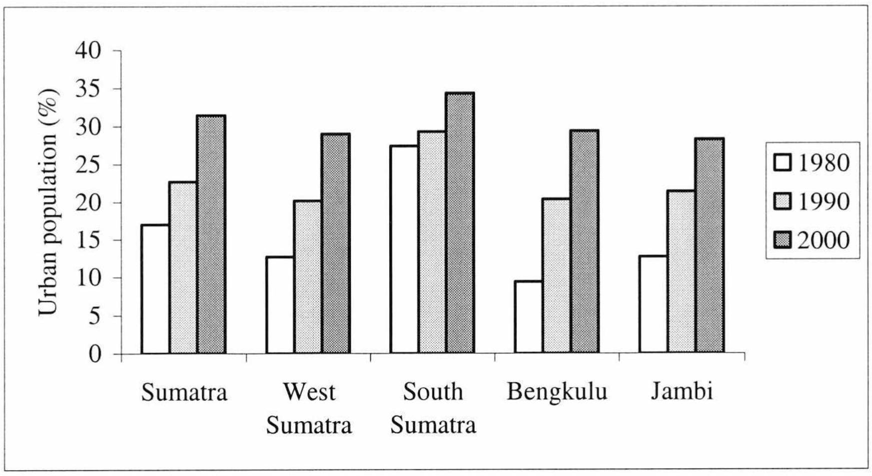


Figure 2.4: The increasing urbanization of Sumatra and the four provinces that span KSNP (based on data from BPS, 2000)

Data collected by the Indonesian Central Bureau of Statistics shows that the human population density for the whole of Sumatra and the four provinces that span KSNP has steadily increased since the 1970s (Figure 2.5). The human population growth in West Sumatra closely follows that of the average for Sumatra, whereas the other three provinces have lower than average population densities. The human population density of Jambi is beginning to decline: a result of the outward migration (Figure 2.6).

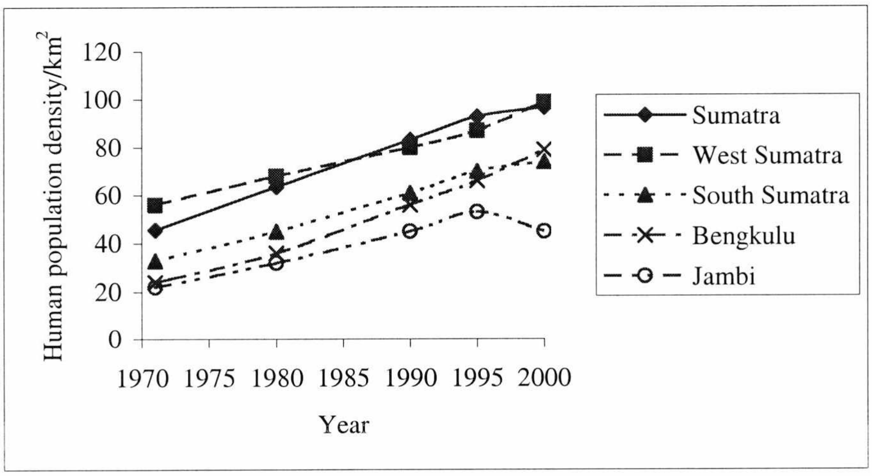


Figure 2.5: Changes in human population density across Sumatra and the four provinces that span KSNP (based on data from BPS, 2000)

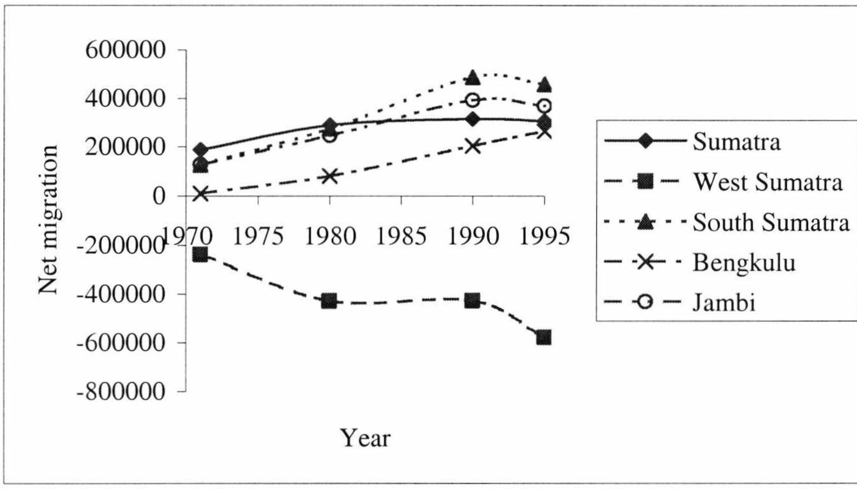


Figure 2.6: Changes in net migration across Sumatra and the four provinces that span KSNP (based on data from BPS, 2000)

The population density on Sumatra (90 person/km^2) is much lower than on the neighboring Indonesian island of Java (951 person/km^2). The overcrowding problem on Java arose because as much as 90% of the island's forest was cut down to alleviate the land shortage problem (Lewington 1997). Still insufficient, the government and international donors sponsored a transmigration program from Java to Sumatra and Kalimantan (Indonesian Borneo), and also from other Indonesian islands such as Madura and Bali, in the 1970s and 1980s. In addition, there was spontaneous migration from these islands, which still continues today. From the 1970s, to present there has been a steady increase in the flow of migrants to the provinces which include KSNP. Bengkulu in particular is currently the recipient of most of the migrants. Many of these migrants to Bengkulu are the second generation of Javanese and Sundanese whose parents settled in South Sumatra, the neighbouring province. In relation to KSNP a number of expanding transmigration settlements lie to the east and south of the reserve (WWF 1989). The Minangkabau of West Sumatra are famous for their *merantau*¸ travelling, around Indonesia (Ananta et al. 2002). Consequently this province is experiencing a substantial negative net migration because many Minangkabau are searching for illegal but better paid labour work in Peninsular Malaysia.

Due to recent changes in administrative boundaries it is not possible to map the demographic changes for villages or districts surrounding KSNP. From the 2000 National Census the highest population densities were recorded from the villages in the districts of Rejang Lebong, Kerinci, and Musi Rawas that are peripheral to KSNP (Figure 2.7, Table 2.2). The lower population densities may reflect the greater proportion of forest within these districts, such as Bengkulu Utara, but at the village level there are clusters of villages with high human population density within these districts (Table 2.2).

Table 2.2: Population density in the 10 districts of the KS region

Province	District	Population density (person/km^2)
Bengkulu	Bengkulu Utara	92
Jambi	Bungo	37
Jambi	Kerinci	324
Jambi	Merangin	30
South Sumatra	Musi Rawas	304
West Sumatra	Pesisir Selatan	213
Bengkulu	Rejang Lebong	480
Jambi	Sarolangun	15
West Sumatra	Sawah Lunto/Sijunjung	67
West Sumatra	Solok	223
Average	-	179

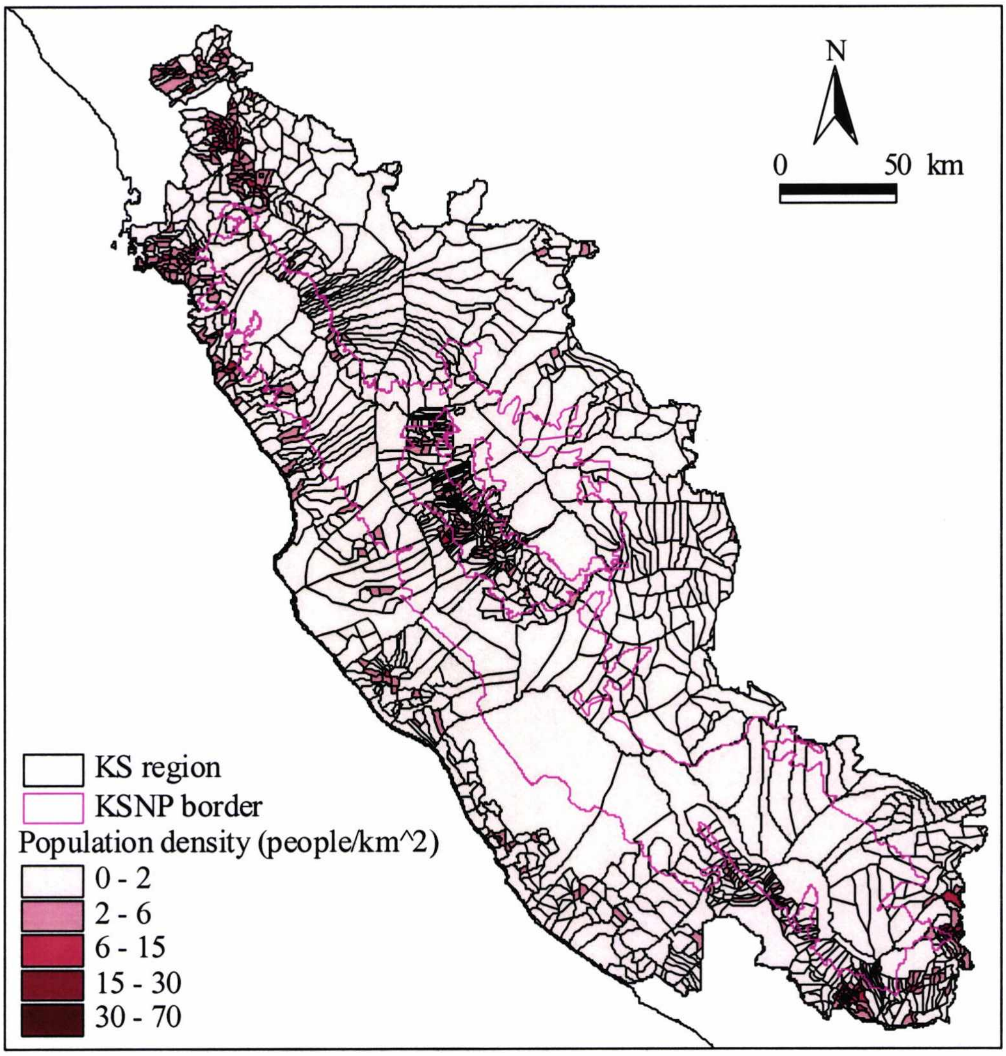


Figure 2.7: Village population density across the KS region

The human sex ratio in Sumatra is close to parity although slightly skewed towards males. During the 1970s Jambi had a sex ratio that indicated a population with a much larger proportion of males compared to females (107.5). However, over the past decades this has moved more towards parity although there is still a male bias (Figure 2.8). Conversely, West Sumatra had a female biased sex ratio (93.7), which may reflect the tendency of the Minangkabau males to search for employment in other provinces or abroad.

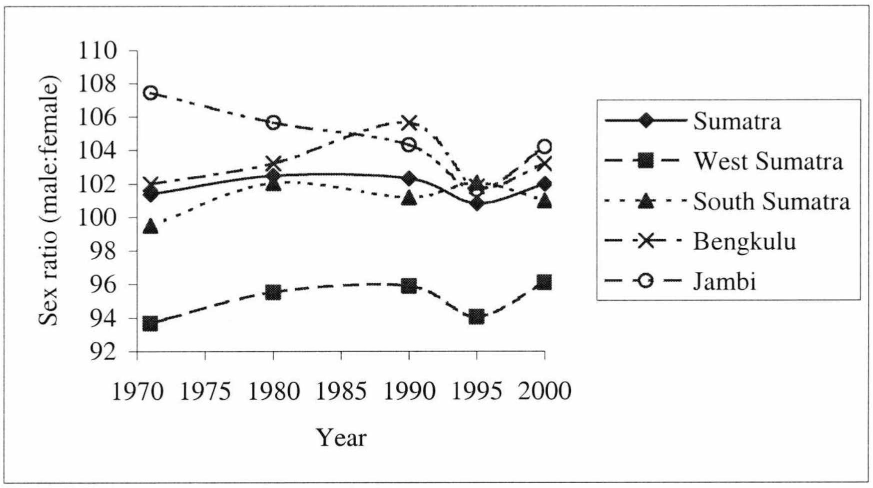


Figure 2.8: Sex ratio in the KSNP provinces

2.5 ECONOMIC ACTIVITIES AND HUMAN-RELATED THREATS

Indonesia has been identified as having the highest number of threatened mammals globally (147 species; second is Brazil with 81 species) (IUCN, 2002). Habitat loss and fragmentation, largely from deforestation, is the most severe threat.

From January 1997 until April 1998 Indonesia and many other southeast Asian countries suffered greatly from drought and forest fires caused by the El Niño Southern Oscillation (ENSO) phenomenon (Stolle and Tomich 1999, Siegert et al. 2001). These conditions led to the large scale clearance of land for oil palm plantations all of which had serious consequences for the forests of Sumatra. The financial crisis that hit Indonesia from 1997 to 1998 saw the Rupiah devalue by 80%, bank interest rates reach nearly 70%, and trading in many sectors of the economy come to a complete standstill. These factors contributed to the country's GDP shrinking by almost 14% (European Commission 2002). The country increased exploitation of its rich natural resources to fuel economic recovery. Control and regulation of this exploitation became severely weakened with the fall of President Suharto in May 1998. The decentralization process that followed led to an increase in illegal logging that quickly spiraled out of control (McCarthy 2000). Decentralization of the natural resource sector had serious consequences: high and unprecedented levels of illegal logging in Sumatra, to which the forests of KSNP were not immune.

Decentralization created a conflict of interests between local governments and conservation officials. Local governments now had more responsibility to raise their revenues. Given that PAs, such as KSNP, did not contribute any land tax to the national exchequer but occupied potentially taxable land weakened political support for conservation officials (Holmes 2001). In trying to tackle problems such as farmland encroachment, oil palm plantation creation or illegal logging, law enforcement from the police of military would be inadequate, especially if they are complicit (Kaimowitz and Ahmad 2003, Linkie and Sibarani 2002).

2.5.1 Mining

In Indonesia, a Presidential decree gives mining priority over all other land uses. During the 1990s a mining boom saw the country's major mineral extraction increase by at least 20% (Holmes 2001). In the KSNP region mining concessions cover 9,918 km^2. Within KSNP, 3,305 km^2 has been designated for mining operations. Gold and silver mining in the southern area of KSNP, in Musi Rawas district, has polluted the rivers in the area around the mining sites. Apart from chronic heavy metal pollution and sedimentation due to uncontrolled runoff from the pit area, local villagers have described episodic fish kills in the river, which could be attributed to cyanide release from the gold winning process (WALHI 2000).

2.5.2 Oil palm production

Oil palm production is a major cause of forest conversion in Indonesia. From 1967 to 1997, oil palm production was one of the fastest growing sectors in Indonesia's economy. Whilst significantly benefiting the country's economy, it was at the cost of displacing large tracts of rainforest and local communities. The ENSO in 1997 reduced oil palm production. Many estates in Sumatra decided to capitalize on the dry conditions by clearing more land through burning (Wakker 1998). This only led to thick haze that made the situation worse for oil palm production because it prevented photosynthesis, killed off important weevil pollinators, and prevented employees being unable to work because of poor visibility and health reasons. Much of this cleared land was never developed due to a precipitous decline in the price of crude oil palm. Poor infrastructure and ethnic unrest meant some oil palm companies performed poorly. However, many oil palm companies were more interested in the extraction of export quality timber in their concessions and so forged strong links with logging companies instead of establishing plantations (Cassons 2000). There is little information on the creation of oil palm plantations at the expense of primary forest, other than that there is a relationship (Osgood 1994). Over 10 years ago, the actions of transmigrants clearing forest for their small scale farming were thought to be the main cause of deforestation (FAO 1990, World Bank 1990, Barbier et al. 1993). However, over the last 10 years, oil palm plantations and industrial forest concessions have become the main cause of deforestation in Indonesia (Casson 2000, 2003).

2.5.3 Small scale farming

Traditional shifting cultivation farming that involves fallow periods and forest conservation is a rarity on Sumatra (Tomich and van Noordwijk 1995). Pioneer farming is one of the most destructive systems for forest cover and is commonly practiced by transmigrants searching for new land (Sunderlin and Resosudarmo 1996). In the provinces of South Sumatra and Bengkulu, pioneer farmers have cleared forest right up to the KSNP border and in some cases have started clearing inside the national park. These communities at the forest edge are some of the poorest in Indonesia. With short time horizons they are likely to deforest more, although these farmers may lack the capital to turn land into production. Land tenure rights are often unclear in these areas. In the provinces of West Sumatra and Jambi, communities generally tend to be well established and so land tenure is securer. In Bengkulu many of the newer farmers do not own the farmland they have created and there is a large influx of transmigrants searching for land, both of which decrease land security. The main crop types grown in the KS region include coffee, patchouli, cinnamon, chili, rice and vegetables such as cabbage. In the province of West Sumatra there are long established rubber estates.

2.5.4 Commercial logging concessions

Nearly all of the commercial logging in Indonesia is being done at an unsustainable rate (World Bank 1995). Although there is much rhetoric from the Indonesian Government about making serious attempts to tackle the corruption involved with commercial logging operations, many of the logging barons continue to operate with impunity (EIA-Telepak 2003). On Sumatra, commercial loggers often find themselves in competition with illegal loggers and therefore have no incentive to exploit the timber responsibly. There are currently 13 commercial logging concessions abutting

KSNP: five are active, three are non active, and five are of status unknown (because they were not the focus of the logging concessions component of the KS- Integrated Conservation and Development Project, ICDP).

2.5.5 Small investors

These are typically urban businessmen or government servants who acquire farmland and then hire rural labourers to clear and manage plots comprised of a few hectares of tree crops, such as cinnamon or rubber. Acquisitions are often informal and do not, therefore, appear in government statistics, making it difficult to gauge the extent of deforestation from this source (Holmes 2002). What is clear is that the impact of the monetary crisis created opportunities for the better-off farmers, immigrants and urban dwellers with capital to convert forests to profitable crops (Angelson and Resosudarmo 1999). The situation is still complex, varying from province to province, and warrants further research because the power vacuum created coupled with weak law enforcement makes illegal logging and encroachment into protected areas more likely.

2.6 CONSERVATION VALUE AND MANAGEMENT OF KSNP

While most existing PAs in Asia are small (Dinnerstein and Wikramanayake, 1993), Sumatra has two of the largest, GLNP and KSNP. Despite the problems associated with decentralization of the natural resource sector the need still remains to bring these forests under sustainable management (Holmes 2001). Between 1997 and 2002 the World Bank financed an ICDP for KSNP to secure the biodiversity of the park and stop further habitat fragmentation. The project design proposed an integrated approach by i) linking park management to regional development and spatial planning; (ii) coordinating implementation; (iii) regular monitoring and enforcement; (iv) increasing staff and in-service training; and (v) improving resource management and service delivery (World Bank 1996[a]). To help achieve this village conservation agreements were established and special and traditional use zones were set up for KSNP, allowing people with legitimate rights to utilize park resources. The ICDP met with difficulties at various levels. It is too early to determine if the implementation of the co-management process within designated park zones has been successful.

Monitoring of the zoning agreements and the village conservation agreements along with the local people will decide their success or otherwise (Anon. 2002).

The importance of KSNP to the long-term persistence of tigers in the wild was reflected in its high ranking score and designation as a level I tiger conservation unit (Wikramanayake et al. 1998). Patrolling and law enforcement are conducted by the Tiger Protection and Conservation Units (TPCU) that were established in 2000 by Fauna and Flora International, the Indonesian Academy of Sciences and the Department of Forestry and Nature Protection. Before the TPCUs, there was little in the way of ground level protection for KSNP. These units are a combination of national park personnel and villagers from communities that border the park. The TPCUs have been successful: from 28 TPCU forest patrols conducted during 2000 there were 66 arrests, 10 chainsaw seizures, and 179 confiscations of sawn logs, of which 166 were destroyed and 13 were held as legal evidence. The detection of illicit activities within KSNP has generally increased as patrol units contained a greater number of staff (Linkie et al. 2003).

The vast expanse of the KS region includes large quantities of spatial information on tiger habitat, tiger and prey distributions and the threats for tigers and their prey. In order to guide the research themes in this thesis, community surveys were undertaken. This provided a clearer understanding of what problems farmers and also tigers, their prey, and their habitat faced around KSNP and is documented in the next chapter.

Chapter 3
PIONEER FARMING AROUND KSNP

Newly created farmland in Tapan Valley bordering KSNP (J. Holden)

3.1 INTRODUCTION

For conservationists, large carnivores are often flagship species that can attract international conservation funding and protect the wider biodiversity of an area (Leader-Williams and Dublin 2000). Such western-driven conservation values are often in disharmony with the values of the local communities who live with wildlife and bear the costs (Colchester 1997). To a local farmer, flagship species such as tigers and elephants may represent a livestock predator or a crop pest that cause loss of life or livelihood. To achieve better harmony between conservation programmes and local needs, it is necessary to understand the basis of this dichotomy.

To more clearly understand the loss of species and their habitats, it is important to determine the socio-economic profile of those responsible (Armitage 2002). Common property systems are an important institution for mediating the relationship between population change and environmental outcomes (Curran and Agardy 2002). The substantial negative effects of resource consumption on biodiversity are well documented and occur on many scales. Household dynamics, which are often overlooked, are important agents of biodiversity decline. For example, the consumption of fuel wood is influenced by household size (Liu et al. 2003). It is important to link this information with other demographic and socio-economic factors that might explain unsustainable consumption.

A principal form of resource depletion in the tropics is forest clearance for farmland. This action often worsens the situation for farmers because it can result in increased soil erosion, landslides, flooding, and human-wildlife conflict. Therefore, it is also important to understand whether farmers associate their actions in clearing land with the associated adverse effects of deforestation. The compilation of such information can be helpful in constructing and implementing conservation management and monitoring programmes (Armitage 2003). Obtaining socio-economic information on some of the farming communities involved in deforestation in and around KSNP should assist any strategies that target these communities for outreach programmes.

The farming communities that live adjacent to KSNP co-inhabit a landscape with a PA that excludes access and use, while also harbouring tigers and other large mammals that are important crop pests in the farmland. In such circumstances, the attitudes towards conservation of those living in close proximity to wildlife habitats are strongly influenced by the problems associated with wildlife (Newmark et al. 1993, 1994). If tigers, their prey, and their forest habitat are to be effectively protected, it is crucial to work in cooperation with the local communities. To facilitate this cooperation it is necessary to gain an understanding of the perceptions and attitudes that farmers hold towards the tiger, its prey, and KSNP.

3.1.1 Aims and objectives

This chapter aims to:

- determine the demographic, social, and economic structure of a typical farmland community outside KSNP;
- determine the farming systems and practices of this farmland community;
- determine farmer's knowledge of, and interactions with, KSNP;
- determine their non-timber forest product collection practices; and,
- determine farmer's perceptions and attitudes towards wildlife and tigers in KSNP.

3.2 METHODS

3.2.1 Study site

The Air Dikit study site located in Bengkulu province (2°56'–2°64'S, 101°43'-101°51'E) was selected because it is a recently formed farming community that borders KSNP. Within this area there are still small patches of degraded forest outside of KSNP but the landscape is dominated by farmland that has been created through government sponsored, as well as spontaneous, transmigration from neighbouring Java and South Sumatra province. The Air Dikit site has an altitudinal range from 100 to 300 m that supports a western lowland forest type (Laumonier 1994) and receives an average annual rainfall of 4500 mm.

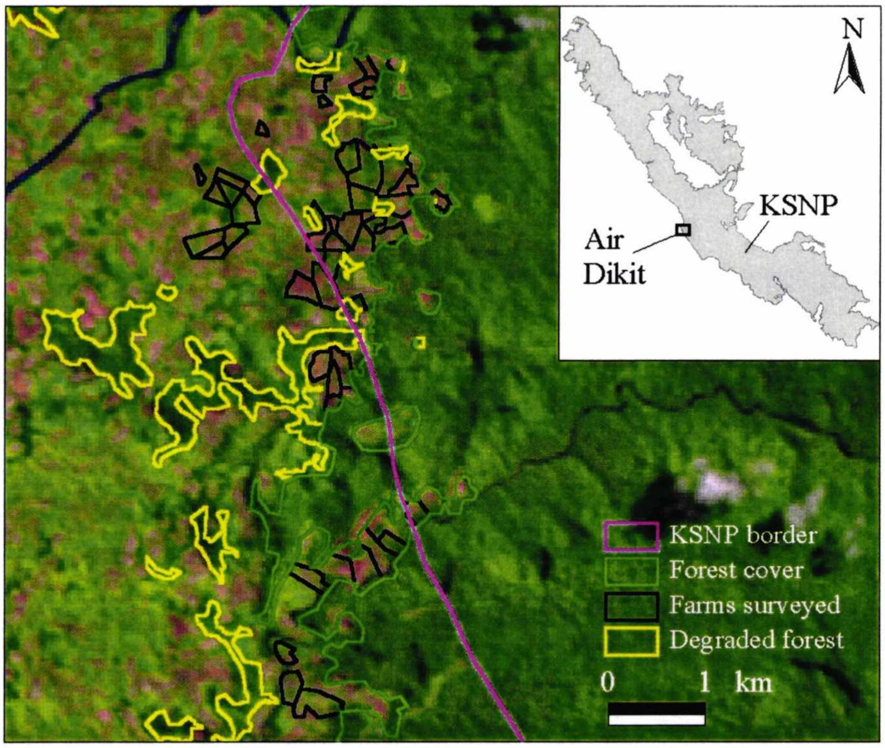


Figure 3.1: Landsat 7 colour composite image of the fifty farms surveyed in Air Dikit, Bengkulu province

3.2.2 Field methods

Between March 2001 and July 2001 semi-structured interviews were administered to 50 farming households. Only the male head of each household was interviewed, both to standardize the sampling group and because it is expected and respectful in this culture. Interviews typically lasted between 30 and 40 minutes. The semi-structured style and less formal approach also fitted with the local culture, and had previously been successful in a pilot study conducted in another study site (Linkie 1999). Pilot testing was conducted to check the reliability, validity, and clarity of the questionnaire with some questions being rewritten before final administration (de Vaus 1999). These survey methods have proven validity for collecting quantitative data on natural resource management and community attitudes towards conservation (Infield 1988, Newmark et al. 1993, Sekhar 1998).

To minimize potential bias in the responses, all interviews were conducted with the aid of a local assistant and an Indonesian university undergraduate student. While there is always a potential for biased responses to interview questions, this concern was minimized through having already worked within this community for 4 months, my own proficiency in *Bahasa* Indonesian, and assistance from the well respected village deputy head. The final questionnaire was divided into five main sections (see Appendix 1 for the full questionnaire):

3.2.2.1 Household information

This section served the two main purposes of determining the demographic and socio-economic characteristics of each household. Once all members of the household were identified, basic demographic data on their age, sex, and marital status was obtained. Socio-economic data was then collected on:

- the highest level of education for all household members;
- the possessions they owned, including field plough, chainsaw, hand saw, work tools, manual crop sprayers, radio, TV, pushbike, motorbike, and cart;
- livestock owned, including the number of chickens, goats, cows, buffalos, and horses; and,
- access to utilities and amenities such as water, sewage, electricity, telephone, and cooking facilities.

3.2.2.2 Farming systems and practices

This section collected information on all agricultural activities, including land tenure, crops grown, use of crops, crop production, problems with farming, and how these problems might be overcome.

3.2.2.3 Farming and KSNP

This section collected information on each farmer's plans to expand their farm, their knowledge on KSNP regarding its boundary, planning, problems, and restrictions on farming or collecting non-timber forest products (NTFPs) inside the PA.

3.2.2.4 Non-timber forest product resource use

This section collected information on NTFP activities in the forest; type of NTFP collected, the frequency of NTFP collection, the number of hours spent forest searching for NTFPs, and the average amounts collected.

3.2.2.5 Farmer's perception and attitudes towards wildlife

This section collected information on each farmer's perceptions on and attitudes towards, wildlife in general and tigers in particular. Farmers were asked to comment on wildlife population trends over the past 3 years from the forest immediately surrounding KSNP. Farmers were then asked to comment on whether their views on the population trends of these species arose as a result of forest being cleared for farmland. Finally, farmers were asked about their perceptions and attitudes towards tigers.

3.2.3 GIS methods

The boundary of each farm unit was mapped using a geographic positioning system (GPS) with an accuracy of 4-5 m. These data were imported into ArcView v3.2 and converted into individual farm polygons within a vector file. The area of each farm was then calculated using the ArcView 'X Tools' extension file. The socio-economic information from the questionnaire survey were then imported into ArcView and added to its corresponding farm. Finally, the position of each farm as to whether it was located inside or outside of KSNP was determined by overlaying the farm polygons with the KSNP border.

3.2.4 Statistical methods

Data were analysed using descriptive statistics and responses were compared using a Chi-squared test. An important statistical development of the last thirty years has been the advance in regression analysis provided by generalized linear models (Guisan et al. 2002). These are mathematical extensions of linear models that do not force data into unnatural scales, and thereby allow for non-linearity and non-constant variance structures in the data (Hastie and Tibshirani 1990). They are based on an assumed relationship (called a link function) between the mean of the response variable and the linear combination of the explanatory variables. Data may be assumed to be from several families of probability distributions, including the normal, binomial, Poisson, negative binomial, or gamma distribution, many of which better fit the non-normal error structures of most ecological data.

For each farm, physical data on farm size and farm position to KSNP and socio-economic information on each farmer's age, highest level of education, family size, and poverty (indicated by the possession of a radio and the number of livestock) were recorded. These data were imported into SPSS v.11 statistical software package (SPSS Inc., Chicago, IL). The continuous data were logarithmically transformed to improve their normality. A multiple logistic regression model or ordinal regression model was used to determine which combination of factors most accurately predicted farmer responses to the following questions:

- Is the amount harvested on your farm more or less than the previous year?
- Do you want to increase the size of your farm?
- Do you think that it is illegal to farm inside KSNP?
- Do you think that it is illegal to collect NTFP from inside KSNP?
- Do you think that it is illegal to collect NTFP from inside a logging concession?
- Do you collect NTFP?
- Do you think that humans and tigers can co-exist?
- Do you think that tigers are threatened?

The addition and removal of independent variables from the regression model was controlled by the Wald statistic with respective *P*-values of 0.05 and 0.1. The performance of the model was evaluated by calculating the area under the curve (AUC) of the receiver operating characteristics plot (Manel et al. 1999, Pearce and Ferrier 2000, Osbourne et al. 2001). These values range from 0.5 to 1.0, and those above 0.7 indicate an accurate model fit, while those above 0.9 indicating a highly accurate model (Swets 1988). In the spatial analysis it was necessary to test for non-independence caused by spatial auto-correlation because landscape features close to each other tend to have similar characteristics (Koenig 1999). The presence of spatial autocorrelation in the model was tested by calculating Moran's *I* statistic (Cliff and Ord 1981) using the Crime-Stat v1.1 software package (N Levine and Associates, Annadale, VA).

3.3 RESULTS

3.3.1 Demographic and socio-economic farmland structure

The average family size for each respondent household was 3.8 people (range of 1 to 8, SD = 1.82; Figure 3.2), with an average age of 22.5 years (range of 1 to 63, SD = 14.08; Figure 3.3). Most respondents had attained primary school level in their education (71.1%), a few had attained junior high school level (21.1%), while only a few had attained senior high school level (7.8%) (Figure 3.4).

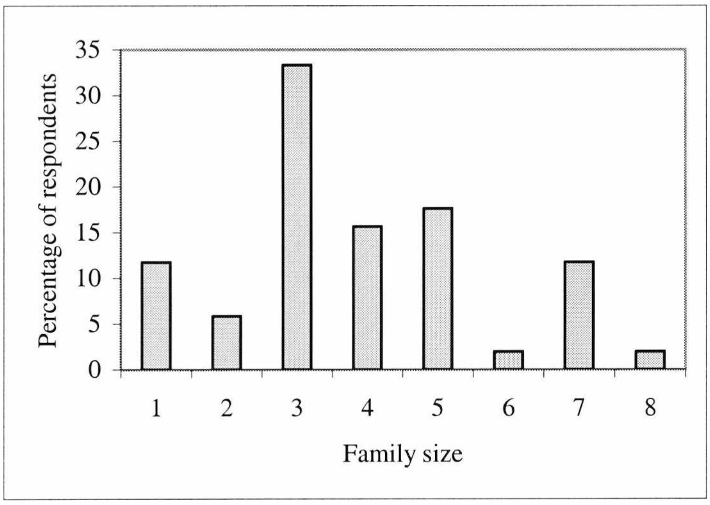


Figure 3.2: Family size among responding households

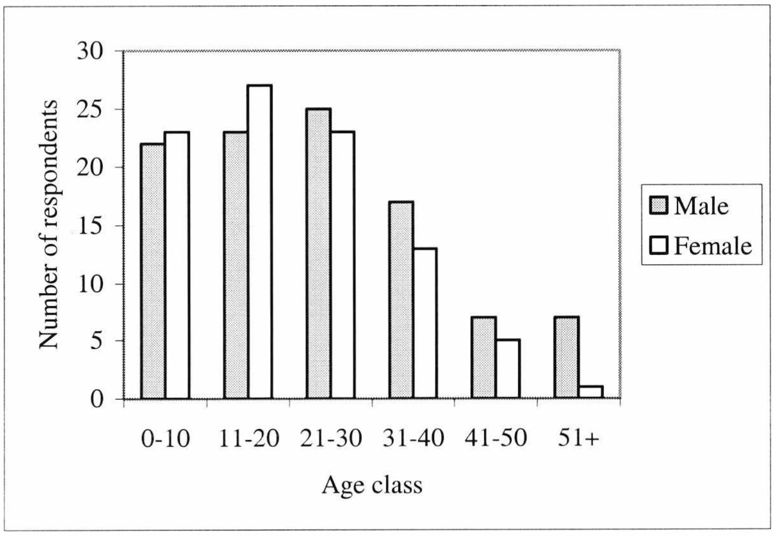


Figure 3.3: Age structures of males and females among responding households

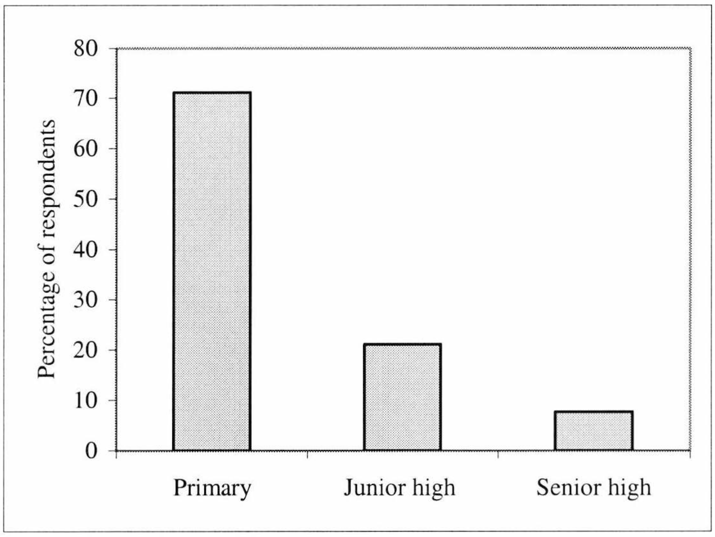


Figure 3.4: Highest level of education among responding households

Most households owned crop sprays (59.6%) radios (51.9%), and chickens (30.8%) (Figure 3.5). A few households owned hand tools (19.2%). However, it was rare for a household to own a television (3.8%), a motorbike (1.9%), or a car (1.9%).

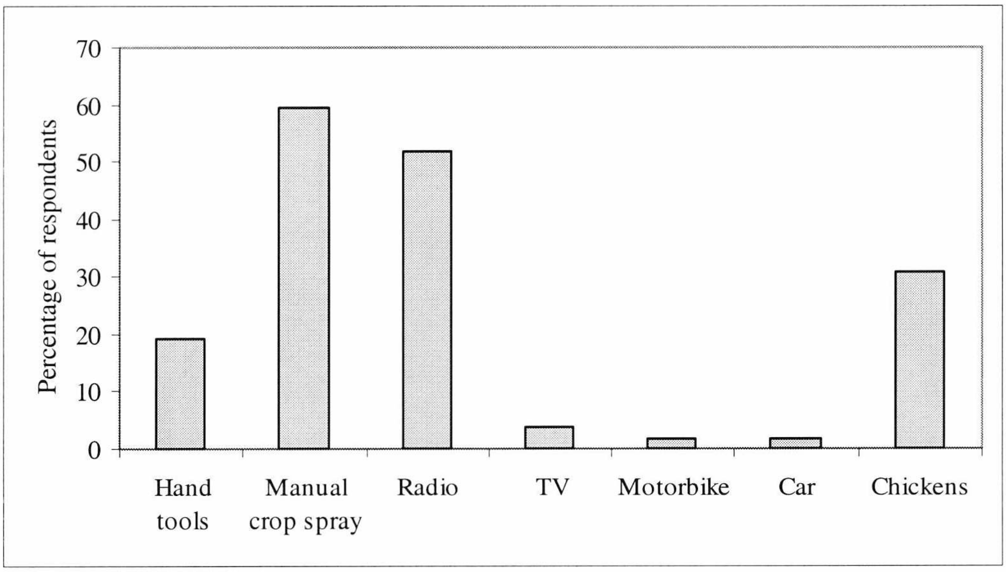


Figure 3.5: Equipment owned among responding households

All respondent's houses had a roof constructed from wooded slats, walls constructed from wood, earth floors, had no electricity and used oil lanterns for light, and used rivers or springs as their water source for cooking, cleaning, and bathing.

3.3.2 Farming systems and practice

The average farm size was 0.39 ha (range of 0.08 to 1.02, SD = 0.23) and the average duration that each respondent had occupied their farm was 3.5 years (range of < 1 to 11, SD = 1.80; Figure 3.6). All farmers had received a land use certificate from the village head, authorised by the sub-district head.

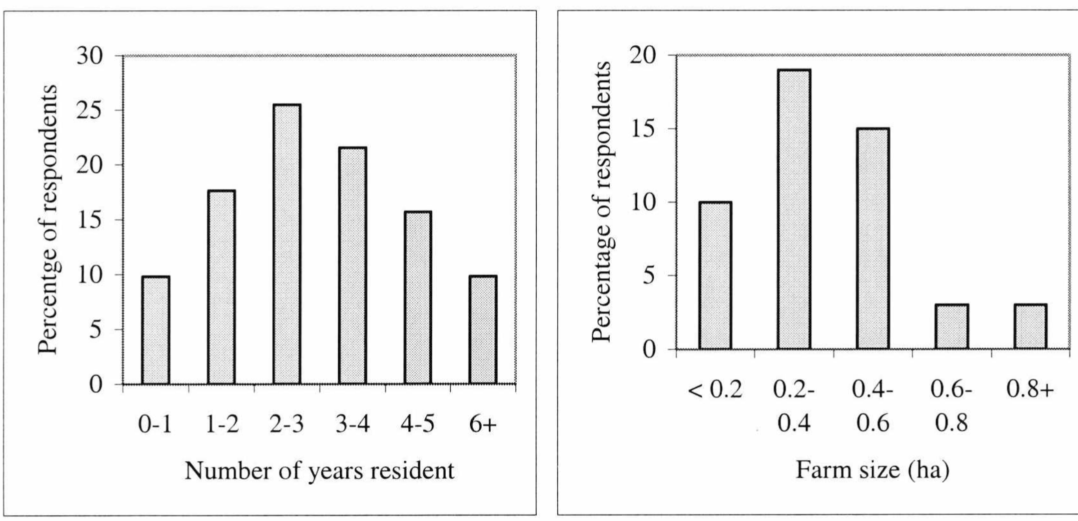


Figure 3.6: Length of residency and farm size among respondents

All farmers interviewed grew crops for both commercial and subsistence purposes. Crops grown were fairly evenly divided between commercial (44.3%) and subsistence (57.4%) crops. The main commercial crops grown were coffee (62%), patchouli (18%), or both coffee and patchouli (20%). The main subsistence crops grown by all farmers were rice, bananas, and chilli. During this study no land was in production for rice because it was out of season.

Farmers had different views on whether or not their harvest yields had changed over the past year ($\chi^2 = 13.84$, df = 3, $P = 0.003$; Figure 3.7). Many (36%) farmers believed their yields had increased, while an equal number believed their yields had decreased (36%). A farmer's response that their harvest yield had changed over the past year was related to their level of education and to $\log_{10}$ number of chickens owned (Table 3.1). Farmers who believed they had a higher harvest yield were better educated and were wealthier, i.e. they owned more chickens. The logistic regression model explained 54.1% of the original observations and was not affected by spatial autocorrelation (Moran's $I = 0.05$, $P > 0.1$). The final model had an AUC value of 0.763 indicating an accurate fit. Most (84%) farmers had not tried to plant crops that were not already traditionally grown in the area. Within the minority (16%) of farmers that had experimented with different crops, few (8%) had planted corn *Zea mays* var. *rugosa*, while black pepper *Piper nigrum* (3%) and watermelon *Citrullus lanatus* (2%). The results from growing corn and black pepper were still unknown because they had not been harvested. The farmer who grew watermelon abandoned this trial when this crop was destroyed during wildlife crop raiding forays. Most farmers used pesticides or herbicides on their crops (92%).

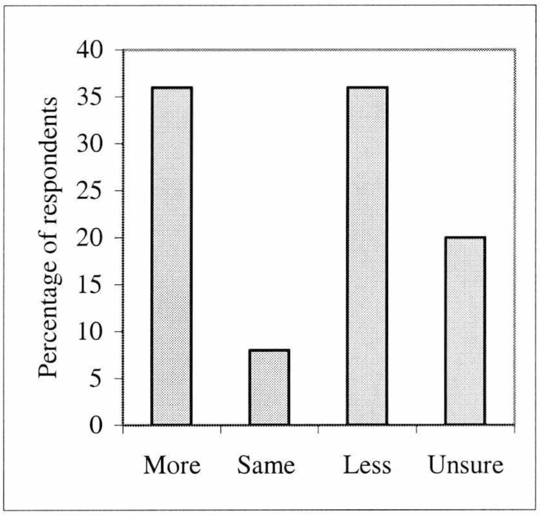


Figure 3.7: Respondent's perceptions over changes in their harvest yield over the past year

Table 3.1: Best multiple logistic regression model describing the relationship between change in harvest yield and physical and socio-economic factors

Factor	Coefficient (β) ± S.E.	df	Wald	*P*
Education				
Primary	1.700 ± 0.774	1	4.827	0.028
Secondary	(included in constant)			
Log_{10} number of chickens	1.842 ± 0.922	1	3.993	0.046
Constant	-2.933 ± 1.168	1	6.300	0.012

Most farmers thought that cutting down the forest would also increase flooding (94%), soil erosion (88%) and attacks from insect crops pests (66%). However, agricultural success was generally thought to be limited by crop raiding by wildlife (90.2%), followed by a decrease in market prices for cash crops (43.1%), and the long distance to the nearest market (37.3%) (Figure 3.8). Other problems mentioned by the farmers were poor communication, natural disasters, crop diseases, poor soil quality, and a poor knowledge of farming techniques. Most (70%) farmers were generally unsure about how they could overcome these problems. Of the remaining farmers, all thought that guarding their farmland might reduce crop raiding by wildlife.

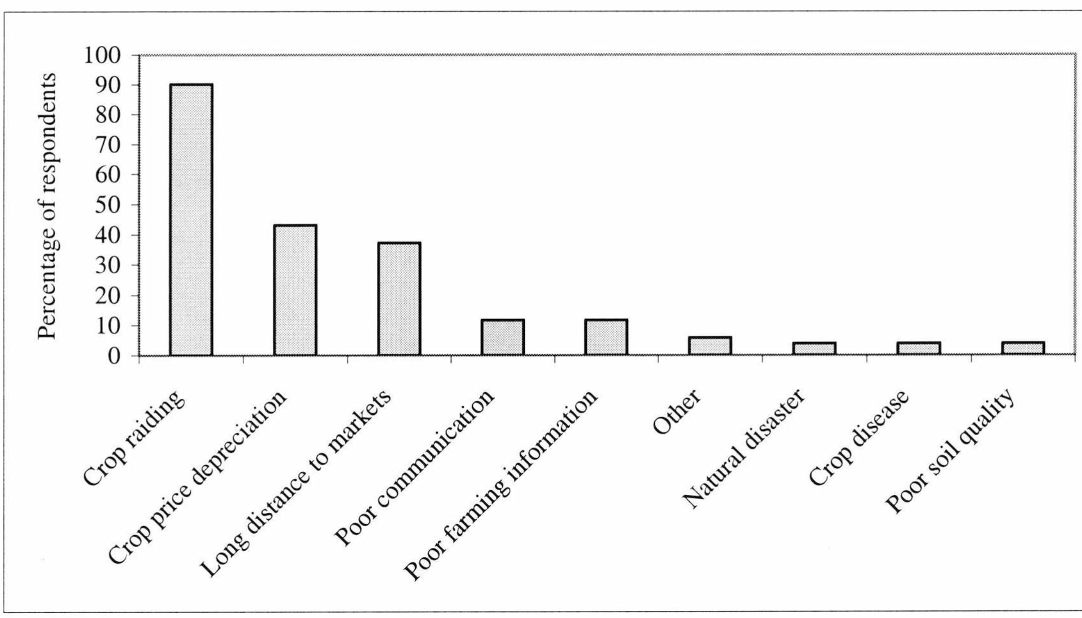


Figure 3.8: Factors considered by farmers to most limit agricultural success

3.3.3 *Farmers and KSNP*

Most (56%) of the farms surveyed were located inside KSNP. All farmers responded that they did not know where the KSNP boundary was. Most farmers did not want to expand their farm (58%), but those who did (42%) wanted to increase their farm by an average of 1.89 ha (range of 0.5 to 4, SD = 0.95). A farmer's desire to increase the size of his farm was related to $\log_{10}$ farm size (Table 3.2). Farmers owning larger farms were more likely to want to increase the size of their farm. The logistic regression model explained 60.0% of the original observations and was not affected by spatial autocorrelation (Moran's $I = 0.03$, $P > 0.1$). The final model had an AUC value of 0.628 indicating a reasonably accurate fit.

Table 3.2: Best multiple logistic regression model describing the relationship between a farmer wanting to increase his farm size and physical and socio-economic factors

Factor	Coefficient (β) ± S.E.	df	Wald	*P*
Log_{10} farm area	2.418 ± 1.118	1	4.678	0.031
Constant	1.549 ± 0.648	1	5.711	0.017

The creation of new or additional farms in Air Dikit would have to occur inside KSNP as there was little space available outside the park borders. Most (82%) farmers thought that there were no restrictions on them opening up new areas of land, which was probably why many (68%) farmers thought that it was not illegal to farm inside KSNP. The response of a farmer as to whether or not it was not illegal to farm inside KSNP was related to whether the farmer wanted to expand his farm. Farmers who thought it was not illegal to farm inside KSNP did not want to expand the size of their farm. The logistic regression model explained 75.0% of the original observations and was not affected by spatial autocorrelation (Moran's $I = 0.04$, $P > 0.1$). The final model had an AUC value of 0.712 indicating a fairly accurate fit.

Table 3.3: Best multiple logistic regression model describing the relationship between a farmer's views on legality of farming inside KSNP and physical and socio-economic factors

Factor	Coefficient (β) ± S.E.	df	Wald	*P*
Want to expand farm size	-1.814 ± 0.776	1	5.466	0.019
Constant	1.591 ± 1.130	1	1.981	0.159

Whilst most (74%) farmers had never had a problem with KSNP officials, some (20%) farmers had, citing farmland expansion into KSNP as the main reason. Most (76%) farmers did not know who designed KSNP, the remaining (24%) farmers correctly identified the Government of Indonesia. Many (58%) farmers had previously met a KSNP official who had come to their village.

Farmers were less clear about whether they were able to farm inside a logging concession. Some (34%) farmers thought they could, some (30%) thought not, and some (36%) were unsure. Most (62%) farmers thought that they were permitted to collect NTFP inside a logging concession. Many (48%) farmers also thought it was legal to collect NTFP from inside KSNP, while some (34%) thought it was illegal, and some were unsure (18%). These responses showed no relationship with any of the physical or socio-economic factors.

3.3.4 Non-timber forest product resource use

Most (60%) farmers collected NTFP from inside KSNP. The NTFP collectors mainly searched for 'damar', a resin from the Diptercarps of *Shorea* spp. and *Hopea* spp. (96.6%). A few farmers also collected *rattan* (16.7%) and *gaharu* or agarwood, a type of fungus that infects the heartwood of *Aquilaria* spp. (3.3%). The *damar* collectors would either enter the forest every 7 days (44.8%) or 2-3 days (37.9%) in search of this NTFP (Figure 3.9). A *damar* collector would spend on average 7.5 hours per trip inside the forest (range of 6 to 10, SD = 0.96; Figure 3.9).

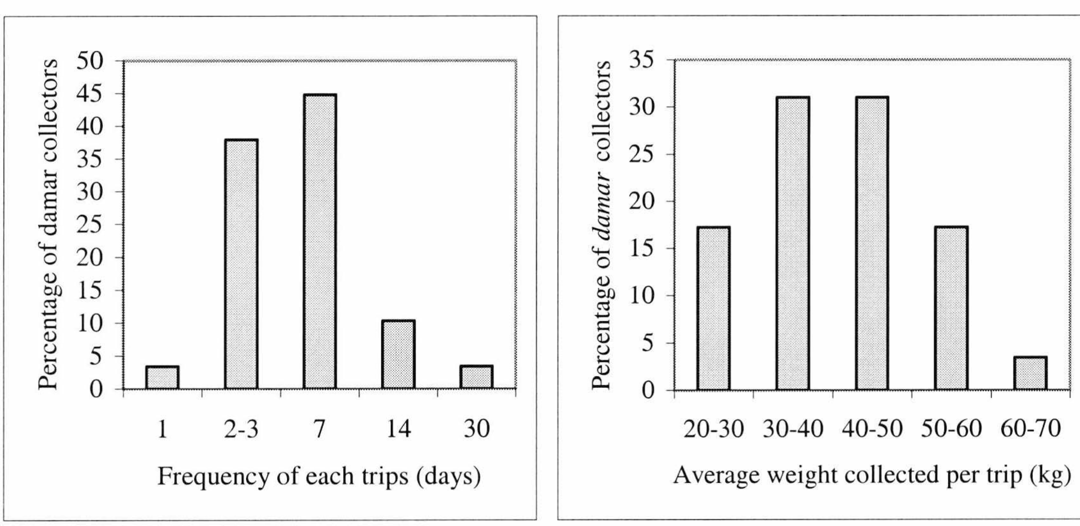


Figure 3.9: Frequency of NTFP collection trips and average amount collected

3.3.5 Farmer's perceptions and attitudes towards wildlife

Many (46.7%) respondents thought that wildlife in KSNP was generally decreasing, while some (22.5%) respondents suggested that wildlife was generally increasing, and a few (11.7%) believed it was stable (Figure 3.10). Many farmers thought that there had been an increase in wild boar (73.5%) and pig-tailed macaque (51.0%). Many farmers also thought that there had been a decrease in porcupine (58.8%), bearded pig

(48.8%), muntjac (73.5%), sambar (72.5%), and mousedeer (73.5%). Farmers were unsure about elephant (69.6%) and tiger (53.9%) population trends, although some (38.2%) did think that tigers were decreasing. For every wildlife species most farmers thought that their populations would decline as a result of clearing forest for farmland (Figure 3.11).

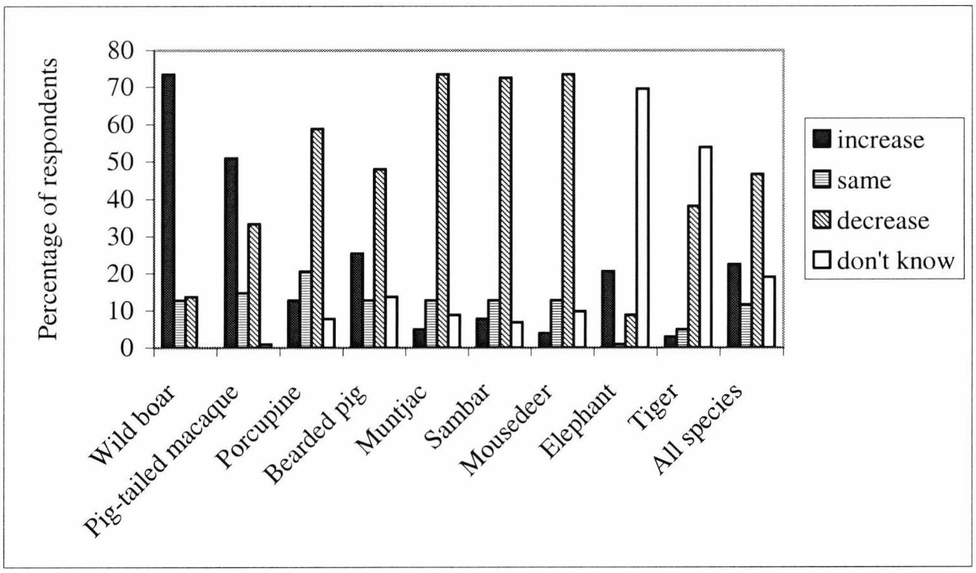


Figure 3.10: Farmer's views on wildlife population trends

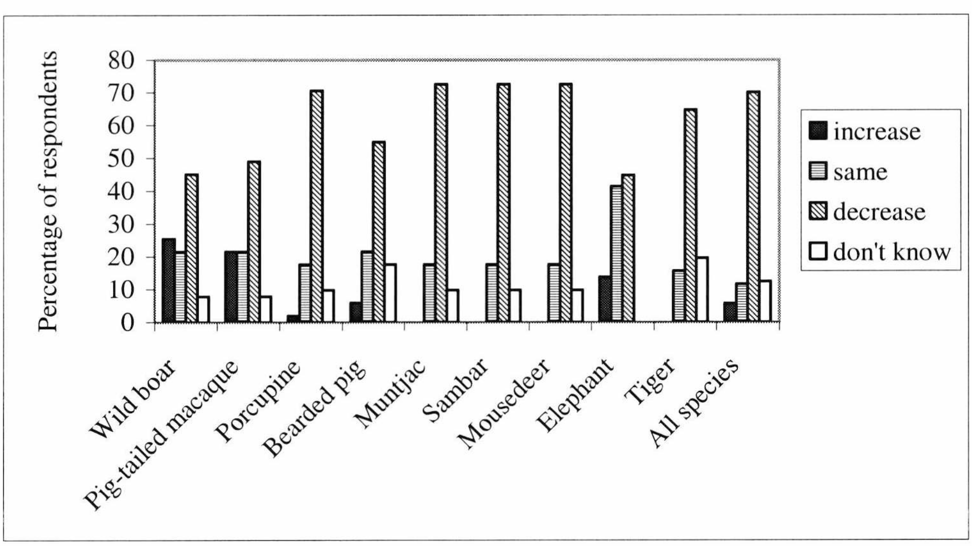


Figure 3.11: Farmer's views on wildlife population trends as a result of converting forest to farmland

Most (90%) farmers thought that the tiger was a good species, which was important to them, and which was important to conserve in KSNP (98%), even though many (68%) farmers also thought that tigers were dangerous to humans. No farmers had ever had a problem with a tiger. There was no general consensus amongst the farmers as to whether tigers and farmers within their area could coexist in the future (yes = 42%, no = 36%, unsure 22%), or if tigers were threatened (yes = 18%, no = 26%, unsure 56%). All farmers thought that national law protected tigers. A farmer's response as to whether humans and tigers could coexist was related to their age (Table 3.4). Farmers who thought that humans and tigers could coexist were more likely to be older. The logistic regression model explained 70.0% of the original observations and was not affected by spatial autocorrelation (Moran's $I = 0.03$, $P > 0.1$). The final model had an AUC value of 0.675 indicating a reasonably accurate fit. The response of whether or not a farmer thought that tigers were threatened showed no relationship with any of the physical or socio-economic factors.

Table 3.4: Best multiple logistic regression model describing the relationship between whether tigers and humans can coexist and physical and socio-economic factors

Factor	Coefficient (β) ± S.E.	df	Wald	*P*
Age	0.088 ± 0.046	1	3.691	0.055
Constant	-2.698 ± 1.521	1	3.145	0.076

3.4 DISCUSSION

KSNP is an important refuge for the Sumatran tiger and its prey. KSNP is also important for the neighbouring communities because the forest provides them with a supplementary income through their collection of NTFP. The farmers therefore have a vested interest in maintaining the forest. However, these pioneer farming communities constitute some of the poorest people on Indonesia and clearing forest for additional farmland is the most convenient way to improve their livelihood. In this study, the negative affects identified with cutting down the forest were a loss of forest habitat and prey for tigers, and a loss of NTFP and an increase in problems associated with crop production.

Most farmers thought that forest loss would result in a population decline of tigers and their prey. This was interesting because it contrasted with most farmer's responses that wild boar and pig-tailed macaque populations were increasing at the forest edge. The farmers also thought that cutting down the forest would lead to increased flooding, soil erosion, and attacks from insect crop pests, but they did not consider these to be the most important factors limiting agricultural success. An overwhelming number of farmers identified crop raiding as the single most important factor limiting agricultural success.

Crop raiding is a major form of human-wildlife conflict in developing countries because it commonly occurs where farmland adjoins forest. In communities with subsistence economies, even small losses from crop raids can be of financial significance and can generate negative attitudes towards wildlife and conservation (Mishra 1992, Oli et al. 1994). Replacing forest with crop rich farms typically favours certain wildlife species, such as wild boar, that can maintain higher population densities near the forest edge. This in turn can support a greater number of large carnivores that come into closer with contact humans. In Dudhwa National Park, India, an increase in ungulates, attributed to the conversion of forest to sugarcane, led to an increase in tiger presence and then an increase in human-tiger conflict, whereby a total of 197 humans and 33 tigers were subsequently killed between 1978-1988 (Khushwah 1990 in Nowell and Jackson 1996). This generates intolerance and antagonism towards wildlife and, if remedial measures are not taken quickly, then retribution killings are likely (Talwar 1999).

In this study, the farmers had never had a problem with tigers and were found to have a generally positive attitude towards them. Furthermore, most farmers thought that conserving tigers in KSNP was important. Indeed some communities around KSNP view the tiger as an ancestral figure who protects them and punishes only those who have violated the *adat* (customary) law (Bakels 1994). Farmer's attitudes may also be linked to the benefits received from KSNP, such as collecting NTFP. Farmers are prohibited from entering KSNP but this law is so rarely enforced that *de facto* it does not exist in many areas of KSNP. This situation does not send out a strong law enforcement message and may explain why farmers thought that there were no restrictions on them opening up new farm inside KSNP. Given the lack of space available outside of KSNP and the desire of many farmers to increase the size of their farm, this poses a real threat. Where access to natural resources is restricted in a PA, communities have been found to have a negative attitude towards the PA and wildlife conservation (Fiallo and Jacobson 1995, Tisdell 1995, Badola 1998, Ashenafi 2001). Around KSNP, the communities were unsure about these restrictions, but frequently entered the national park without regard, possibly because they did not know where the park boundaries were located. A KSNP community ranger (TNKS-mitra) commented that this may be because the boundary markers (*pal batas*) were intentionally removed by objecting villagers.

The competition between farmers and wildlife over space and resources identified in this chapter epitomize the salient issues in tiger conservation across the KS region and across all tiger range states. The clearance of forest for farmland will reduce the amount of habitat for tigers and their prey, and therefore affect their distributions. It may cause an increase in tiger prey at the forest edge and an increase crop raiding, or it may result in edge effects that cause a shift in tiger prey and therefore tigers from the forest edge towards the interior (Kinnaird et al. 2003). The response of tiger prey to deforestation is unclear, as was indicated by the farmers in the questionnaire survey. The remainder of this thesis will now address each of these issues in turn in the subsequent chapters. However, in order to achieve this, a GIS and socio-economic database will be constructed so that spatial and temporal information from the KS region can be incorporated into the subsequent analyses. The following chapter describes how this was done.

Chapter 4

CONSTRUCTING A GIS AND SOCIO-ECONOMIC DATABASE FOR THE KERINCI SEBLAT REGION

The forested landscape of KSNP, an important watershed for areas outside (J. Holden)

4.1 INTRODUCTION

The use of space is implicit to any ecological investigation and is central to ecological theory (Dale 1999, Folt and Burns 1999). Spatial characteristics and relationships in nature are often difficult to identify and hard to display with traditional ground surveys or statistical models. Therefore, the use of a Geographic Information System (GIS) has increased in ecology and conservation studies as it enables mapping of habitat and species distributions, and identification of patterns of association and change, making it a powerful tool for conservation and wildlife management. A GIS is an information technology with the capability of:

- storing, managing, and integrating spatially referenced data relating to points (e.g. individual trees), lines (e.g. rivers, roads), and polygons (e.g. forest boundaries, habitat types, territorial ranges);
- conducting spatial queries (e.g. searching for areas in which a particular species or feature occurs);
- a database from which causal relationships can be derived using statistical modelling methods;
- engaging in geographic analysis (e.g. statistical analysis of relationships between habitat and reproductive success); and,
- displaying data in the form of high-quality maps.

Another facet of a GIS is the opportunity it provides to integrate remotely sensed data (Millington et al. 2001). The decreasing cost of these data means that it is now affordable for most conservation projects. These data, mostly from satellite imagery, are now a cost-effective method for mapping biological resources and their spatial distributions. Various statistical methods, such as maximum likelihood and Principle Components Analysis, are available in current GIS software for the classification of remotely sensed images. Integration of geographical information from remotely sensed images with other sources of geographical environmental information is best managed in a GIS. The GIS allows new maps to be readily constructed, but the accuracy of such maps needs to be quantified.

With rapid changes in land cover now occurring over large areas, remote sensing technology is an essential tool in monitoring tropical forest conditions. The remote and inaccessible nature of many tropical forest regions limits the feasibility of ground-based inventory and monitoring methods over large areas of land. Therefore initiatives to monitor land cover and land use change are increasingly reliant on information derived from remotely sensed data. Such information provides the data link to other techniques designed to understand the human processes behind deforestation (Lambin 1994). A GIS can provide the fine-scale data needed by protected area (PA) managers because the factors that influence deforestation are often site and scale specific (Geist and Lambin 2002). This also allows studies that combine physical and socio-economic data to be conducted at a village or a household level.

Being able to map household locations within a GIS offers the advantage of being able to overlay household locations with their associated socio-economic information within their physical landscape so that patterns can be establish of how the two interact. From this studies have been conducted on how the demography of individual farm units relates to deforestation in the Brazilian Amazon (McCracken et al. 1999) and how support for the development of a PA buffer zone by local people in Nepal was related to different socio-economic factors, such as age, level of education (Nepal and Weber 1994).

A GIS is particularly important for large PAs, which often have limited financial resources and need to focus their efforts for greatest effect (Leader-Williams and Albon 1988). For a large PA such as KSNP, the physical, biological, and socio-economic information embedded within a GIS can help guide management decisions as to where budgets for law enforcement and community outreach might be focussed. For tiger conservation, remote sensing and a GIS are essential tools for analysis and planning at landscape and population levels (Smith et al. 1998). They allow tiger habitat distribution to be mapped at a resolution fine enough to monitor populations and identify priorities (Dinerstein et al. 1996).

4.1.1 Aims and objectives

This chapter aims to explain the methods used to:

- produce radiometrically and geometrically corrected colour composite images for the KSNP region; and,
- produce physical and socio-economic GIS coverages for the KSNP region.

The data and coverages so produced form the basis for the subsequent analyses in Chapters 5, 6, 7, 8, and 9.

4.2 METHODS

The methods used to derive the GIS coverages are explained in two sections. The first section explains how remote sensing coverages were produced for the years 1995, 2001, and 2002. These years contain the most comprehensive and best quality remotely sensed data available from NASA. The second section details how the physical and socio-economic factors were constructed within a GIS. All data were created and manipulated using the GIS and remote sensing software Idrisi v32 and the GIS software ArcView v3.2.

4.2.1 Producing the remote sensing coverages

The remote sensing data came from several sources but all were resampled to gain a resolution of 100 m and to be projected using the WGS84 UTM 47s reference system. The methods used to produce the required data and the final remote sensing coverages are described below:

4.2.1.1 Landsat satellite images

Landsat satellite images are widely used for mapping vegetation and land-cover and have a 30 m resolution (Lauer et al. 1997). For this study 13 Landsat images from the years 1995, 2001, and 2002 were used with an additional 6 Landsat images from the years 1994 to 2000 to be used for crosschecking datasets (Table 4.1). These data were from Landsat TM and Landsat ETM+, obtained from the Basic Science and Remote

Sensing Initiative (http://www.bsrsi.msu.edu/trfic/index.html and http://www.landsat.org), costing between US$25 and US$600 each.

Table 4.1: Details of the Landsat images used in the study

Path/Row	Date	Bands used	Path/Row	Date	Bands used
127/061	19th June '02	1 – 5, 7	126/062	5th May '00	1 – 5, 7
125/062	20th May '02	1 – 5, 7	125/062	16th Aug '99	1 - 5, 7
126/062	11th May '02	1 – 5, 7	126/061	23rd Sept '96	2, 3, 4
126/061	24th Mar '02		127/061	13th Aug '96	2, 4, 5
126/061	9th June '01	1 – 5, 7	125/062	18th Aug '97	1 – 5, 7
127/061	31st May '01	1 – 5, 7	127/061	7th Jun '94	2, 4, 5
126/061	24th May '01	1 – 5, 7	126/061	17th Jun' 95	2, 4, 5
126/062	22nd May '01	1 – 5, 7	126/062	17th Jun '95	2, 4, 5
126/062	21st Mar '01	1 – 5, 7	125/062	14th Jul '95	2, 4, 5
127/061	7th Jan '01	1 – 5, 7			

The Landsat TM images needed to be radiometrically corrected, manipulated to remove the effects of atmospheric haze and enhanced to maximise the visual distinctiveness of the different vegetation types (Lillesand and Kiefer 1994). The Landsat ETM+ images were already geo-referenced, but for greater accuracy and standardization with the other Landsat TM images they underwent the same geometric and radiometric procedures. Previous studies have shown that selecting one band each from the visible, near infrared, and mid-infrared spectral regions results in the optimal waveband combination for vegetation discrimination (DeGloria 1984, Horler and Ahern 1986, Sader 1989). Therefore it was decided to use bands 2 (visible), 4 (near infrared), and 5 (mid-infrared) from each image. For the South Sumatra area (path 125, row 062) a complete cloud-free image was not available for 2001 or 2000, so a good quality image from 1999 was used in conjunction with these images to mosaic the best image. The images from 1994, 1996, and 1997 were used later on to check the land cover interpretation from the 1995 images.

Each of the relevant bands from each image was imported into Idrisi and the ***PCA*** module of Idrisi was then used to carry out a principal component analysis of each image. The PCA process produces a series of bands, where each contains data that are completely uncorrelated with the others bands. This is in contrast with the original TM bands where correlation levels are generally high because land-cover types often have similar reflectance values. The high levels of correlations between the bands typically mean that the first two or three PCA images often explain more than 95% of the information of the original bands (Eastman 1999).

In contrast, the atmospheric conditions that produce haze do not reflect and absorb different light wavelengths in the same way and are seldom correlated. This means that the information in the reflection values caused by haze and often striping (caused by a detector going out of adjustment in a given band) tend to be contained in separate PCA images. The PCA images can be used to reconstruct the original TM bands, based on the results of the PCA, and so the haze and striping can be removed by excluding their associated PCA images when reconstructing the new bands. Therefore, the PCA bands and results table were used to reconstruct the original bands using the ***Image Calculator*** in Idrisi to produce bands that were less affected by atmospheric haze and recording errors.

Despite these improvements, it was still difficult to distinguish between the land-cover types in the different bands. This was because the values recorded by Landsat satellites are converted using a linear transformation to Digital Numbers (DNs) between 0 and 255. These DNs are also converted into integers to minimise the digital space needed to store them. Unfortunately, this means that if one pixel in a band has an exceptionally high or low value, then the remaining pixels will have very similar DN values after the linear transformation. These pixels with very different values tend to be produced by errors in the detection or recording process, and they are not uncommon in a TM band, given the large number of pixels they each contain.

This PCA method was used to increase the visual contrast of the different bands used in this analysis. This was achieved using the "linear with saturation" option of the ***STRETCH*** module in Idrisi. This identifies the low and high cut-off values for each image and reclassifies the lowest DNs as 0 and the highest DNs as 255. The remaining DNs are then stretched between the values 1 and 254. The low and high cut-off values were decided by examining DN histograms from each band.

The final stage in preparing the Landsat TM and ETM+ images was to geo-register each band so that they accurately represented the land-cover on the ground. This was done by identifying points that were visible both on the satellite images and on the available roads and rivers coverages. At least 35 points were identified for each image and the ***RESAMPLE*** module in Idrisi was used to geo-register each band of each image. From this, the root-mean-square (RMS) error was estimated to compare the sample of measurements with their true values, so that the accuracy of the geo-corrected image could be evaluated, and extra identifying points used where necessary.

4.2.1.2 Colour composite images

Colour composite images were created using the ***COMPOSIT*** module in Idrisi on all the images. An RGB (Red-Green-Blue) composite image was produced from each TM image by combining bands 5, 4 and 2 in this order.

4.2.2 Producing the GIS coverages

A variety of methods were used to produce the KS GIS coverages and these are described below. In each case the coverages were modified to have the same geographic reference system, WGS 84 UTM-47s. Eight physical and four socio-economic factors were mapped.

4.2.2.1 Physical factors

The physical coverages constructed for the KS region were based on data obtained from *Bakosurtanal* (National Coordination Agency for Surveys and Mapping) or collected during this study. The physical factors included: elevation; slope; protected area status; soil; public roads; logging roads, rivers, and settlements. The latter four coverages were converted into proximity maps.

The digital elevation model (DEM) was based on 100 m interval contour maps that had been digitised from 1:50 000 paper maps produced by *Bakosurtanal*. These contour lines were converted into TINs format using the "3D Analyst" extension in ArcView. This was then converted to a 100 m raster format and exported into Idrisi. The slope coverage was derived from the DEM using the ***SURFACE*** module in Idrisi.

The initial soil coverage was digitised from a 1:100,000 paper map obtained from the Indonesian Inter Spatial Province plan. The soil map followed the US based system of soil classification. A list of the soil types found in the study area was sent to Dr Jan Hof (KS ICDP Geology specialist) who grouped the soils according to their agricultural potential based on drainage, fertility, and composition. This information was used to re-classify the original soil map into two categories ("poor" and "good" agricultural potential) and this was then converted to a raster format in ArcView.

The locations of the KSNP boundary markers (pal batas) were digitised from 1:50,000 maps produced by the KS ICDP, to produce the protected area status map. Data on commercial forest sectors were obtained from *Bakosurtanal.* For the KS region these included logging concessions (HPHs) and estate crop plantations boundaries. More accurate data on the HPHs were obtained from the KS ICDP.

The road coverage was based on data digitised from 1:50,000 paper maps produced by *Bakosurtanal.* This was imported into Idrisi, converted to a raster format and the ***DISTANCE*** module was used to produce the distance from roads coverage. The logging roads coverage was constructed from global positioning system (GPS) location data collected from field surveys. Logging roads were also identified from 2000 and 2001 satellite images, converted into the GIS by on-screen digitising, and checked in the field using a GPS unit. These data were imported into Idrisi GIS software, converted to raster format and the ***DISTANCE*** module used to produce a coverage for the distance from logging roads. The positions of river were derived from 1:50,000 digitized maps from *Bakosurtanal.* River layers were converted to a raster format and then the ***DISTANCE*** module in Idrisi used to produce a coverage for the distance from rivers.

A map of village settlements was obtained from *Bakosurtanal.* This dataset was a series of points, whereby each point corresponded to the nucleus of a single village polygon. Given the small size of the settlement polygons and the large size of the study area, using a point location as a settlement polygon proxy was considered feasible in producing the final distance map. Each point was converted into a raster format and the ***DISTANCE*** module in Idrisi used to produce the coverage for distance from settlements.

4.2.2.2 Socio-economic factors

Datasets on socio-economic factors were obtained from the Indonesian 2000 village population census (*survei potensi desa, PODES 2000*) provided by the Indonesian Central Statistics Bureau (*Biro Pusat Statistik, BPS*). These datasets contained all of the administrative boundaries for villages, sub-districts, districts, and provinces in the KS region. The village boundary data were used to produce the socio-economic coverages of economic activity, capital availability, village development, and distance from market.

Economic activity can be measured through the size of the productive labour force, which is usually defined as the number of 17–55 yr olds in a population. This information was not recorded in *PODES* 2000, which instead recorded the number of registered voters (persons 17 yrs +) for each village. Although these figures include those over 55 yrs old, it was still considered appropriate, firstly because the average age of mortality across the KS region was 65.9 yrs (BPS 1998), and secondly because many villagers over 55 yrs continue to work as there is no other real alternative (pers. obs). The number of registered voters was divided by the village area to derive the density of the village productive labour force.

The density of satellite dishes in a village was used as an indicator of capital availability. This index was calculated by dividing the number of satellite dishes (*parabola*) in a village by the number of households (*rumah tangga*). Similarly, the proportion of households that received electricity (*listrik PLN*) in a village was used as an indicator of village development. The average time from each household to the nearest market in a village was converted to minutes and used in the analysis.

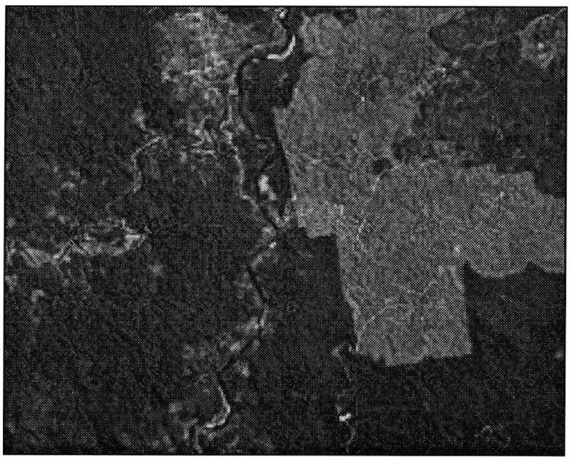
4.3 RESULTS

4.3.1 Remote sensing coverage

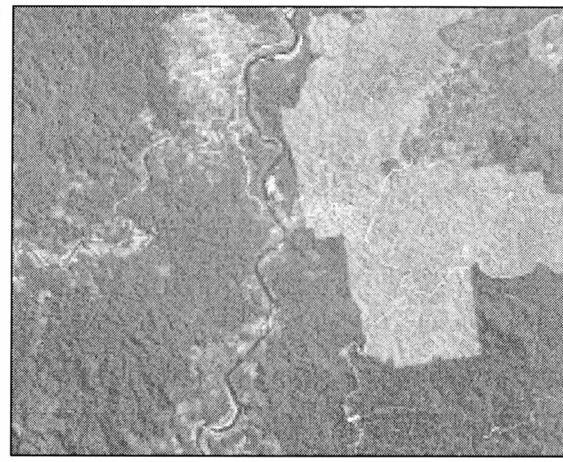
4.3.1.1 Landsat image enhancement

The visual contrast in each band was greatly improved using a PCA and contrast stretching on the Landsat images (Figure 4.1). The mean and standard deviation DN value for each band increased after stretching, thereby resulting in less skewed data spread more widely between 0 and 255.

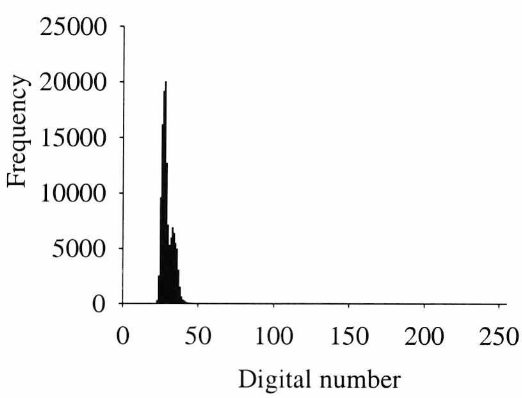
Band 2 before stretching



Band 2 after stretching

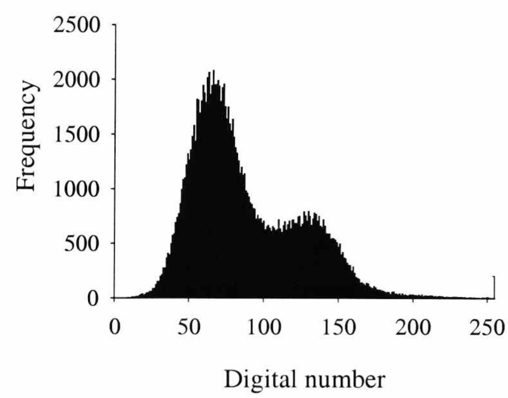


Band 2 before stretching



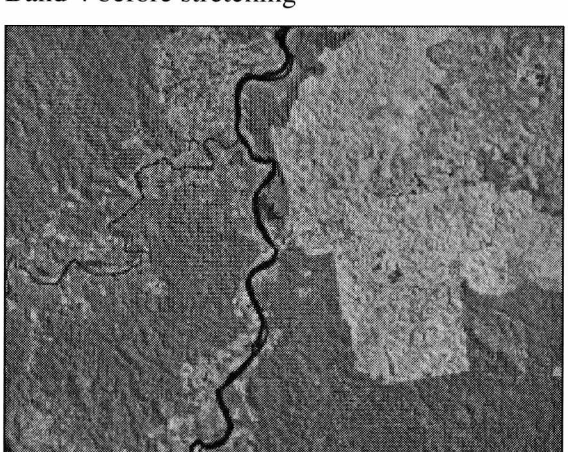
Mean = 29.47, Standard deviation = 3.74

Band 2 after stretching

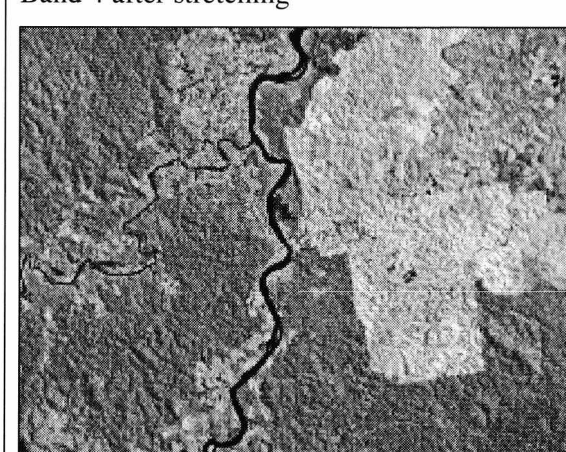


Mean = 88.12, Standard deviation = 36.81

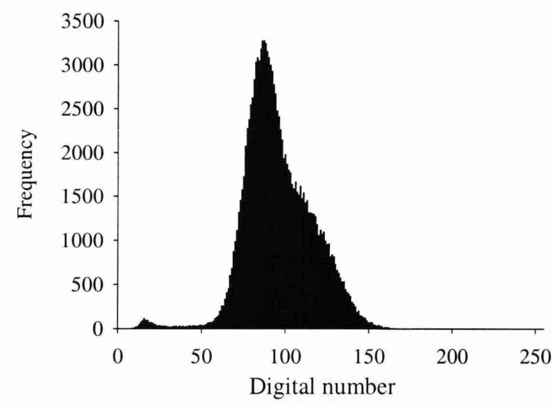
Band 4 before stretching



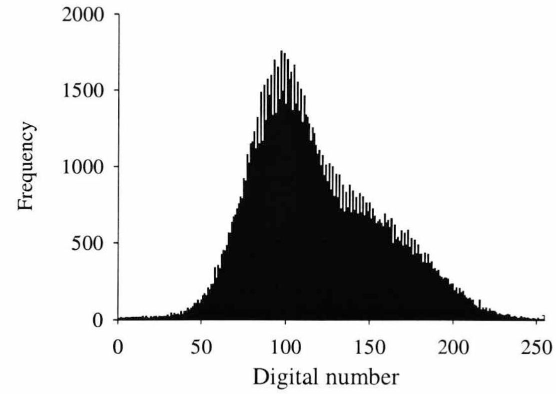
Band 4 after stretching



Band 4 before stretching



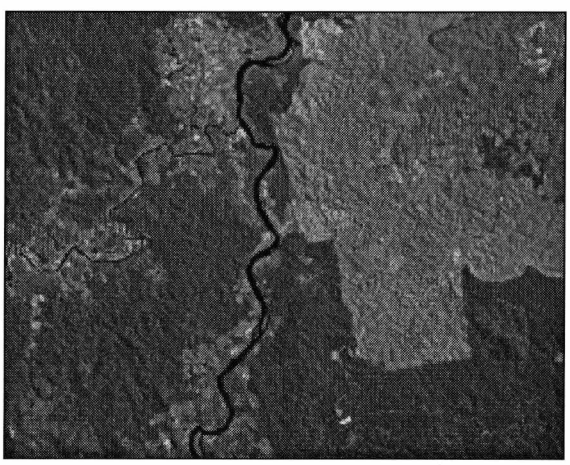
Band 4 after stretching



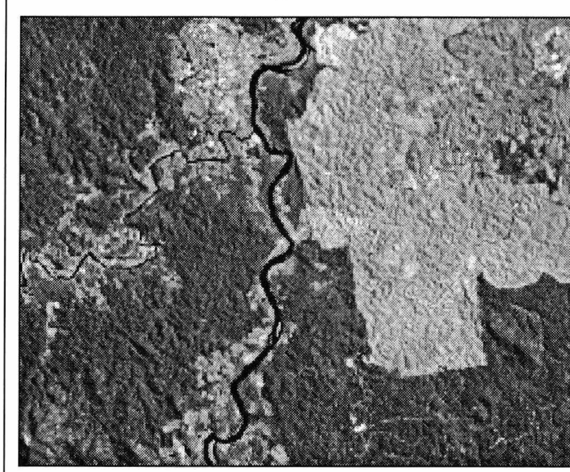
Mean = 95.62, Standard deviation = 20.42

Mean = 117.20 Standard deviation = 40.49

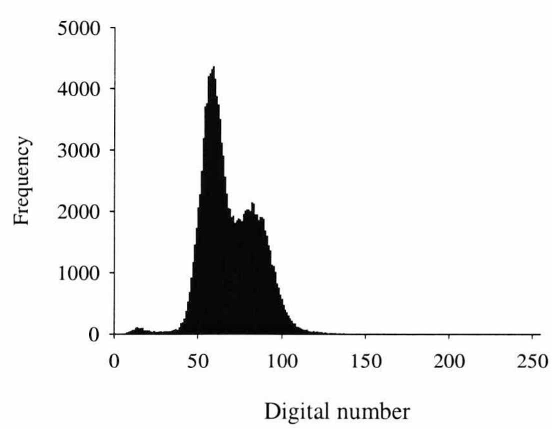
Band 5 before stretching



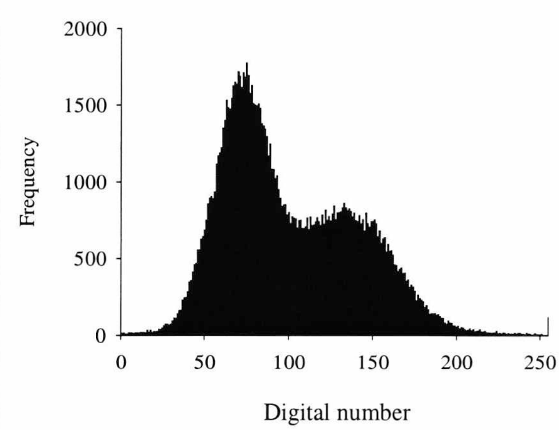
Band 5 after stretching



Band 5 before stretching



Band 5 after stretching



Mean = 68.67, Standard deviation = 16.42

Mean = 99.87, Standard deviation = 40.85

Figure 4.1: Three examples of the effects of contrast stretching on visual distinctiveness and reflectance values on a) Band 2; b) Band 4; and c) Band 5.

4.3.1.2 Colour composite image

The colour composite image constructed from the enhanced Landsat band gives much clearer definition to the different land cover types. The colour black indicates water, dark green is forest, light green and purple are estate crop plantations, and pink is small scale subsistence farmland (Figure 4.2). Nineteen radio- and geo-metrically corrected colour composite images were created. The data were resampled using the nearest neighbour technique with a linear (first order) mapping function that had a mean RMS error of 82.1 (SD = 8.34).

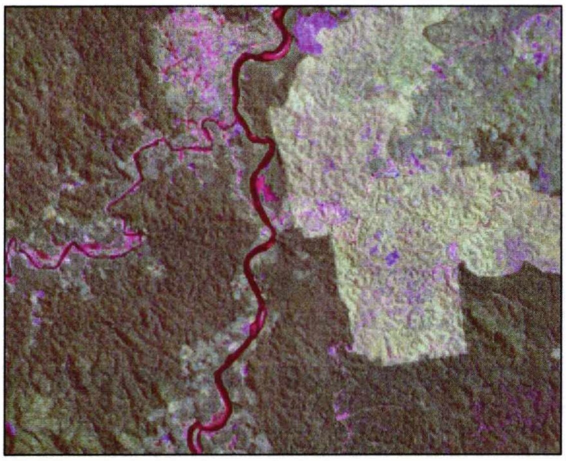


Figure 4.2: Extract from a colour composite map combining Landsat TM bands 5, 4, and 2

4.3.2 GIS coverage

The physical coverages produced an altitudinal range for the KS region, starting at sea level by the west coast, and rising to 3,805 m at the peak of Mount Kerinci located in the centre of the region (Figure 4.3).

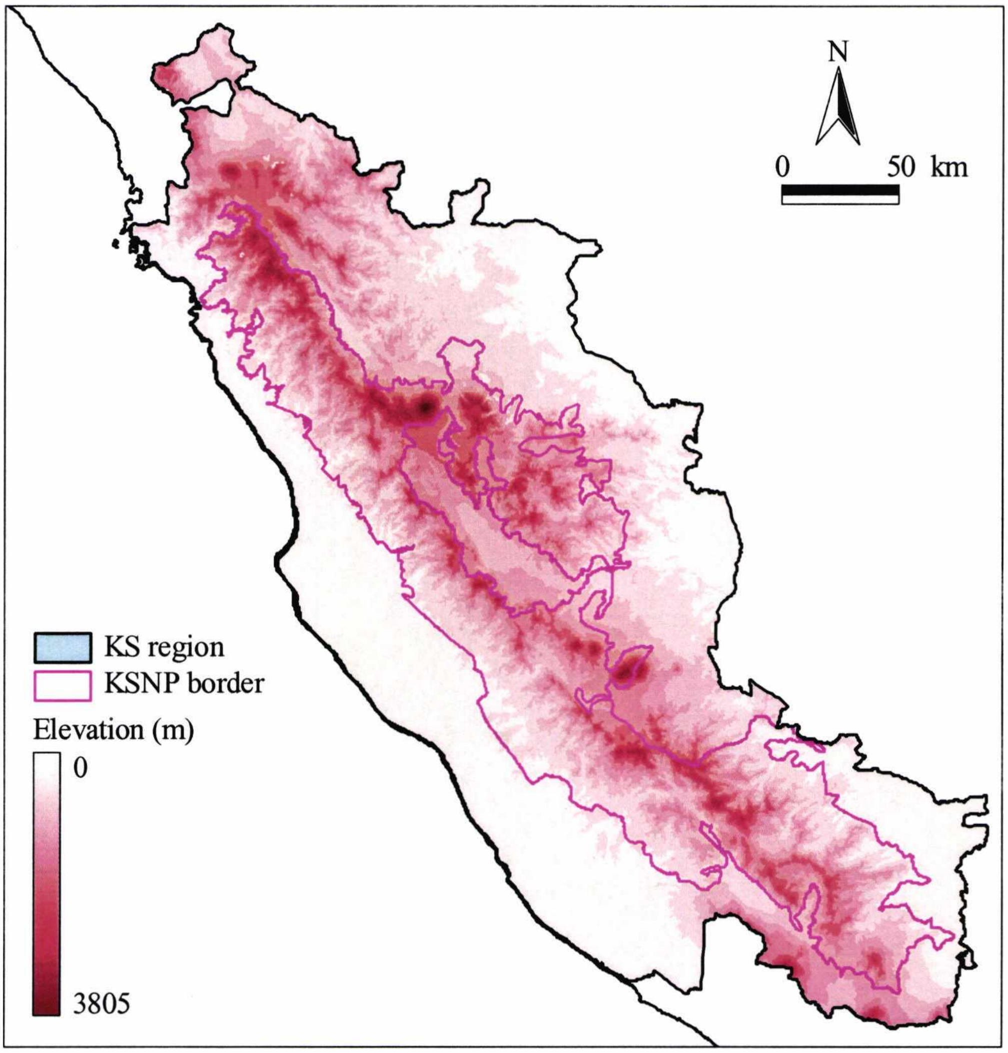


Figure 4.3: Digital elevation model across the KS region showing the boundary of KSNP

The average elevation of forest for the whole KS region is 829 m (Figure 4.4a). The western range forest is higher on average (875 m) than the eastern range forest (785 m) (Figures 4.4b and 4.4c). The average elevation for forested areas inside KSNP is 982 m (Figure 4.4d) as the topography outside of KSNP is generally flatter than the topography inside (Figure 4.5).

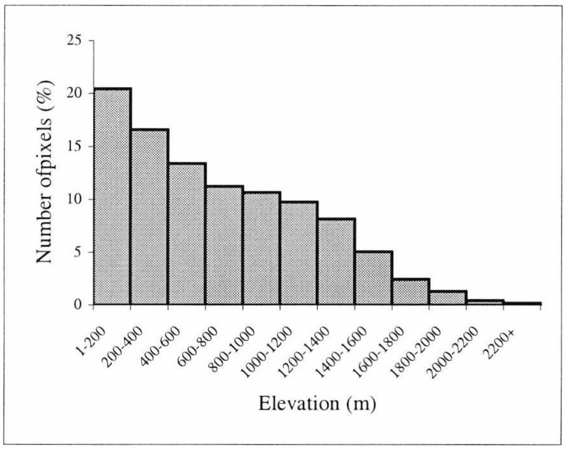


Figure 4.4a: Altitudinal distribution of forest for the KS region

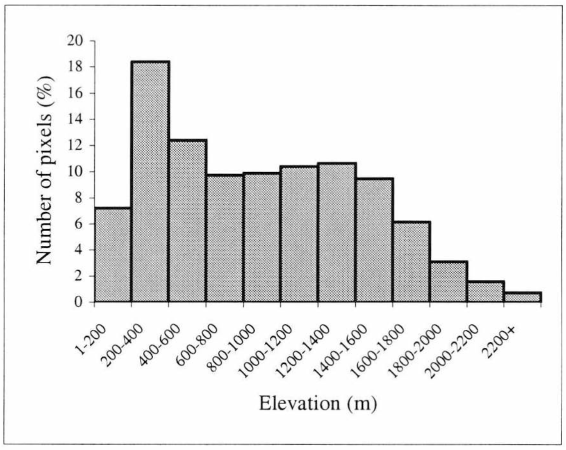


Figure 4.4b: Altitudinal distribution of all forest in the western range

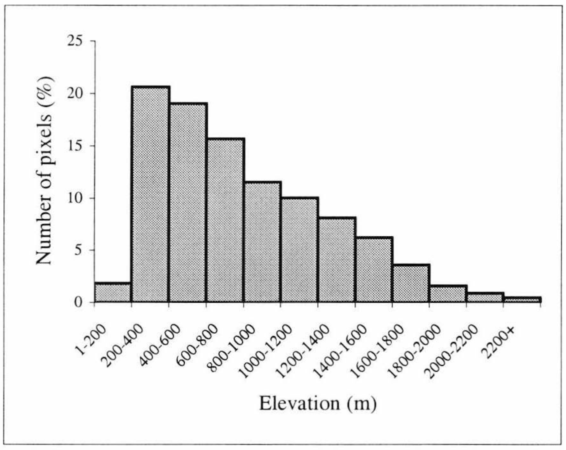


Figure 4.4c: Altitudinal distribution of all forest in the eastern range

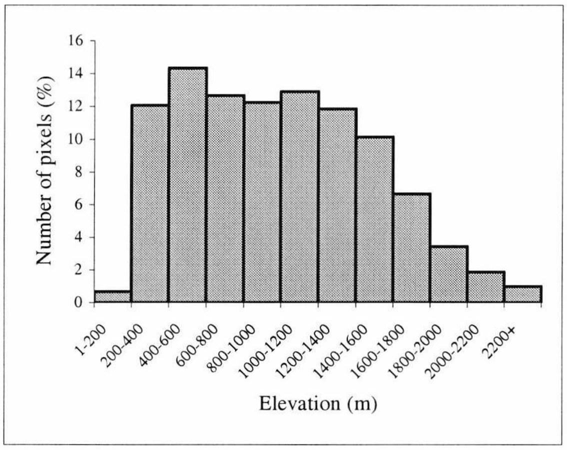


Figure 4.4d: Altitudinal distribution of forest within KSNP

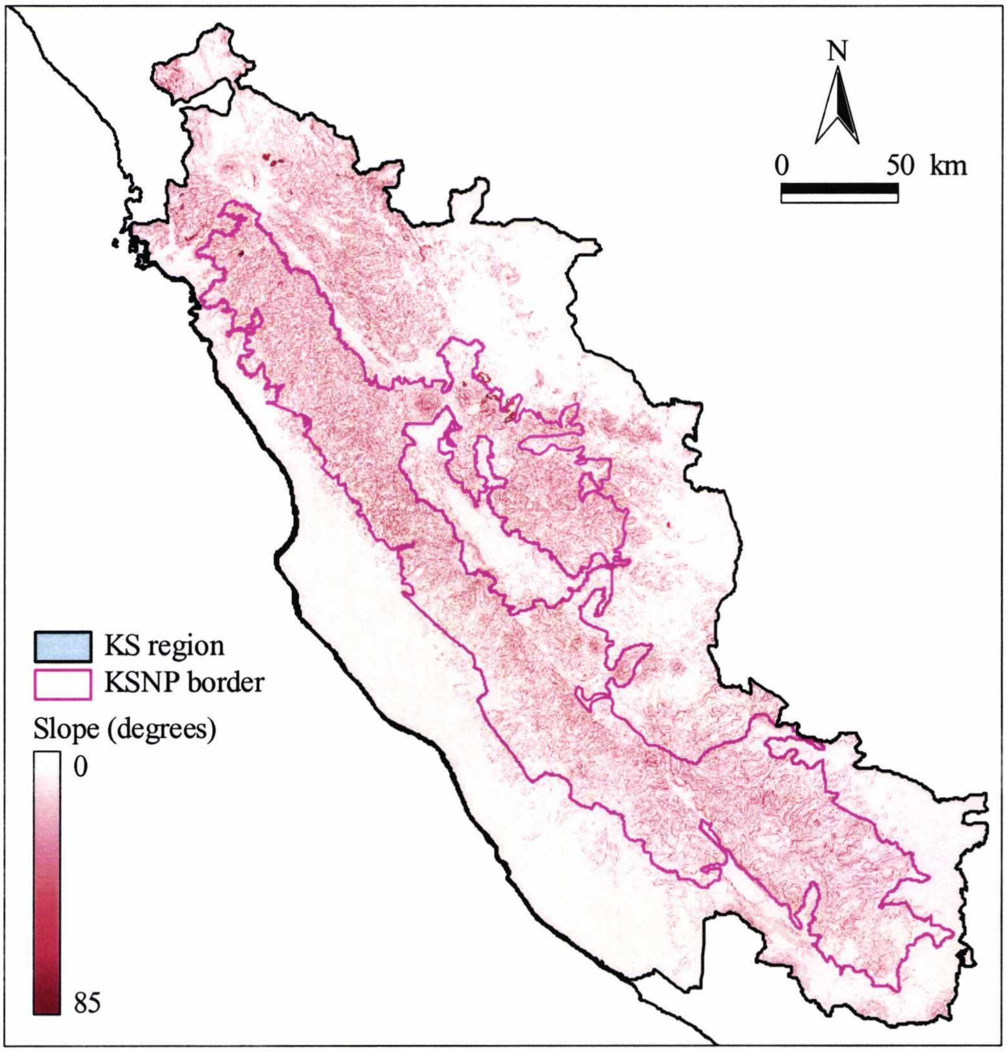


Figure 4.5: Slope across the KS region showing the boundary of KSNP

In the central and southern sections of the KS region, HPHs adjoin most of the KSNP border. In the northern section of Pesisir Selatan and Solok, the reverse is true. The few estate crop plantations in the region are located in Bengkulu Utara in the southwest and Solok in the northeast. While there is overlap between the boundaries of the plantations and the HPHs, there are plantations bordering KSNP. Roads and logging form a diffuse network in the KS region making most areas outside of KSNP accessible (Figure 4.6).

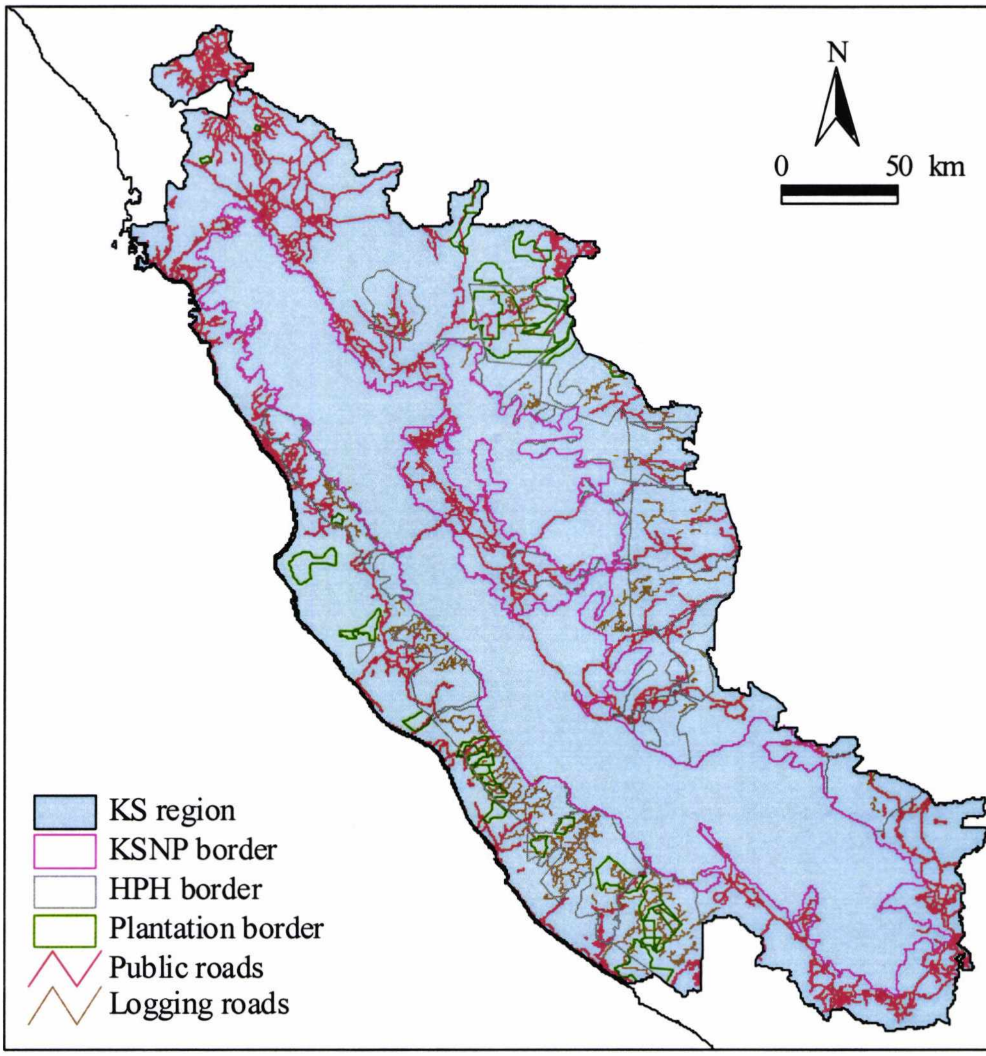


Figure 4.6: The position of HPHs, estate crop plantations, roads and logging roads across the KS region showing the boundary of KSNP

The position of settlements is concentrated around the northern section either side of KSNP, in the central enclave, and around the southern tip. Over the rest of the KS region settlements are spread thinly and are peripheral to forested areas (Figure 4.7).

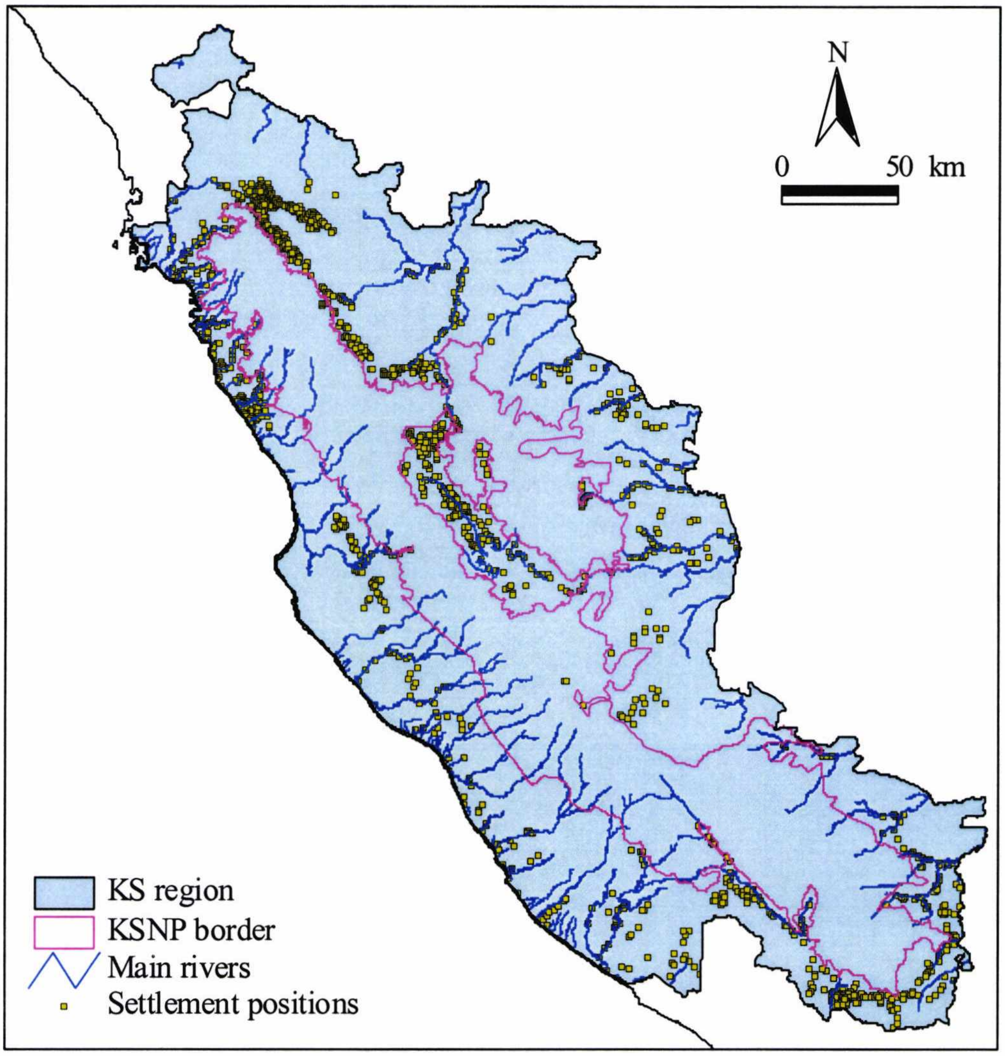


Figure 4.7: The position of settlements and rivers across the KS region showing the boundary of KSNP

Seven soil types were classified as having 'good' agricultural potential and five as having 'poor' agricultural potential (Table 4.2). Based on this classification the KS region comprised 13,686.1 km^2 of 'good' soil and 47,426.7 km^2 of 'poor' soil (Figure 4.8).

Table 4.2: Details of the soil types, their area, and their agricultural potential found in the KS region

Soil type	Area (km^2)	Agricultural potential	Soil type	Area (km^2)	Agricultural potential
Dystrandepts	4229.0	Good	Humitropepts	1145.5	Good
Dystropepts	30265.8	Poor	Kandiudults	1151.7	Poor
Eutropepts	1175.3	Good	Paleudults	1463.5	Good
Haplohumults	1908.4	Good	Tropaquepts	12739.6	Poor
Hapludox	2957.2	Poor	Tropopsamments	312.4	Poor
Hapludults	2637.1	Good	Troposaprists	1127.3	Good

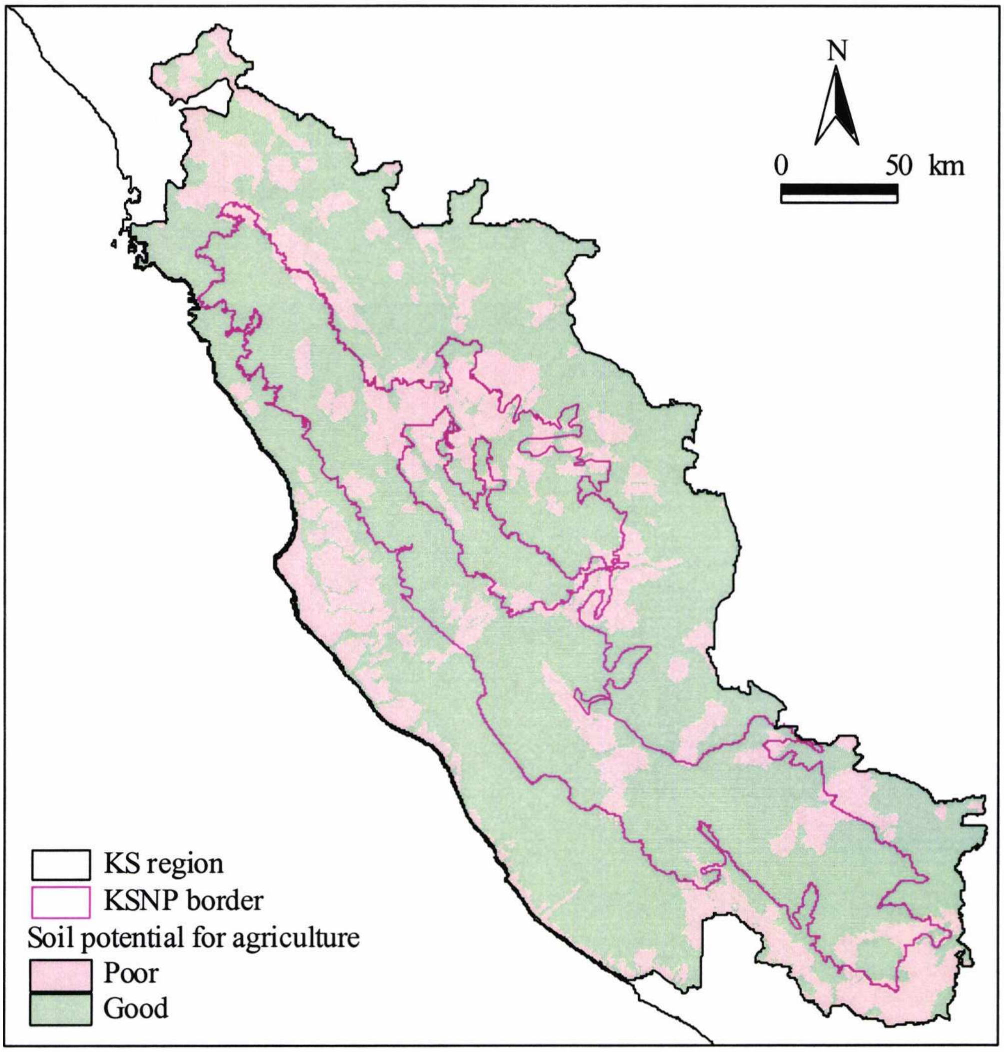


Figure 4.8: Agricultural potential of soil in different areas across the KS region

Based on the density of the productive labour force, the central Kerinci enclave, the northern tip and southern tip of the KS region were found to have the highest (Figure 4.9).

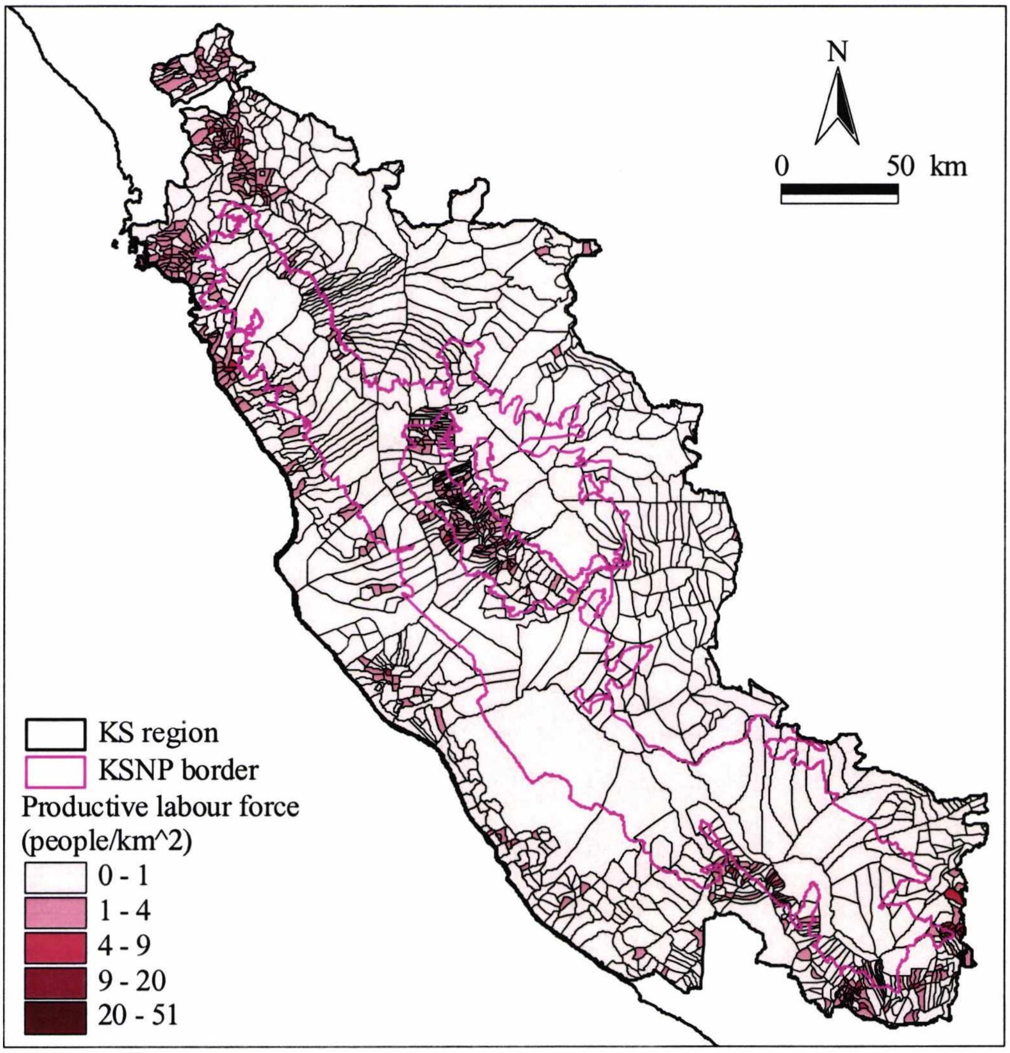


Figure 4.9: Economic activity in different village areas in the KS region based on the density of the productive labour force

Based on the proportion of satellite dishes, the villages with greater capital availability were mainly clustered in the Kerinci enclave and central sections of the region. In the northern and southern sections, where there were higher concentrations settlements, there were also clusters of villages with greater capital availability (Figure 4.10).

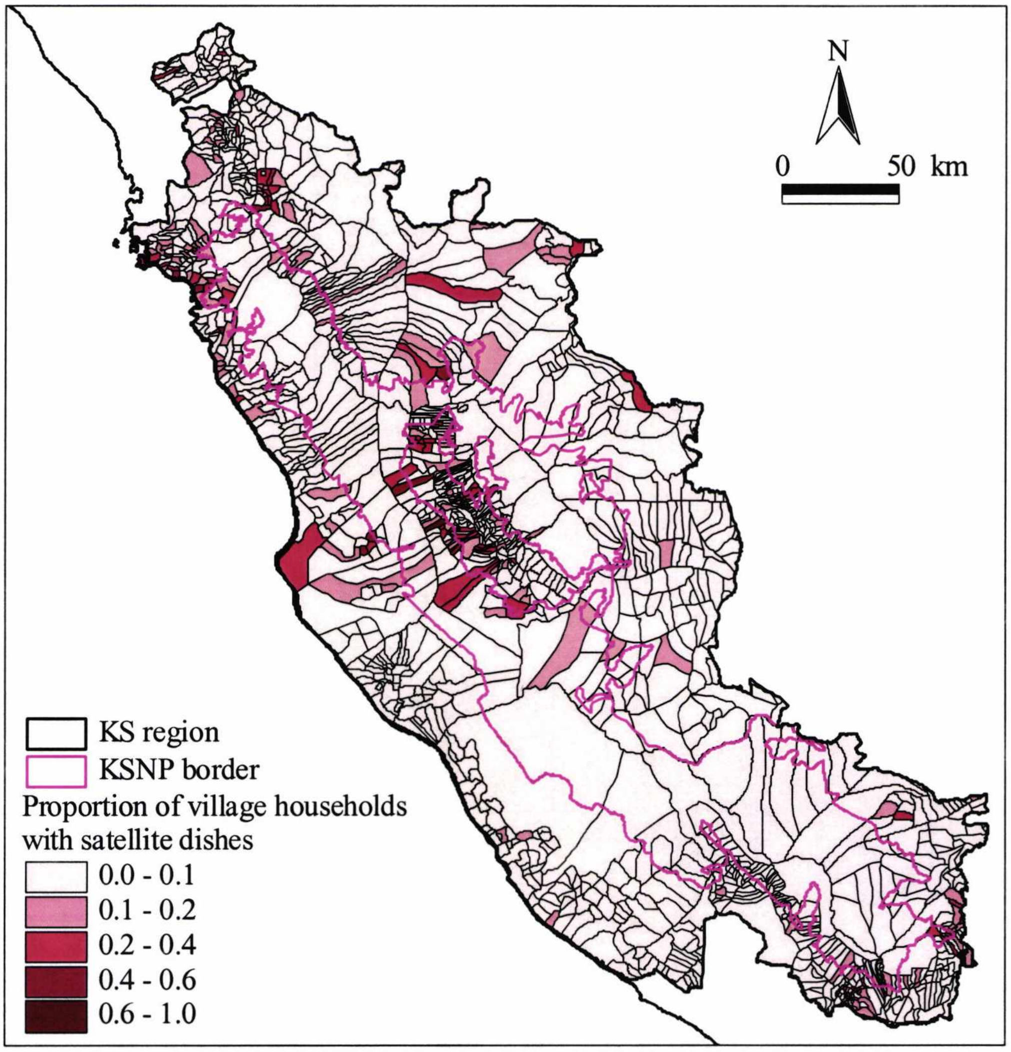


Figure 4.10: Index of capital availability in the KS region based on the proportion of households with satellite dishes

Based on the proportion of households with electricity the villages in the central and northern sections of the region tended to be much more developed than those in the eastern and south-eastern sections. In the far south village development was similar to that in the north (Figure 4.11).

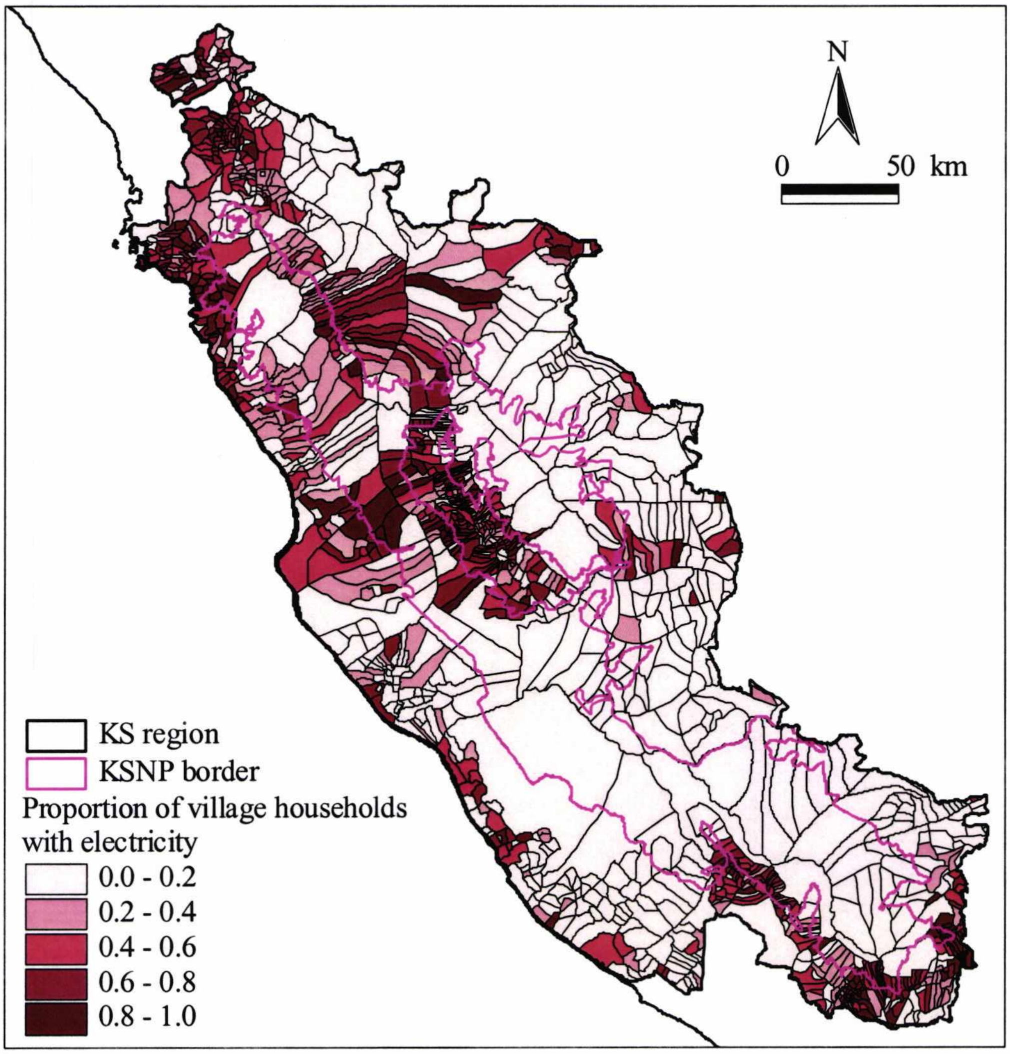


Figure 4.11: Index of development in the KS region based on the proportion of households with electricity

Villages along the east side of the region had to travel for a longer time to reach the nearest market than those located along the west coast. The central east section was provided with only a few public roads in comparison with areas than had a shorter travel time (Figure 4.12).

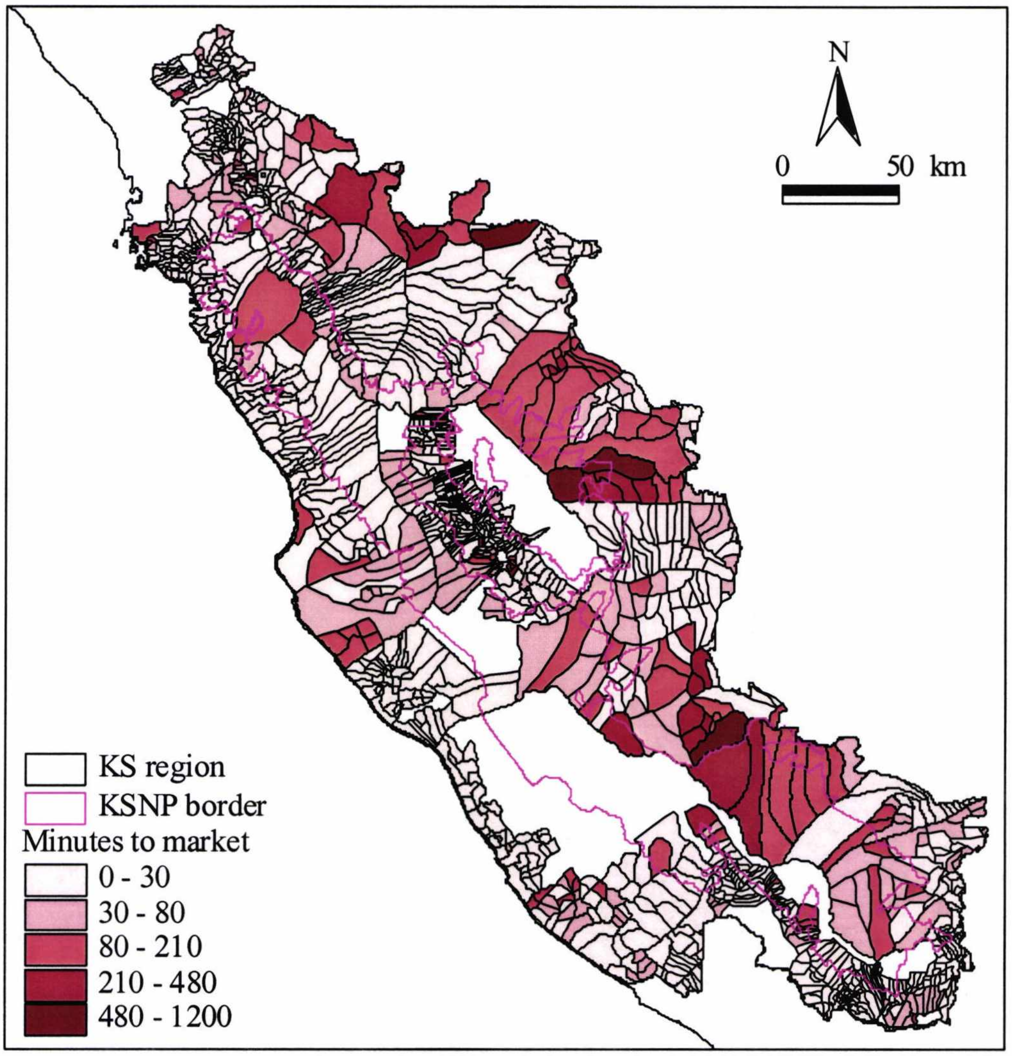


Figure 4.12: Distance to the nearest market in the KS region

4.4 DISCUSSION

The image enhancement arising from contrast stretching removed the skewed distribution of the grey level so that the amount of information was greatly increased. This enabled differences between pixels, and therefore land types, to be distinguished visually. The DN value histograms for the bands before manipulation exhibit a long "tail" because the bands contain a small number of pixels with very high and very low DNs. Therefore, the linear transformation gives the majority of pixels very similar DN values, despite possibly large differences in their wavelength reflectance properties. This obviously makes any land-cover classification based on these images much more prone to errors, and there are great benefits in carrying out some type of "contrast stretching" to correct this problem. However, the RMS error recorded was permissible, given the image resolution (100 m) and the wide dispersion of the geo-referencing points.

The methods used have produced the GIS maps and data that will be used in subsequent analyses. Such data are new and have not been previously created for KSNP. The maps and data so created will be used in turn to map forest cover and forest cover change in the KS region and in the different forest sectors.

Chapter 5
FORESTS AND THE FOREST SECTOR


Moss forest on Mount Tujuh inside KSNP (J. Holden)

5.1 INTRODUCTION

Indonesia is endowed with one of the largest expanses of tropical humid forests in the world. Although not entirely clear, forest cover is estimated at around 95-100 million ha (World Bank 2000, FWI/GFW 2002). These forests are important nationally for their economic and social values, and internationally for their biodiversity and carbon sequestration values.

The forests on the Indonesia archipelago have a remarkably rich biodiversity with high levels of endemism. It was therefore noteworthy that Indonesia was among the first signatories to the 1994 Convention of Biological Diversity. Indonesia was also one of the first countries to prepare and implement a National Biodiversity Strategy and Action Plan, paving the way for financing priority protected areas (PAs) and expanding its PA system. However, there were still problems with implementing this strategy as only one PA, namely KSNP has been fully gazetted. Furthermore, most national parks remain under-funded, and designated money is not spent effectively or transparently. The biggest threat to Indonesia's forests came in 1997 when the country's 'miracle growth' during the 1970s and 1980s came to an abrupt halt.

During the financial crisis Indonesia turned to its traditional economic base in natural resources to fuel its economic recovery. Unfortunately, this led to high and unprecedented levels of illegal logging and to the deliberate setting of forest fires that quickly spiraled out of control. Many of the fires in Indonesia were lit to clear forested areas for palm oil production or shifting cultivation. The forest fires primarily occurred on the Indonesian islands of Sumatra and Borneo (Kalimantan) and were further fuelled by an El Niño Southern Oscillation induced drought (Stolle and Tomich 1999, Siegert et al. 2001).

Between 1997 and 1998 these fires affected about 47,000 km^2 of lowland and peat swamp forest in Indonesia (BAPPENAS-ADB 1999). They produced carbon dioxide emissions equal to the emissions produced by Europe for a single year. These widespread fires resulted in dense haze across Southeast Asia, causing respiratory health problems, a substantial decrease in the region's tourism, transportation delays, and accidents on land, air and sea. The economic costs were estimated between

US$2.3 and US$3.2 billion, or up to US$6 billion if carbon emissions are included (Taconi 2003). Whilst large amounts of money were made from illegal logging or turning forest into oil palm production, no one was brought to court.

To emerge from this financial crisis, Indonesia further exploited its natural resources, upon which it still depends on. Previous Governments pledged to bring forest production under sustainable management by 2000, but towards the end of the 1990s it was estimated that over 70% of Indonesian log production came from illegal sources (EIA 1999). The annual burning of rainforests on Sumatra and Kalimantan continue (Jakarta Post 2003[1, 2]). So far, there has been poor governance of the forest sector.

5.1.1 Forest sector

The Basic Forestry Law in Indonesia was began in 1967 and enabled foreign and domestic private companies to extract timber from the forest-rich Outer Islands. The present Indonesian forest sector was established under the 1984 Forest Land Use Plan (*Tata Guna Hutan Kesepakatan* or *TGHK*). Provincial governments assigned forest for either protection, conservation, production, or conversion. These forest use boundaries were later re-evaluated in 1992 under the Spatial Management Plan. This resulted in an increase of protection forest (from 300,000 km^2 to 350,000 km^2), no significant change in conservation forest (still 190,000 km^2), and a decease in production forest (from 640,000 km^2 to 340,000 km^2). However, the biggest change occurred in the conversion forests, used for transmigration resettlements and tree crop plantations, which decreased from 300,000 km^2 to 80,000 km^2.

5.1.2 Commercial forest sector

Commercial timber harvests have been the dominant concern in the implementation of Indonesian forest policy. The management of the forest sector has catered to the commercial timber industry based on a system of forest concession rights (known as *Hak Pengusahaan Hutan,* or HPH), industrial forest or timber plantation concessions (known as *Hutan Tanaman Industri,* or HTI), and estate crop plantations. The concessions are licensed to private enterprises or to special state-owned enterprises (known as *Badan Usaha Milik Negara,* or BUMN), some of which are responsible for rehabilitating revoked concessions. The concession system epitomizes the political patronage in Indonesia: politically well connected individuals accrue substantial financial gains (Barber 1997). These schemes, to their detriment, rarely recognized local communities as interest groups (World Bank 1993).

5.1.2.1 Logging concessions

The Basic Forestry Law provided the Ministry of Forestry and Estate Crops (MOFEC) the authority to grant HPH *timber concession* licenses in areas designated as production and limited production forests. This previously granted the HPH holder a license for 20 years under the proviso that the concessionaire follows the principles of sustainable forest management as prescribed by the Indonesian selective logging and planting system. The licenses were intended to maintain the forest as permanent production forest. However, the management activities of 13 HPHs monitored in northern Sumatra showed that eight were being converted to cultivate oil palm (FWI/GFW 2002)

5.1.2.2 Industrial forest concessions

HTIs were established inside production forest and were granted permits to clear designated areas that would then be replanted with commercial tree species (Barr 2001). There are three distinct types of HTI: pulpwood plantations, non-pulp, and HTI-transmigration. The converted forestland was used for transmigrant settlements. Government sponsored and spontaneous transmigration programmes resulted in large net migration outflows from densely populated Java (945 people/km^2) to neighbouring islands such as Sumatra (88 people/km^2) and Kalimantan (20 people/km^2) (BPS 2000). The transmigration programme has been heavily criticized because transmigrants resorted to slash and burn practices, either for lack of adequate land, lack of appropriate agricultural skills, or poor soil productivity (World Bank 2000).

5.1.2.3 Estate crop plantations

From 1996 to 2001 oil palm production increased by 81% to 225,430 km^2 (BPS 2000). This increase has been well supported by changes in Indonesian Government policy, such as decreases in oil palm export taxes, permit revocation for failure to develop estates, and state forestry companies granted permission to convert 30% of their concession to oil palm. Initially oil palm estates were established on converted forestland, as Government regulations stipulated.

However, the existing rules of land allocation and forest classification are widely ignored. The process by which forest areas are declared conversion forest is not transparent. The problem is perpetuated by the lack of clarity about boundaries between conversion and non-conversion forests, and variable definitions of what constitutes a conversion forest. Between 1997 and 1998, primary forest was deliberately burnt to render it as degraded forest with reduced conservation value, and making it permissible for oil palm conversion.

5.1.3 Protected forest sector

The protected forest sector in Indonesia contains an official (protection forest and protected areas) and unofficial (customary forest) element. The latter has no legal status and is often ignored by government.

5.1.3.1 Protection forest

Protection forests (*Hutan lindung*) were primarily created to maintain and protect vegetation cover, soil stability on steep slopes and watershed areas. Protection forest is not available for commercial logging or conversion for other commercial activities, although this is known to happen. Urban small investors (section 2.5.5) were considered responsible for the substantial deforestation occurring in Bukit Seligi Protection Forest in Riau and Ogan Komering Ilir in South Sumatra (Riau Post 1999, Kompas 1999).

5.1.3.2 Protected areas

Even though less than 10% of Indonesia's land is designated as conservation areas, it does have a fairly well designed and biogeographically representative PA system (Jepson and Whittaker 2002). Nearly all PAs are under-funded and poorly managed, but the human pressure on the natural resources did not create a problem until recently. After decentralization this changed and illegal logging for timber and land for agriculture now threaten most conservation areas.

5.1.3.3 Customary forest

The customary forest or *hutan adat* are 'tribal lands' that have been passed on through generations of indigenous peoples. They are traditionally managed according to indigenous cultural practices and regulations. These land titles are subject to customary laws, which are unwritten laws, and are therefore not formally documented (Walijatun and Grant 1996). Under the previous Basic Forestry Law, community-based rights to forest resources were recognized, in that they are left more or less alone, only so long as the state is disinterested or unable to exploit, reserve, or lease those resources itself (Lynch and Talbot 1995, Stockdale and Ambrose 1996). Hence, it has been commonplace for the state to reject or neglect community-based rights to forest resources where state interests prevail. However, the recent decentralization of government (Act No. 22/1999) has allowed for more community-driven biodiversity management creates the opportunity for greater village governance, recognition of adat rights and participation in natural resource management (Bennett 2001, BAPPENAS 2003).

5.1.4 Forest distribution in the KS region

In Indonesia the amount of forest and forest change within the forestry sector has yet to be estimated from reliable data at a fine resolution (<1 km^2 at the district level). This information is necessary if the forest is to be accurately inventoried so that it can be properly managed. Important to this is the amount and change of the different forest types within the forestry sector, such as Sumatran lowland forest that is predicted to have disappeared by 2005 (Holmes 2001). This chapter therefore seeks to map forest and forest change for the various forest sectors in the KS region.

5.1.5 Aims and objectives

This chapter aims to:

- Map forest cover and forest change in the KS region between 1995 and 2001;
- Calculate the amount of lowland, hill, submontane, and montane forest in the KS region;
- Calculate the amount of forest change in the different forest sectors for each district;
- Determine the forest types most under threat in the KS region;

- Determine the two most threatened forest types for each district and each forest sector.

5.2 METHODS

The composite images produced in Chapter 3 were converted to .jpg format using the ***JPGIDRIS*** module in Idrisi, displayed in ArcView and used to on-screen digitise the position of forest and cloud on the images. Deforestation in the study area was often an incremental process, and so sometimes there was no distinct boundary between forest and the neighbouring degraded forest. In these cases, a judgement was made as to the position of the forest boundary, based on the colour and colour pattern of the forest on the images and the position of the patches of subsistence agriculture (*ladang*) that often follow deforestation. The presence of forest in areas that were covered by cloud on all the relevant images was decided by using another forest coverage developed from 6 Landsat images taken in 1994, 1996, 1997, 1999, and 2000.

5.2.1 Forest coverage

Forest cover maps were constructed by the on-screen digitising of colour composite images. The interpretation of the 2001 forest cover map was ground-truthed by checking the accuracy of areas classified as forest and non-forest and modified to increase their reliability.

The 1995 forest coverage was constructed by manipulating a copy of the 2001 forest coverage and adding blocks of forest that were present in 1995 and not in 2001. The position of cloud within the forest patches in the 1995 images was also digitised and the final coverage was imported into Idrisi and rasterised.

5.2.2 Forest type coverage

Forest types were classified based on Laumonier's (1994) system developed for KSNP (Table 2.1). Detailed forest types were assigned based on their elevations and aspects (Table 2.1). Forest was mapped in detail using this system that identified different forest types in the east and west of the KS region. Therefore, the first stage in producing the forest type coverage was to on-screen digitise two polygons that covered the east and west sections of the study area. These were used to create mask coverages and the ***OVERLAY*** module in Idrisi was used to produce two new DEMs, showing the east and west sections. These DEMs were classified using the system described in Section 3.2.2.1 and combined using the ***OVERLAY*** module. To allow more direct comparisons with other studies the eastern and western forest was then reclassified into four broader forest categories: lowland, hill, submontane, and montane (Table 2.1). Finally, these were multiplied by the 1995 and the 2001 forest coverages to produce the final forest type coverages.

5.2.3 Forest change between 1995 and 2001

The 1995 and 2001 maps were overlaid to determine the location of deforestation between 1995 and 2001. Deforestation in this study was defined as total forest clearance. The annual deforestation rates (%/yr) for the whole KS region and for each study site were calculated by dividing the percentage of forest loss from the respective areas at the start of the period by the time period (in years) over which it occurred. This calculation was then applied to forest and forest loss inside KSNP, HPHs, and estate crop plantations to give their respective deforestation rates in the KS region and for each district. There were no HTIs located in the KS region.

The areas of deforestation were then overlaid on a DEM to calculate the elevation of each pixel of forest that had been cleared. These areas of forest were then reclassified into the corresponding forest types so that the amount of lowland, hill, submontane, and montane forest could be calculated for inside KSNP, HPHs, and estate crop plantations to derive their respective deforestation rates in the KS region and for each district.

5.3 RESULTS

5.3.1 Forest and forest change in the KS region

The KS region covered 38,846.8 km^2 and contained 22,327.1 km^2 of forest in 1995, representing 57.5% of the region. A total of 1278.4 km^2 of forest was cleared between 1995 and 2001 (Figure 5.1), equivalent to a mean deforestation rate of 0.96%/yr, as a result forest covered 21,048.7 km^2 in 2001, representing 54.2% of the region.

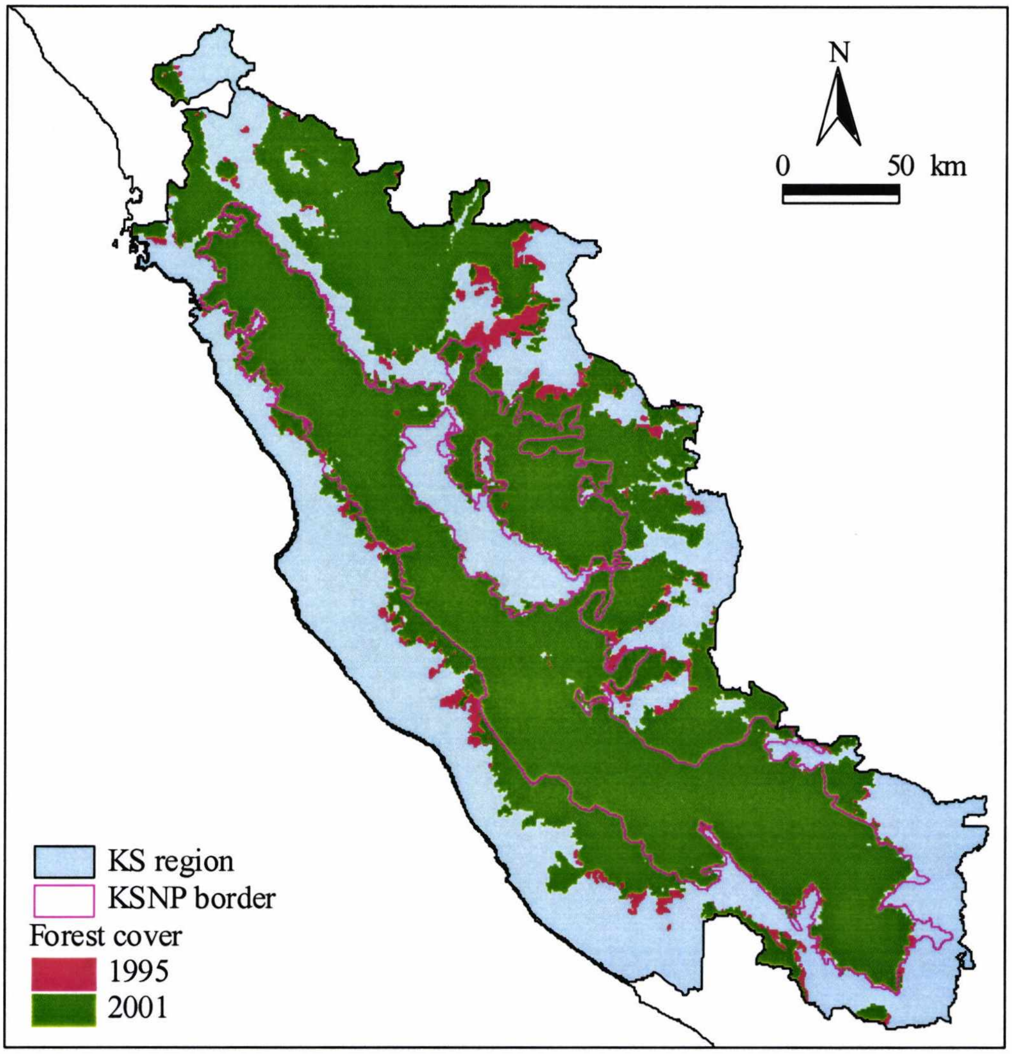


Figure 5.1: Forest cover in 2001 and forest loss between 1995 and 2001 in the KS region

5.3.2 Forest and forest change at the district level

Each district contained on average just over 50% forest in 1995 (Table 5.1). The largest districts of Solok, Bengkulu Utara, and Pesisir Selatan contained the most forest in 1995. These districts also lost the largest amounts of forest: Solok (314.7 km^2); Bengkulu Utara (194.9 km^2); and Pesisir Selatan (118.5 km^2). The annual deforestation rate for Bengkulu Utara (0.91%/yr) was similar to that recorded across the whole KS region, whereas that for Solok (1.26%/yr) was much higher. The highest deforestation rates were recorded in Bungo (1.6%/yr) and Sawah Lunto/Sijunjung (2.36%/yr), but these rates may be high because these districts were small. Large sized districts that had notably low levels of deforestation were Kerinci (0.60%/yr), Musi Rawas (0.39%/yr), and Pesisir Selatan (0.62%/yr).

Table 5.1: Change in forest distribution and cover for each district in the KS region from 1995 to 2001

		Forest cover			
District	District (km^2)	Forest in 1995 (km^2)	% district with forest in 1995	Forest in 2001 (km^2)	Deforestation 1995-2001 (%/yr)
Bengkulu Utara	7013.5	3552.8	50.7	3357.9	0.92
Bungo	1978.3	1317.9	66.6	1190.8	1.60
Kerinci	3719.2	2155.9	58.0	2078.6	0.60
Merangin	4634.5	2936.4	63.4	2743.3	1.10
Musi Rawas	4192.3	2498.8	59.6	2439.9	0.40
Pesisir Selatan	5932.5	3189.7	53.8	3071.2	0.62
Rejang Lebong	2864.4	1388.8	48.5	1283.3	1.27
Sarolangun	667.7	584.6	87.6	572.1	0.35
Sawah Lunto/Sijunjung	1129.6	536.2	47.5	460.4	2.37
Solok	6714.8	4166.0	62.0	3851.3	1.27
Total	38846.8	22327.1		21048.7	

5.3.3 *Forest and forest change in KSNP*

KSNP contained 12,657.7 km^2 of forest in 1995 (Table 5.2). Inside KSNP, the total amount of forest cleared between 1995 and 2001 was 207.5 km^2, equivalent to a mean annual deforestation rate of 0.28%/yr. In 2001, KSNP was reduced to 12451.0 km^2 of forest. The highest rates of forests clearance inside KSNP were in Rejang Lebong (0.68%/yr) and Solok (0.53%/yr), but these rates were still lower than the average across the whole KS region.

Table 5.2: Change in forest distribution and cover for each district in KSNP from 1995 to 2001

	Forest cover				
District	KSNP (km^2)	Forest in 1995 (km^2)	% KSNP with forest in 1995	Forest in 2001 (km^2)	Deforestation 1995-2001 (%/yr)
Bengkulu Utara	2178.6	2163.0	99.3	2155.9	0.05
Bungo	339.1	334.3	98.6	334.0	0.02
Kerinci	2305.5	2054.5	89.1	2002.7	0.42
Merangin	1524.5	1453.4	95.3	1428.2	0.28
Musi Rawas	2452.4	2240.8	91.4	2210.3	0.23
Pesisir Selatan	2640.0	2547.0	96.5	2523.7	0.15
Rejang Lebong	1285.7	1154.1	89.8	1107.0	0.68

Sarolangun	3.7	3.7	100.0	3.7	0.00
Sawah Lunto/Sijunjung	35.2	34.0	96.6	34.0	0.00
Solok	797.1	673.0	84.4	651.6	0.53
Total	13561.7	12657.7	Mean = 94.1	12451.0	Mean = 0.24

5.3.4 Forest and forest change in HPHs

The HPHs contained 4805.9 km^2 of forest in 1995 (Table 5.3). Inside HPHs the total area of forest cleared between 1995 and 2001 was 681.5 km^2, equivalent to a mean deforestation rate of 2.96%/yr. In 2001, HPHs were reduced to 4124.5 km^2 of forest. The amount of forest remaining inside HPHs varied between 5% (Kerinci) and 87.3% (Sawah Lunto/Sijunjung). The HPHs in most districts still contained nearly 50% forest in 2001. The districts with the largest areas designated for HPHs were Bengkulu Utara (3034.1 km^2) and Merangin (2788.5 km^2) and the amount of forest in these districts in 1995 were 1307.7 and 1313.6 km^2, respectively.

Table 5.3: Change in forest distribution and cover for each district in HPHs from 1995 to 2001

	Forest cover				
District	HPH (km^2)	Forest in 1995 (km^2)	% HPH with forest in 1995	Forest in 2001 (km^2)	Deforestation 1995-2001 (%/yr)
Bengkulu Utara	3034.1	1307.7	43.1	1135.4	2.20
Bungo	1365.0	930.8	68.2	831.4	1.78
Kerinci	235.8	22.3	9.5	12.3	7.45
Merangin	2788.5	1313.6	47.1	1166.2	1.87
Musi Rawas	489.4	222.4	45.4	197.3	1.88
Pesisir Selatan	656.6	207.1	31.5	171.5	2.87
Rejang Lebong	0.5	0.5	100.0	0.5	0.00
Sarolangun	139.6	133.2	95.4	121.9	1.42
SawahLunto/Sijunjung	184.1	42.4	23.0	36.1	2.47
Solok	828.0	625.9	75.6	451.9	4.63
Total	9721.1	4805.9	Mean[a] = 53.9	4124.5	Mean[a] = 2.95

[a] Rejang Lebong excluded from mean calculation due to its small area

5.3.5 Forest and forest change in estate crop plantations

The estate crop plantations contained 498.5 km^2 of forest in 1995 (Table 5.4). Inside estate crop plantations, a total of 204.2 km^2 of forest was cleared between 1995 and 2001, equivalent to a mean deforestation rate of 5.91%/yr. In 2001, estate crop plantations were reduced to 294.3 km^2 of forest. Out of the five districts that had land assigned for these plantations Bengkulu Utara had the largest areas of plantation (614.4 km^2) that were being cleared of forest at high rates (4.02%/yr). Sawah Lunto/Sijunjung and Solok also had large areas under production that were being cleared at faster rates (5.92 and 8.83%/yr, respectively).

Table 5.4: Change in forest distribution and cover for each district in estate crop plantations from 1995 to 2001

	Forest cover				
District	Plantation (km^2)	Forest in 1995 (km^2)	% plantations with forest in 1995	Forest in 2001 (km^2)	Deforestation 1995-2001 (%/yr)
Bengkulu Utara	612.4	142.3	23.2	108.0	4.02
Bungo	54.1	14.0	25.9	13.9	0.05
Kerinci	-	-	-	-	-
Merangin	-	-	-	-	-
Musi Rawas	-	-	-	-	-
Pesisir Selatan	141.7	0.8	0.6	0.0	16.67
Rejang Lebong	-	-	-	-	-
Sarolangun	-	-	-	-	-
Sawah Lunto/Sijunjung	340.3	67.9	20.0	43.8	5.92
Solok	464.8	273.6	58.9	128.5	8.83
Total	1613.2	498.5	Mean[a] = 12.8	294.3	Mean[a] = 8.87

[a] Pesisir Selatan excluded from mean calculation due to its small area

5.3.6 Forest type and forest type change in the KS region

The detailed forest type map showed that there were still large blocks of lowland forest in the western section of the KS region, particularly in the southeast area covered by Bengkulu Utara (Figure 5.2). In the east and northeast areas, hill forest tended to occur outside of KSNP, whereas in the western areas hill forest was inside. Submontane and montane was mainly situated inside KSNP.

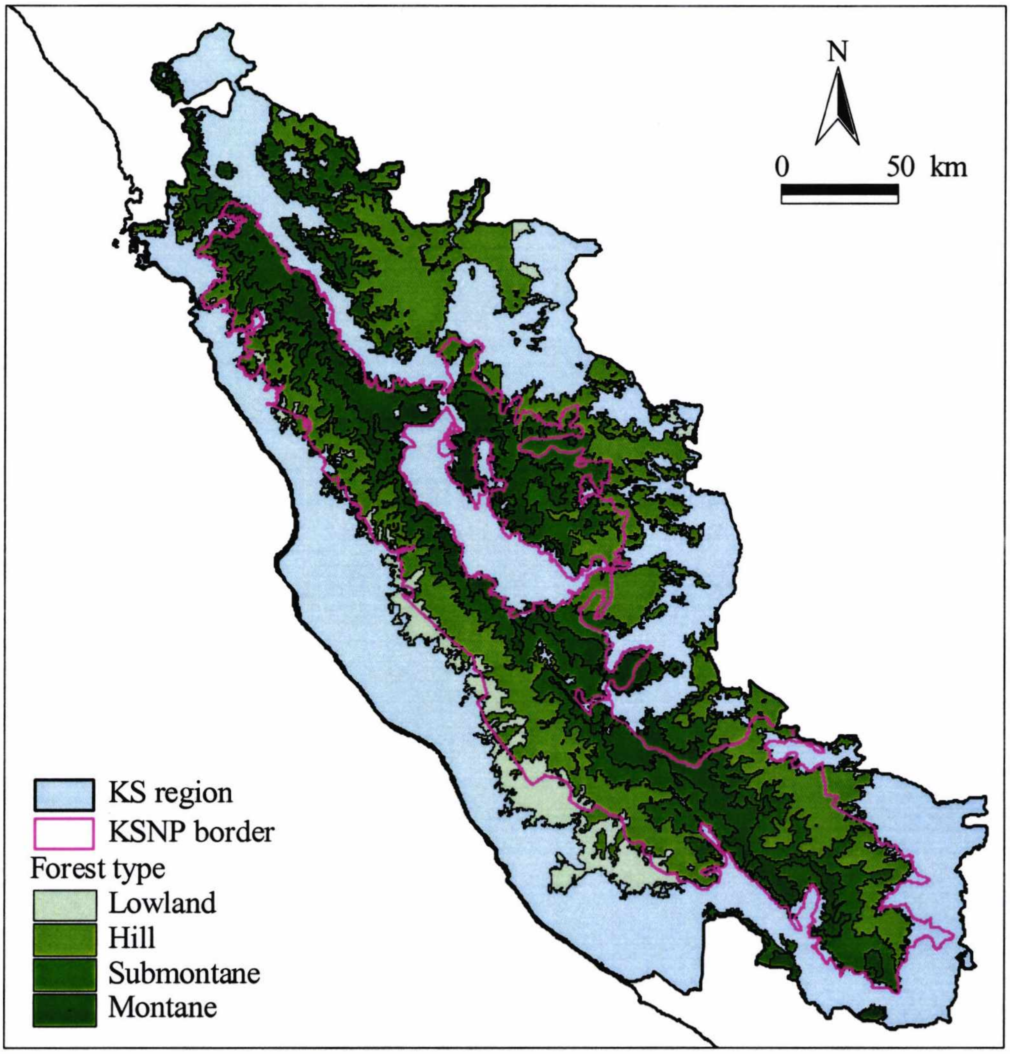


Figure 5.2: Broad forest types in 2001 for the KS region

Hill forest was the most abundant forest type in the KS region, representing 9693.1 km^2 in 1995, or over 40% of all forest (Table 5.5). Submontane was the next most abundant representing 6614.6 km^2, or about 30% (Table 5.6). Hill forest also suffered the largest amounts of deforestation between 1995 and 2001 with 588 km^2 of forest being cleared, equivalent to a mean deforestation rate of 1.01%/yr. Lowland forest had the next largest amounts of forest loss with 368.3 km^2 being cleared, but lowland forest underwent the most rapid transformation with a mean deforestation rate of 2.61%/yr.

Table 5.5: Change in forest distribution and cover for each forest type in the KS region from 1995 to 2001

Forest type	Forest in 1995 (km^2)	% total forest type in 1995	Forest in 2001 (km^2)	Deforestation 1995-2001 (%/yr)
Lowland	2355.3	10.6	1987.0	2.60
Hill	9693.1	43.5	9105.1	1.02
Submontane	6614.6	29.7	6375.7	0.60
Montane	3644.3	16.3	3562.2	0.38
Total forest cover	22307.3	100.0	21030.1	

5.3.7 *Lowland forest and forest change in the districts*

Bengkulu Utara contained the most amount of lowland forest in 1995 with 1568.9 km^2 (Table 5.6). This represented nearly 70% of all lowland forest in the KS region. The loss of lowland forests in this district was 1.96%/yr, much higher than overall forest loss in the region.

Table 5.6: Change in lowland forest distribution and cover for each district in the KS region from 1995 to 2001

		Lowland forest		
District	Forest in 1995 (km^2)	Forest in 2001 (km^2)	Proportion of forest (%)	Deforestation 1995-2001 (%)
Bengkulu Utara	1568.9	1384.9	69.7	11.7
Bungo	73.3	61.3	3.1	16.4
Kerinci	0.0	-	--	-
Merangin	17.3	7.6	0.4	56.3
Musi Rawas	61.3	50.0	2.5	18.4
Pesisir Selatan	465.1	397.5	20.0	14.5
Rejang Lebong	0.0	-	-	-
Sarolangun	0.0	-	-	-
Sawah Lunto/Sijunjung	123.9	64.5	3.3	47.9
Solok	45.6	21.3	1.1	53.2
Total	2355.3	1987.0	100.0	Mean = 31.2

The distribution of lowland forest within Bengkulu Utara was predominantly within HPHs (68.4%), then KSNP (24.3%), then estate crop plantations (7.3%). The clearance of lowland forest was greatest within plantations (4.06%/yr), then HPHs (2.36%/yr), and finally KSNP (0.30%/yr).

5.3.8 Hill forest and forest change in the districts

The majority of hill forest (80%) was divided between the five districts: Bengkulu Utara (16.3%), Merangin (13.8%), Musi Rawas (15.0%), Pesisir Selatan (13.7%), and Solok (21.2%) (Table 5.7). Solok had a much highest deforestation rate in comparison (1.84%/yr).

Table 5.7: Change in hill forest distribution and cover for each district in the KS region from 1995 to 2001

		Hill		
District	Forest in 1995 (km^2)	Forest in 2001 (km^2)	Proportion of forest (%)	Deforestation 1995-2001 (%)
Bengkulu Utara	1494.2	1484.9	16.3	0.6
Bungo	805.9	690.9	7.6	14.3
Kerinci	300.9	290.4	3.2	3.5
Merangin	1333.5	1251.7	13.8	6.1
Musi Rawas	1413.6	1366.0	15.0	3.4
Pesisir Selatan	1291.8	1246.2	13.7	3.5
Rejang Lebong	84.0	73.6	0.8	12.4
Sarolangun	427.4	415.2	4.6	2.9
Sawah Lunto/Sijunjung	376.0	359.8	4.0	4.3
Solok	2165.9	1926.3	21.2	11.1
Total	1494.2	9105.1	100.0	6.1

Within these five districts hill forest was predominantly inside KSNP (69.4%), then HPHs (28.2%), and finally plantations (2.3%). The largest amounts of hill forest inside KSNP were located in Bengkulu Utara (1334 km^2), Musi Rawas (1171.7 km^2), and Pesisir Selatan (1034.2 km^2). From these Bengkulu Utara had the lowest rate of hill forest loss (0.01%/yr), then Pesisir Selatan (0.26%/yr), then Musi Rawas (0.39%/yr). Within HPHs the only significant amounts of hill forest were located in Merangin (928.9 km^2) and Solok (554.9 km^2).

5.4 DISCUSSION

The correlates of deforestation may differ from region to region (Bawa and Dayanandan 1997). Between 1990 and 1997, high and ranging deforestation rates were experienced in Côte d'Ivoire (1.1–2.9%), Madagascar 1.4–4.7%/yr), Brazilian Amazonian belt (0.9%/yr-4.4%/yr), Colombia-Ecuador border (~1.5%/yr), south-eastern Kalimantan (1.0–2.7%/yr) and southern Vietnam (1.2–3.2%/yr) (Achard et al. 2002). The annual deforestation rate of 0.96%/yr across the KS region was lower than that recorded from the Leuser Management Unit (LMU), a similar sized protected area in North Sumatra. Between 1985 and 2000 the LMU recorded an average forest loss of 340 km^2/yr or 1.50%/yr (LMU unpublished data). Both of these deforestation rates are much lower than the 3.2-5.9%/yr recorded from unprotected areas on central Sumatra (Achard et al. 2002).

Deforestation inside KSNP (0.28%/yr) was even lower, especially when compared to Bukit Barisan Selatan National Park (BBSNP) in South Sumatra. Since 1985, BBSNP in southern Sumatra has lost 28% of its original forest cover and between 1985 and 1999, satellite imagery recorded a deforestation rate of 2%/yr inside BBSNP (Kinnaird et al. 2003). This was equivalent to an average lowland deforestation rate of 1.93%/yr. Large areas of forest were cleared for coffee production (O'Brien and Kinnaird 2003). Although conversion of forest by coffee farmers is an important threat in the KS region it is only occurring in small patches inside the national park border. Forest conversion by subsistence farmers inside HPHs was much higher at 2.96%/yr. Solok and Bengkulu Utara lost the most amounts of forest inside HPHs.

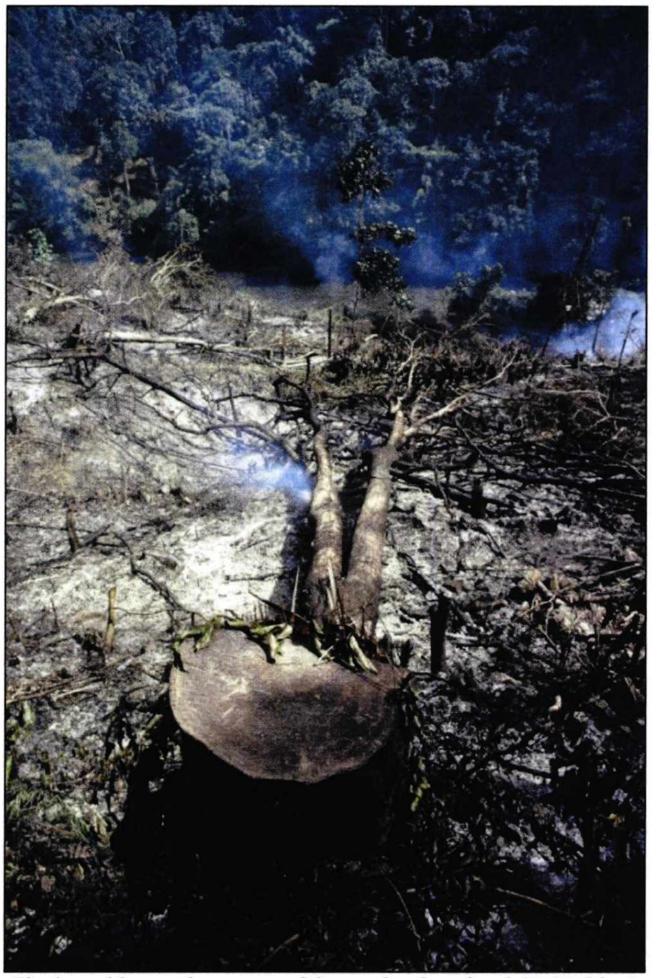
Bengkulu Utara, located in the southerly section of Sumatra, is experiencing a large net inflow of transmigrants. The paucity of available space for creating new farmland in Southern Sumatra has resulted in transmigrants moving towards more central districts, such as Bengkulu Utara, to find land. This situation has resulted in massive deforestation in Southern Sumatra. In Solok and Bengkulu Utara the designation of HPHs inside villages may create further land insecurity and lead to communities clearing more forest to secure land for their livelihood. In central Sumatran provinces increased production of oil palm plantations has resulted in large areas of forest being replaced by these plantations. The area represented by estate crop plantations in the

KS region was relatively small because large parts of the land outside KSNP was favoured for HPHs.

The KS region still contains a reasonable amount of lowland forest (1987 km^2). Lowland forest has been identified as the most threatened forest type in Indonesia (Holmes 2001). In the KS region, lowland forest experienced the largest relative losses with 15.6% having disappeared since 1995. Most of this clearance occurred in Bengkulu Utara, which contained nearly 70% of the region's lowland forest. Again this loss was attributed to subsistence farming activities. Hill forest is the next most threatened forest type in the KS region. Since 1995, 6.1% of the original cover had disappeared. This forest, although predominantly split between five districts, was subjected to disproportionate levels of clearance. Of these Bengkulu Utara had the lowest rates, probably because the more accessible and favoured lowland forest was acting as a buffer to the hill forest. Solok had lost 11.06% of its original hill forest cover since 1995, which represented the greatest amount between these districts.

Indonesia has sought to manage its forests prudently through a major governance reform (the National Forest Program). This offered the opportunity to "move toward more local participation in resource allocation decisions, greater accountability by regional governments, a refocusing of central agencies on policy and oversight" (Holmes 2001). In reality and in general, decentralization of the natural resource sector encouraged irresponsible resource management, resulting in rampant illegal logging and overexploitation (FWI/GFW 2002, Jepson et al. 2002). From the KS region the location of forest and different forest types has been identified and their rates of deforestation calculated. The next chapter therefore investigates what factors are causing this deforestation and determines which areas are most at risk of forest loss in the future.

Chapter 6
MAPPING AND PREDICTING DEFORESTATION



Slash and burn clearance of forest for farmland (J. Holden)

6.1 INTRODUCTION

Tropical rainforests are some of the most species-rich habitats on earth. They provide important biological, social and economic services. Even though the global incentives for maintaining intact tropical ecosystems far exceeds the economic value of converting these to alternative land uses, such as farmland, their destruction and degradation continues (Balmford et al. 2002). This threat in the form of habitat loss and fragmentation is one of the most severe facing species across their ranges because it is usually irreversible (Hitlon-Taylor 2000, Mace and Balmford 2000).

There is therefore the need to reduce tropical deforestation, but the factors causing this deforestation are often complex. It is important to gain accurate information on the causes of forest loss. An important source of this information comes from remotely sensed data, which has led to more accurate estimates of deforestation rates and location (Green and Sussman 1990, Sánchez-Azofeifa et al. 1999, Trejo and Dirzo 2000) and to the identification of the key factors involved (Dirzo and Garcia 1992, Vina and Cavelier 1999). Previous studies have tended to either focus on assessing the physical factors that explain deforestation, such as proximity to roads or elevation (Sader and Joyce 1988, Dirzo and Garcia 1992) or the social factors, such as poverty and capital markets (Barbier 1997). In order to better understand the deforestation process studies of tropical deforestation must investigate the interactions of both physical and socio-economic factors, but such studies are generally lacking (Sunderlin and Resosudarmo 1996, Lambin 1997, Laurance et al. 2002).

Reducing rates of deforestation will involve action at a range of political levels (Whitten et al. 2001). On Sumatra, forest is being cleared by illegal loggers and by commercial and subsistence agriculturalists, leading to recent estimates that all of the island's lowland forest will be cleared within several years (Holmes 2001). In order to prevent the threats posed by illegal logging and encroachment it is vital to patrol the existing protected areas (PAs) and enforce its boundaries (Bruner et al. 2001; Sánchez-Azofeifa et al. 2003). Large PAs often have limited financial resources and need to focus their efforts for greatest effect (Leader-Williams and Albon 1988). So there is a priority to identify vulnerable sites that require urgent protection (Pressey and Taffs 2001). This can be achieved by determining the correlates of deforestation and using these to predict future deforestation patterns (Linkie et al. 2004). This approach is especially relevant to KSNP, because it occurs on the Indonesian island of Sumatra that has some of the highest deforestation rates in the tropics (Laurance 1999; Holmes 2001).

A further intervention to mitigate the deforestation caused by agricultural expansion in the KS region might involve community outreach programs or community development. A strategy to adopt this for KSNP was an Integrated Conservation and Development Project (ICDP). Across Asia, many ICDPs have attempted to integrate biodiversity conservation with socio-economic development of villages living around tiger reserves (MacKinnon et al. 1999, MacKinnon 2001). One of the aims of the KS-ICDP was to reduce the amount of habitat loss caused by deforestation and fragmentation.

ICDPs are have been criticized because of the indirect and ambiguous conservation incentives that they offer (Ferraro 2001), which can impede efforts to reduce forest loss (Sayer et al. 2000, du Toit et al. 2004). Of particular concern in the KS region is the excessive deforestation in frontier areas because edge effects may have detrimental effects on tigers and their prey (Chapters 7 and 8). This forest clearance is a widespread problem across the tropics and it tends to be more severe when frontier communities lack property rights (Dorner and Thiesenhusen 1992). Under these conditions forests can be converted to agriculture; converted to plantations, logged for their timber, or other large scale projects; or alternatively remain as forest. A local community may claim ownership on an area of forest based on customary principles, but this is no guarantee for sustainable forest management. Customary forests are often subjected to whimsical confiscation by the government in Indonesia (World Bank 2000). If villages have insecure and customary property rights, and central government assigns forestland within their administrative boundaries to large scale production projects, conserving these forest sustainably through customary approaches will be difficult if not impossible.

Secure tenure for a logging company is not synonymous for sustainable forest management. Managing tropical forests in this way is rarely as financially profitable as rapid and uncontrolled logging (Kaimowitz 2003). When a HPH boundary falls within a village it creates competition with the local community. From Sumatra and Brazil, the designation of forested areas within villages to large scale projects led to excessive deforestation as local communities here entered a race to clear forestland first in order to stake their claim to it (Alston et al. 1995, Angelsen 1999). Alternatively, customary land designated as a HPH may subsequently be sold to migrants, who then clear it for agriculture (Suyanto et al. 2000). Logging operations create inroads that further promote deforestation, because the cost of local agricultural expansion is lowered (Angelsen 1999, Linkie et al. 2004). Previously under the Suharto government, many communities in Indonesia were denied access to customary forest. This worsened their poverty. Now these communities believe that they have a valid claim to compensation or restoration of land use rights against the Government or HPHs or plantation companies (Holmes 2001). This has created additional pressures on the forests and brought communities into greater land conflict with these agencies (Poffenberger and McGean 1993).

In the KS region, HPHs occupied the greatest area out of the large scale projects and potentially pose a large threat (Chapter 4). Between 1996 and 2002, KSNP was the focus of a US$46M ICDP (World Bank 2003). This project aimed to reduce the pressure on the forest resources of KSNP. This provides the opportunity to assess how the ICDP and how insecure land tenure rights in combination with physical and socio-economic factors influence deforestation in the KS region.

6.1.1 Aims and objectives

This chapter aims to:

- assess the influence of various physical predictors of deforestation across the whole KS region;
- develop a predictive deforestation map based on these factors;
- predict future forest loss and forest fragmentation patterns;
- test the accuracy of forest loss predictions using forest change data from 2001 to 2002;
- evaluate the relationship between KS-ICDP villages and deforestation; and,
- assess the influence of various physical and socio-economic factors on deforestation within villages at the forest edge deforestation;

6.2 METHODS

6.2.1 Statistical methods

All datasets in this study were first tested to determine their probability distribution type so that they could be analysed using the appropriate GLM. From this two different analyses were carried out as part of this work and are described below. In the first analysis, a multiple logistic regression model was used to determine the predictors of deforestation across the entire KS region. The significant factors were used to construct a predictive forest loss map for the KS region. Only the physical factors were used in this analysis because they had complete coverage across the KS region. The socio-economic factors were excluded from this analysis because they did not extend inside forest polygons and could therefore not be used to construct a deforestation risk map. In the second analysis, a Poisson-lognormal model was used to determine the correlates of deforestation within villages at the forest edge. For this the socio-economic factors had to be adjusted to compensate for village boundaries that continued inside areas that still remained forested and had no human inhabitants. All factors used these analyses were obtained from the datasets developed in Chapter 4.

6.2.2 Deforestation patterns analysis

This analysis assessed the combination of physical factors on patterns of deforestation across the entire KS region. The 1995 and 2001 forest coverages were combined and used to produce two sets of polygons. All the polygons showed areas that were cloud-free in 1995 but some showed patches of land that were cleared of forest between 1995 and 2001, while others showed patches of forest that were not cleared. The "Animal Movement" extension in ArcView was then used to identify 150 randomly chosen points that were more than 2 km apart from the existing points in the cleared, and 150 points in the uncleared, polygons (Figure 6.1). The "Summarize Zones" module of ArcView was used to determine the elevation, slope, distance to logging roads, distance to roads, distance to settlements, distance to rivers, soil type and PA status of each point location.

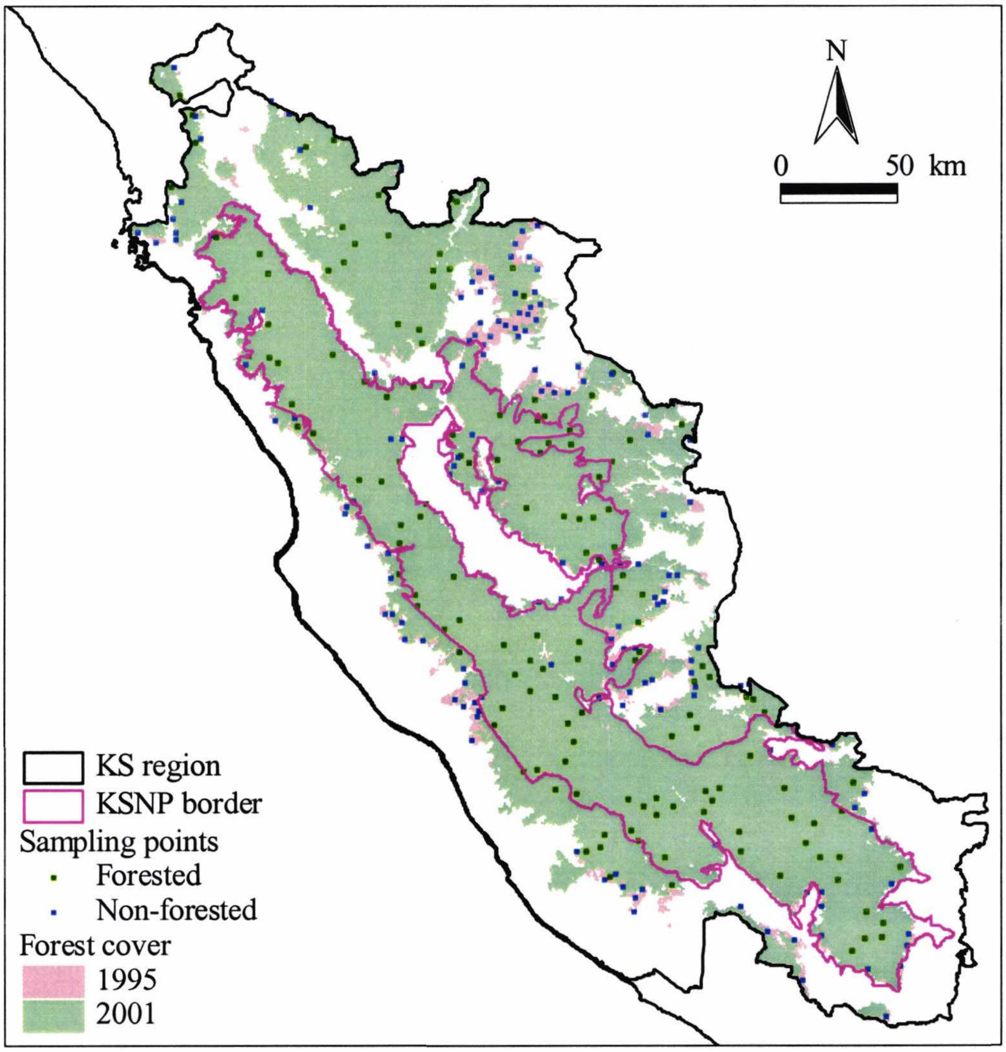


Figure 6.1: Sampling points used in the multiple logistic regression analysis

These data were imported into SPSS v.11 statistical software package (SPSS Inc., Chicago, IL). The continuous data were logarithmically transformed to improve their normality. A multiple logistic regression model was used to determine which combination of factors most accurately predicted forest loss in the KS region. The addition and removal of independent variables from the regression model was controlled by the Wald statistic with respective *P*-values of 0.05 and 0.1. The performance of the model was evaluated by calculating the area under the curve (AUC) of the receiver operating characteristics plot (as in section 3.2.4). The presence of spatial auto-correlation in the model was tested by calculating the Moran statistic of the regression unstandardized residuals using the Crime-Stat software (Levine 2000).

6.2.3 Calculating and predicting forest loss and fragmentation

From the final logistic regression model the probability of forest clearance (P) was determined by,

$$Y = \beta_0 + \sum \beta_i X_i$$

where β_0 is the constant coefficient, β_i represents the significant independent variable coefficients, and X_1 represents their associated independent variables. Through incorporating the natural exponential (e) into the previous equation the risk of deforestation map for the KS region was constructed by,

$$P = e^Y / 1 + e^Y$$

The model coverage was constructed using the "***Image Calculator***" in Idrisi and the ***OVERLAY*** module was used to multiply this by the 2001 forest coverage, to assign each 100 m^2 pixel of forest a probability of clearance and produce the final risk of deforestation coverage.

This coverage was then used to model future deforestation patterns by producing ten new coverages that showed the predicted forest cover at ten different stages. It was assumed that the rate at which a pixel of forest will be cleared would be proportional to its modelled risk of deforestation. Therefore, the first deforestation stage coverage was produced by reclassifying the risk of deforestation coverage to only contain pixels with a risk value of 0.9 or less. The second deforestation coverage contained pixels that had values of 0.8 or less and this process was repeated so that the tenth coverage contained pixels that had a deforestation risk of 0.1 or less (Table 6.1)

Table 6.1: Predicted deforestation risk stages

Deforestation stage	Predicted probability of clearance (P)
1	> 0.9-1.0
2	> 0.8-0.9
3	> 0.7-0.8
4	> 0.6-0.7
5	> 0.5-0.6

6	> 0.4-0.5
7	> 0.3-0.4
8	> 0.2-0.3
9	> 0.1-0.2
10	0-0.1

These ten coverages were then used to calculate the predicted trends in forest area loss and changes in patch size and number. The amount of forest remaining at each stage of deforestation, and the amount of forest lost in between stages, were then calculated. For calculating patch size and number, only forest blocks greater than 1 km^2 were included, because a large number of very small forest blocks would distort the mean fragment size at each deforestation, resulting in forest fragmentation appearing as more dramatic than it actually was. The forest type map constructed in Chapter 3 was used to calculate the predicted patterns of forest type loss. At each stage the mean forest patch size, excluding those less than 1 km^2, and number of patches were calculated to give an indication of forest fragmentation.

The predictions of this model were tested by randomly selecting 100 points in areas that still contained forest in 2002 and 100 points in areas that had been cleared of forest between 2001 and 2002. A Mann-Whitney U test was then used to find whether those sites that had been cleared by 2002 had a higher predicted risk of clearance from the 1995 to 2001 model than the sites that had not been cleared.

6.2.4 Forest status and the ICDP

The KS-ICDP has focussed on 74 villages since 1997. In this analysis, nine of these villages were excluded because eight contained no forest within their administrative boundaries and one contained only 0.3 km^2 of forest. Village boundary data were obtained in a digital format from the Indonesian Central Bureau of Statistics. The area of forest in 1995 and 2001 was extracted for each ICDP village so that the proportion of deforestation could be calculated. This was repeated for a random subset of non-ICDP villages.

From the 1085 villages located within the KS region a random subset of 65 non-ICDP villages was selected following the main criteria used for the ICDP villages, which contained forest, and intersected or adjoined KSNP, or logging concessions that adjoined KSNP. The proportion of forest loss within non-ICDP villages was calculated using the same methodology for that of forest loss in ICDP villages. However, proportion of forest loss was found to be related to village area (univariate GLM, $n = 130$, $F = 5.87$, $P = 0.016$). So in order to test whether ICDP status was the only factor responsible an additional criterion of village size was entered into the selection algorithm. The non-ICDP subset was selected using the "Animal Movement" extension in ArcView (Figure 6.2). A Mann-Whitney U test was used to determine if there was any significant difference in the proportion of deforestation between the ICDP focal villages and non-ICDP villages.

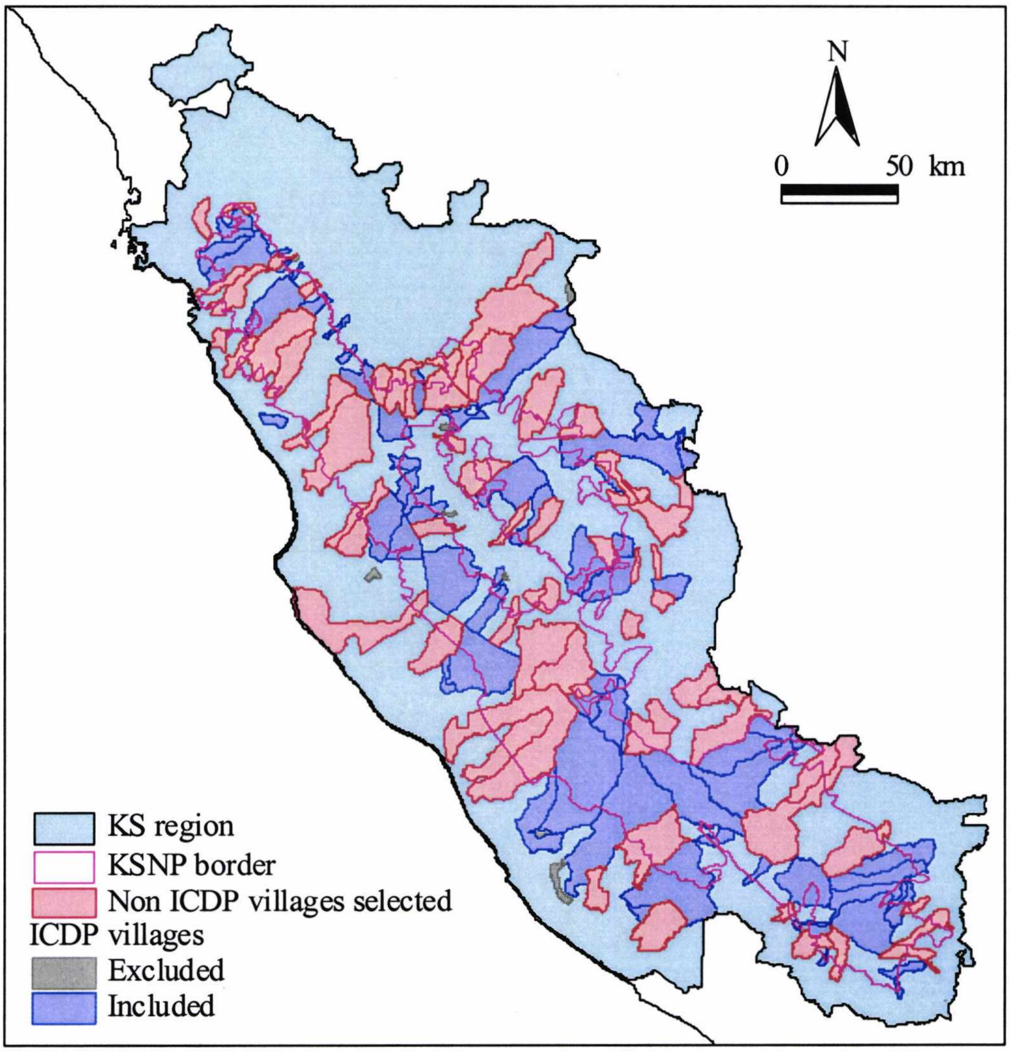


Figure 6.2: ICDP villages and subset of non-ICDP villages used for the deforestation analysis

6.2.5 Edge deforestation patterns analysis

This analysis assessed the combination of physical and socio-economic factors on patterns of deforestation within villages at the forest edge. From 645 villages, only those that contained more than 0.5 km^2 in 1995 within their boundaries were included in the analysis (n = 245). A preliminary analysis was performed to determine if deforestation was significantly different in villages that were partially or fully occupied by HPHs (n = 145) with villages that were not (n = 95). If there was a significant difference then the subsequent deforestation analysis would be refined to only those villages with HPH occupancy. From this the "Summarize Zones" module of ArcView was used to determine the mean values of elevation, slope, distance to logging roads, distance to roads, distance to settlements, distance to rivers, productive labour force, village development, village poverty, and proportion of a village occupied by a HPH, and the median value of soil type for each cell. The proportion of forest loss was then extracted for each cell.

These data were imported into MS Excel 2000, converted into a text (.txt) file, and then imported into the GLIM v.4 statistical software package (The Numerical Algorithms Group Inc., Downers Grove, IL). The HPH proportional data were transformed (raised by x^2) to improve their linear fit. The other continuous variables were either logarithmically or power transformed for the same reason. A Poisson-lognormal model was used to determine which combination of factors most accurately predicted the proportion of forest loss within villages. The presence of spatial auto-correlation in the model was tested by calculating Moran's *I* statistic (as in section 6.2.2).

6.3 RESULTS

6.3.1 Deforestation analysis

From 1995 to 2001, the physical factors that best explained the probability of an area being cleared of forest were related to $\log_{10}$ elevation, to $\log_{10}$ distance to settlements, to $\log_{10}$ distance to public roads, to $\log_{10}$ slope and to protected area status (Table 6.2, Figures 6.3-6.7). Forested areas that were at lower elevations, nearer to settlements and roads, on flatter terrain and outside of KSNP were more likely to be cut down.

The logistic regression model explained 77.5% of the original observations and was not affected by spatial autocorrelation (Moran's $I = 0.02$, $P > 0.1$). The final model had an AUC value of 0.828 indicating an accurate fit.

Table 6.2: Best multiple logistic regression model describing the relationship between landscape variables and deforestation across the KS region

Factor	Coefficient (β) ± S.E.	df	Wald	*P*
Log_{10} elevation	-1.47 ± 0.542	1	7.37	0.007
Log_{10} distance to settlements	-1.68 ± 0.665	1	6.43	0.011
Log_{10} distance to public roads	-1.41 ± 0.565	1	6.28	0.012
Log_{10} slope	-0.853 ± 0.340	1	6.30	0.012
Protected area status				
Outside PA	-0.949 ± 0.389	1	5.96	0.015
Inside PA	(included in constant)			
Constant	16.429 ± 3.423	1	23.06	<0.001

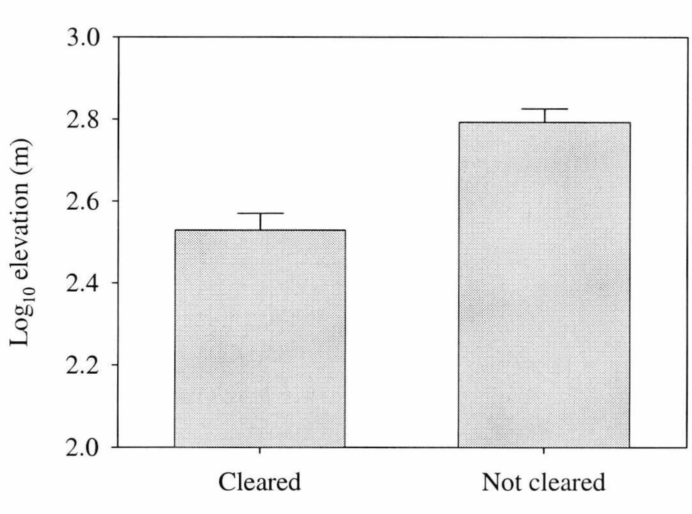


Figure 6.3: Likelihood of forest clearance related to mean log_{10} elevation (with S.E. bars)

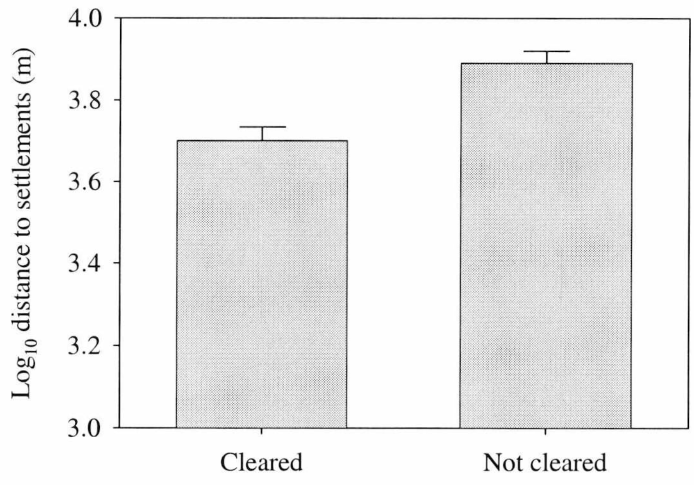


Figure 6.4: Likelihood of forest clearance related to mean $\log_{10}$ distance to settlements (with S.E. bars)

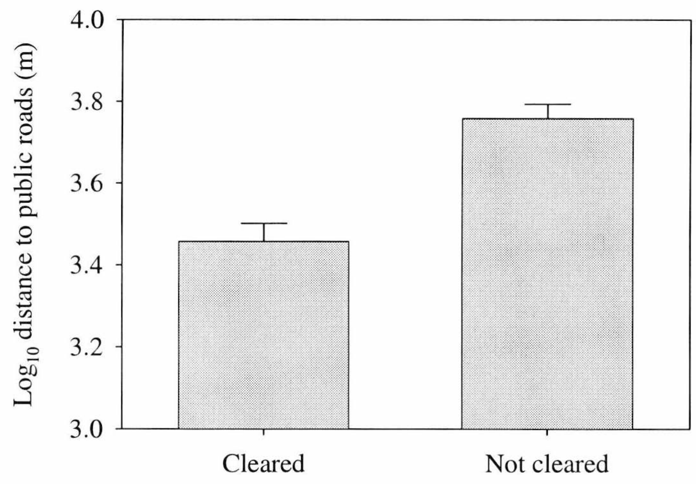


Figure 6.5: Likelihood of forest clearance related to mean $\log_{10}$ distance to public roads (with S.E. bars)

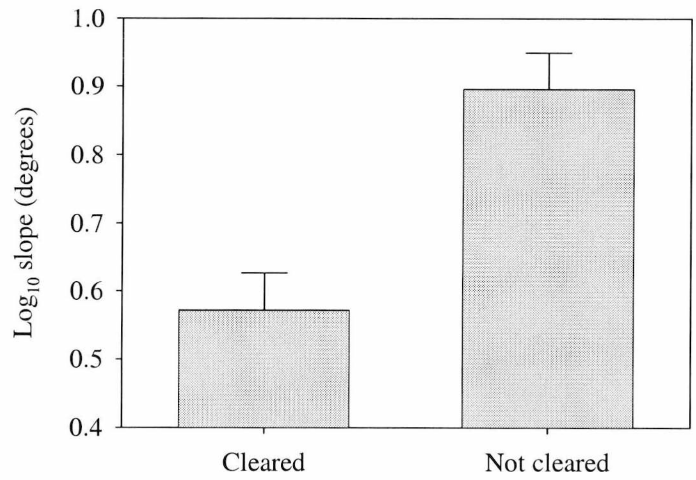


Figure 6.6: Likelihood of forest clearance related to mean $\log_{10}$ slope (with S.E. bars)

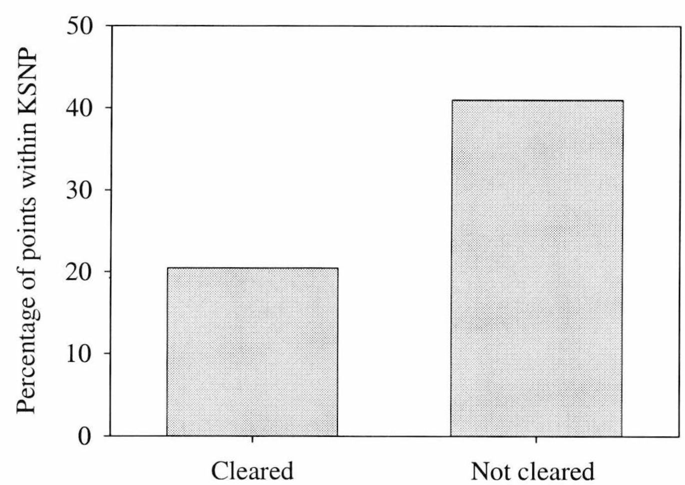


Figure 6.7: Percentage of points found within KSNP

6.3.2 Predicting future forest loss

The coverage produced for risk of deforestation predicted that the large patches occurring outside of KSNP in the northeast, east and southwest of the KS region were most susceptible (Figure 6.8). Forest within KSNP was generally less at risk. However, areas in the central section of KSNP were found to be highly susceptible to clearance because an asphalt public road divided them.

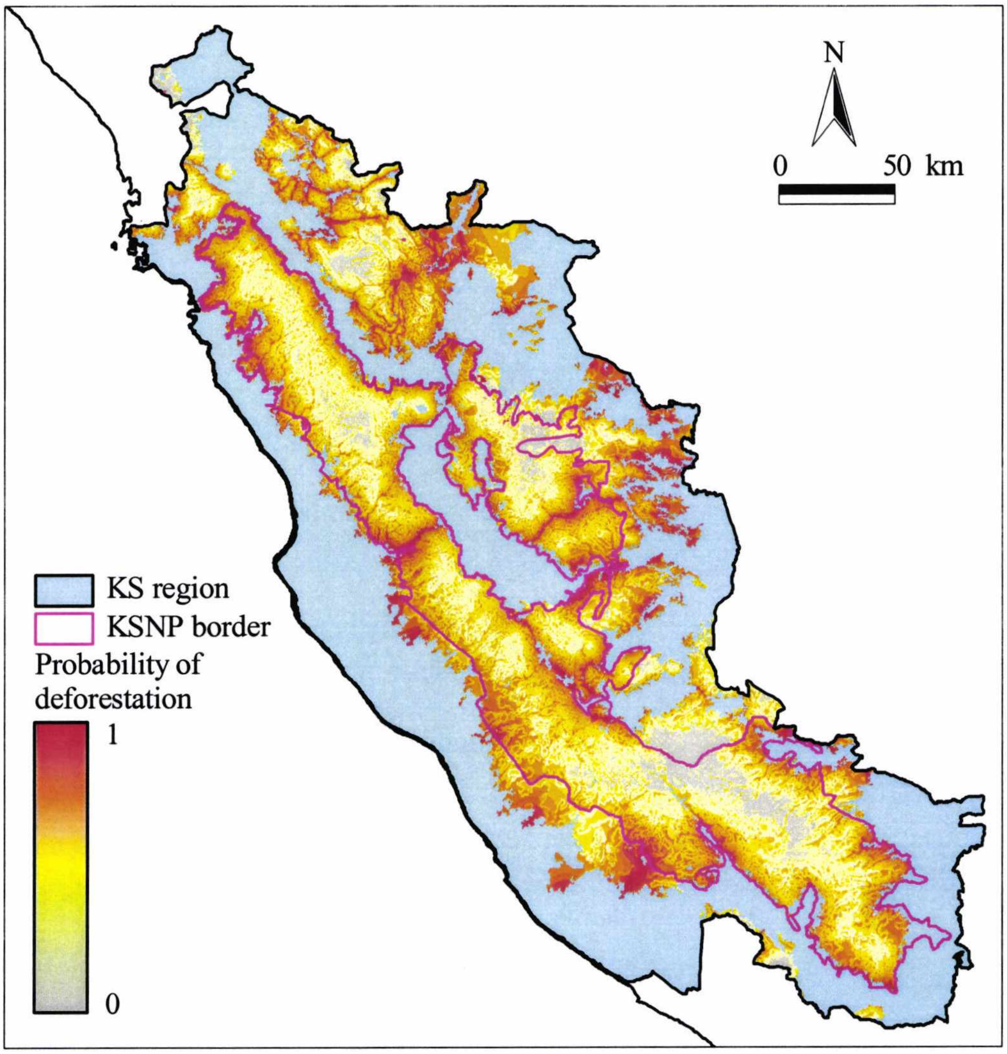


Figure 6.8: Predicted risk of deforestation in the KS region

Randomly selected sites that were cleared of forest between 2001 and 2002 had a mean predicted deforestation risk of 0.601 based on the 1995-2001 habitat threat model (Figure 6.9). In contrast, the randomly selected sites that were not cleared of forest between 2001 and 2002 had a mean predicted deforestation risk of 0.266, which was significantly lower than that of the cleared sites (n = 200, Mann-Whitney U = 1243.0, Z = -9.18, $P < 0.001$).

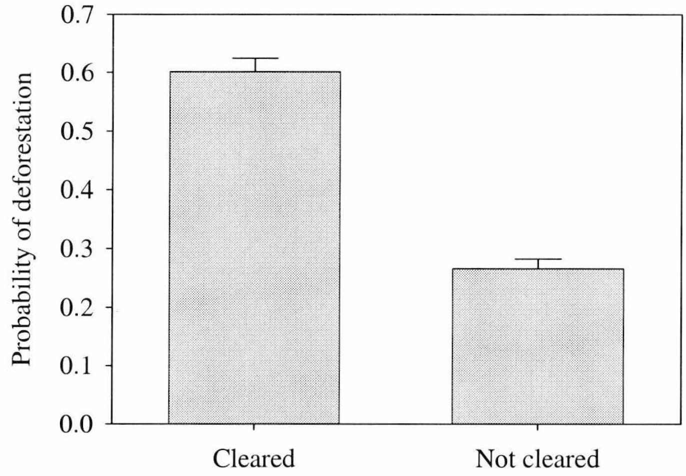


Figure 6.9: Mean predicted probability of 1995 to 2001 deforestation model for clearance that occurred in 2002 (with S.E. bars)

6.3.3 Predicted forest fragmentation patterns

Increasingly larger areas of forest are lost from stages 1 to 9 (Table 6.3, Figures 6.10 and 6.11). The predicted pattern of deforestation suggests that forest losses will steadily decline until deforestation stages 4 ($P > 0.6 - 0.7$) when the losses will increase more rapidly until stage 8 ($P > 0.2 - 0.3$), when only a small amount of forest will remain.

Table 6.3: Details of predicted effects of deforestation on total forest cover

Deforestation stage	Probability of clearance category (P)	Area of forest loss (km^2)	Forest area remaining (km^2)	Mean patch size (km^2)	Number of patches
Year 2001	-	-	21130.9	340.0	62
1	> 0.9 – 1.0	279.4	20851.5	205.4	101
2	> 0.8 - 0.9	608.0	20243.5	199.2	101
3	> 0.7 - 0.8	1005.2	19238.4	172.2	111
4	> 0.6 - 0.7	1381.7	17856.7	153.0	116
5	> 0.5 - 0.6	1797.3	16059.4	126.7	125
6	> 0.4 - 0.5	2415.3	13644.1	100.1	135
7	> 0.3 - 0.4	3206.2	10437.9	95.2	108

8	> 0.2 - 0.3	4104.5	6333.4	61.0	101
9	> 0.1 - 0.2	5013.9	1319.5	23.4	52
10	0.0 - 0.1	1319.5	0.0	0.0	0

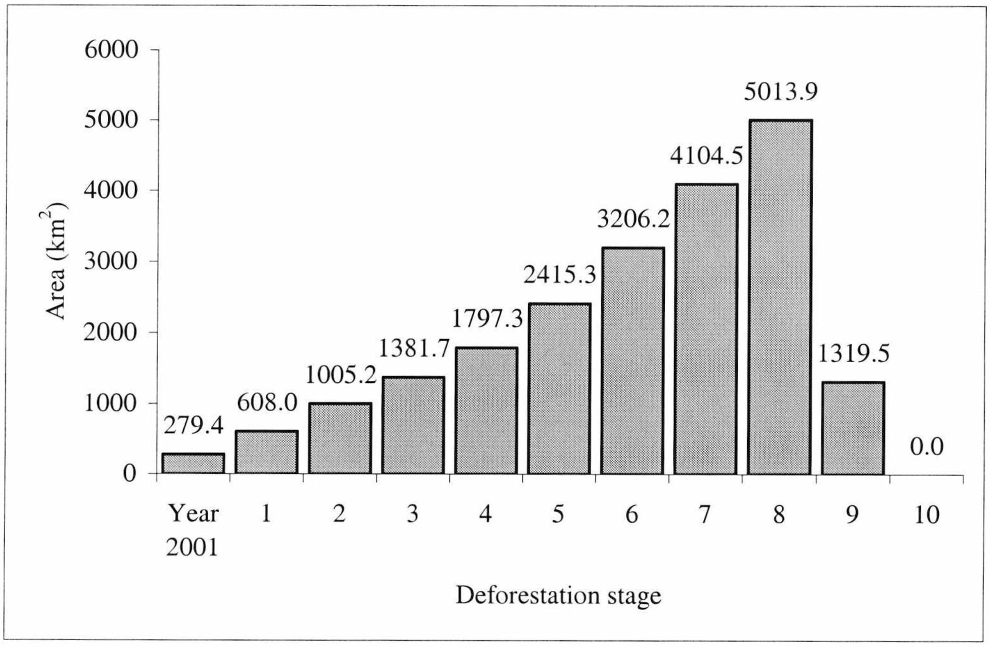


Figure 6.10: The areas of forest belonging to each deforestation stage in the KS region

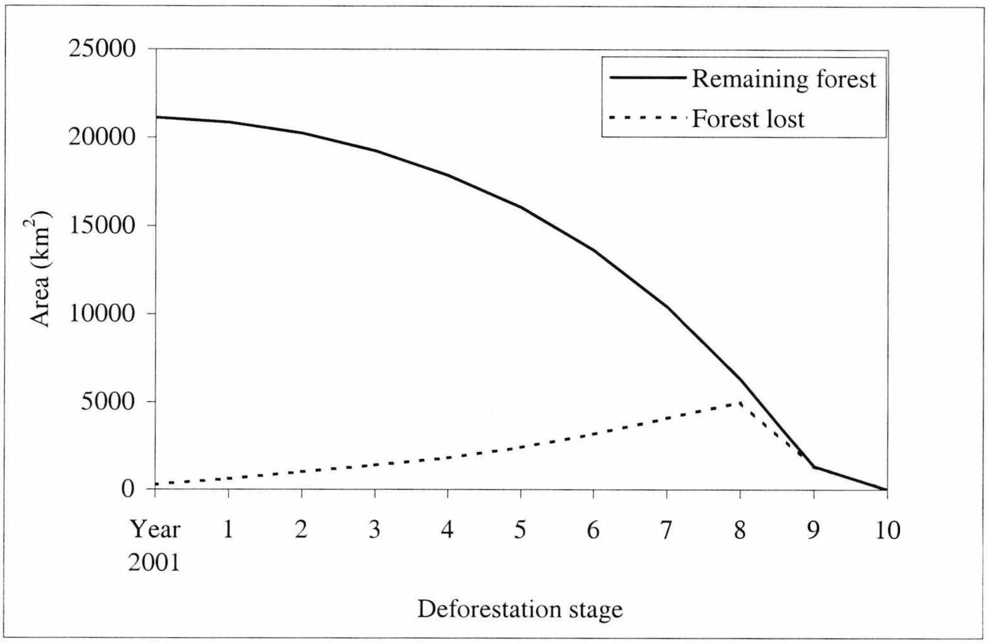


Figure 6.11: Predicted sequential and total forest loss in the KS region

Patch size decreases rapidly as large blocks of forest are fragmented early on (deforestation stage 1, $P > 0.9 - 1.0$). After this fragmentation of forest blocks slows markedly because forest patches are predicted to shrink rather than fully split (Table 6.3, Figure 6.12). The small amount of lowland forest declines steadily from the first stage of deforestation and is predicted to disappear much quicker than the other forest types. Hill forest is then predicted to be cleared (stage 4), followed by submontane forest (stage 6) and montane forest (stage 7), after which all types decline rapidly (Table 6.4, Figure 6.13).

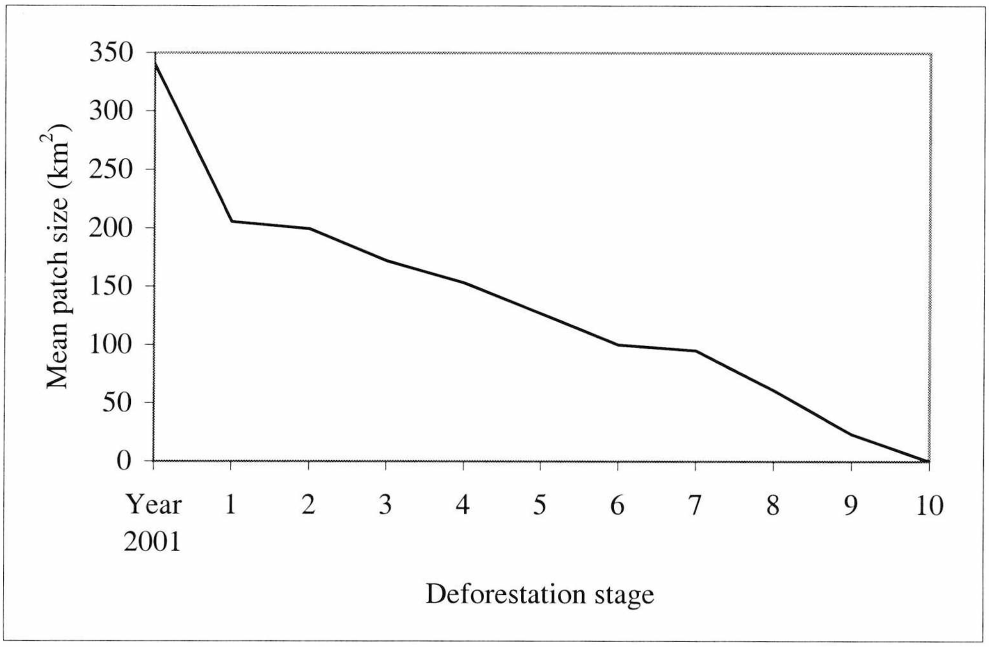


Figure 6.12: Predicted sequential change in forest patch size due to deforestation in the KS region

Table 6.4: Predicted deforestation patterns for the different forest types in the KS region

Deforestation stage	Forest type (km^2)			
	Lowland	Hill	Submontane	Montane
Year 2001	3213.3	7930.0	6417.8	3569.7
1	3078.9	7838.1	6373.3	3561.3
2	2801.1	7623.4	6270.3	3548.7
3	2379.2	7262.7	6090.7	3505.8
4	1890.6	6722.4	5821.2	3422.6
5	1391.7	5976.7	5415.3	3275.7

6	883.3	4958.5	4813.3	2989.0
7	439.3	3581.8	3906.0	2510.8
8	139.7	1875.6	2631.6	1686.5
9	2.7	191.3	692.4	433.1
10	0.0	0.0	0.0	0.0

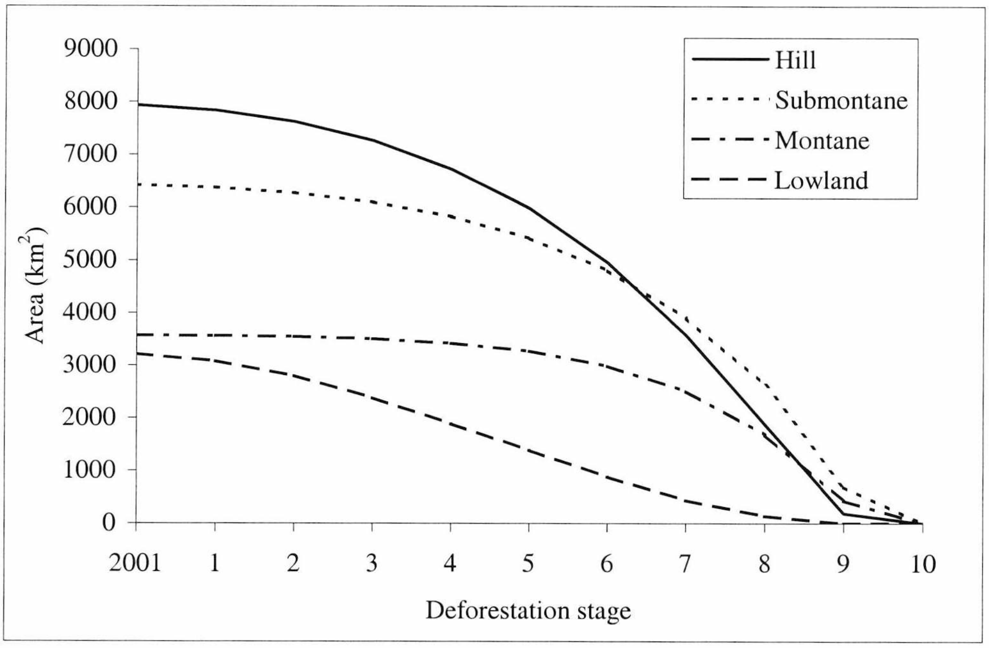


Figure 6.13: Predicted deforestation patterns for forest types in the KS region

6.3.4 Forest loss and ICDP status

The average size of ICDP villages was 125.3 km^2, compared with 122.1 km^2 for non-ICDP villages used in the analysis. Between 1995 and 2001 the average forest loss was 0.127 (S.E. = 0.033) within the ICDP villages, compared to 0.130 (S.E. = 0.029) within the non-ICDP villages. There was no significance difference in the proportion of forest loss between ICDP and non-ICDP villages (n = 130, Z = -0.947, U = 1971.5, P = 0.343; Figure 6.14).

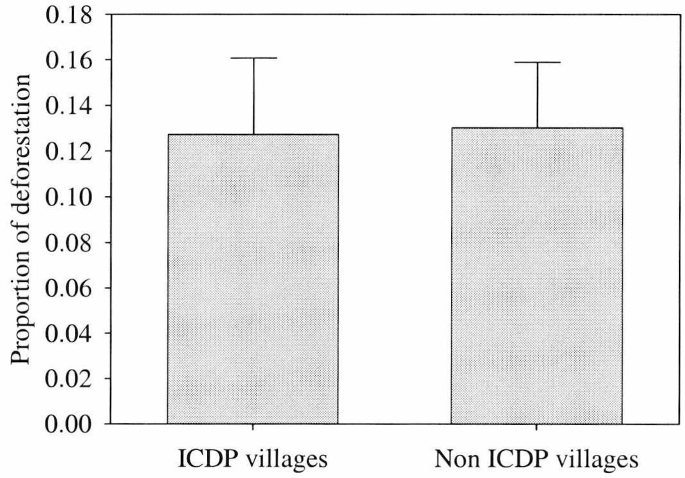


Figure 6.14: Mean forest loss in ICDP and non-ICDP villages (with S.E. bars)

6.3.5 Forest loss and HPH status

Villages that had a proportion of their forest assigned to a HPH had a much higher proportion of forest clearance than those with no HPHs located within their boundaries ($n = 240$, $Z = -5.165$, $U = 4190.0$, $P < 0.0001$; Figure 6.15).

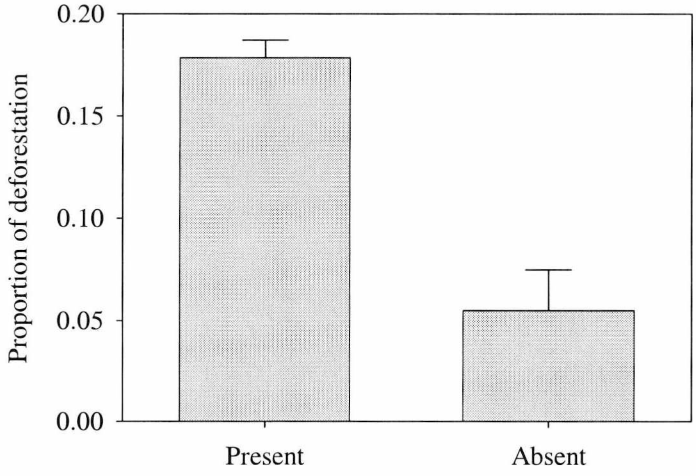


Figure 6.15: Mean forest loss in villages with a HPH present or absent within their boundaries (with S.E. bars)

6.3.6 *Edge deforestation analysis*

Between 1995 and 2001, the physical and socio-economic factors that significantly explained an area of forest being cleared was related to the transformed proportion of a village assigned to a HPH ($P < 0.05$) and to transformed slope ($P < 0.05$) (Table 6.5, Figures 6.16 and 6.17). Forested areas that were located in villages with a larger proportion of their area assigned to HPHs and on flatter terrain were more likely to be cut down. The Poisson-lognormal model explained 62.2% of the original observations and was not affected by spatial autocorrelation (Moran's $I = 0.02$, $P > 0.1$).

Table 6.5: Best generalized linear model describing the relationship between landscape variables and forest edge deforestation in the KS region

Factor	Scaled deviance	Estimate ± S.E.	df	*P*
Transformed HPH occupancy	38.641	0.964 ± 0.642	2	<0.05
Transformed slope		-0.012 ± 0.007	2	<0.05
Constant		-1.982 ± 0.579	2	<0.05

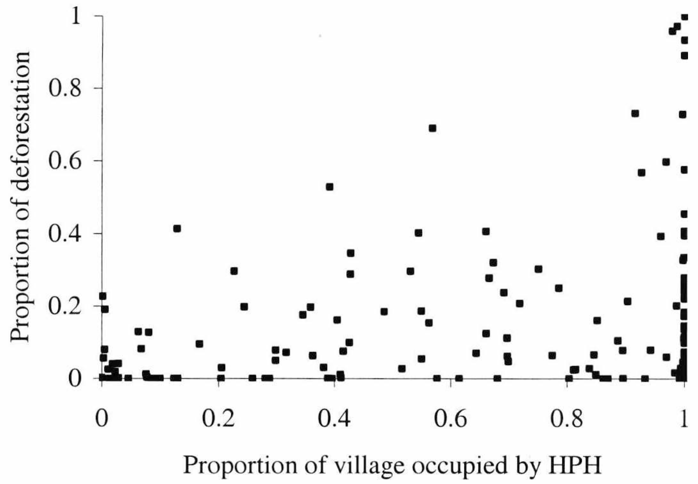


Figure 6.16: Proportion of deforestation compared to the amount of a village occupied by a HPH

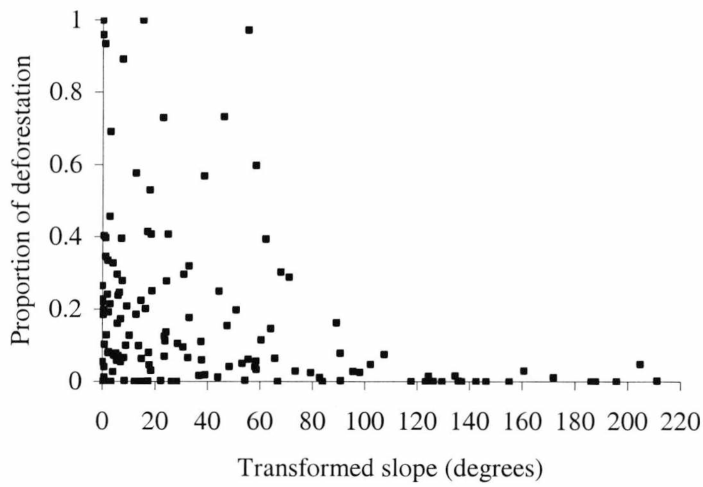


Figure 6.17: Proportion of deforestation compared to transformed slope

6.4 DISCUSSION

The results from this study highlight the critical role of accessibility, particularly from roads, in determining deforestation. This should serve as a caveat for major development projects including mines, roads and dams that threaten tiger reserves (World Bank 1996[b]). Across the KS region deforestation trends were complex, and were explained by five proximate factors: elevation, settlements, public roads, slope and PA status. The deforestation model was found to accurately predict forest loss, large patches of forest occurring outside of KSNP in the northeast, east and southwest were identified as being most at risk. The deforestation model can therefore be used to decide where future patrolling effort and community outreach programmes should be focussed. However, those villages participating in the ICDP did not have lower levels of deforestation than villages not participating. To reduce deforestation the ICDP would have to offer communities viable alternatives with greater incentives to not clear forest. A greater incentive might be securing land property rights. Villages without HPH occupancy had substantially lower forest conversion to agriculture than those occupied by a HPH, probably because there was less land conflict. The edge deforestation trends in the KS region are explained by two proximate factors: HPH occupancy and slope. Villages with a larger proportion of forest assigned to HPHs were in a more open access system. Under such a "tragedy-of-the commons" scenario farmers are encouraged to increase their amount of farmland to stake a claim before it is 'usurped'. This suggests that the current policy for natural resource use in Indonesia through logging concessions has a strong association with the deforestation caused by small scale farming activities. The current policy also has serious implications for tiger conservation on Sumatra.

The spatial distribution model for deforestation across the KS region included several factors that were all related to accessibility. It was unsurprising that the position of public roads was important in determining deforestation patterns (Sader and Joyce 1988, Mertens and Lambin 1997, Laurance et al. 2002). The protection status of forest was found to be important showing that the existence of KSNP even as a 'paper park' did indeed play a role in forest protection. The view that even poorly funded PAs can be partially effective has been supported by recent findings based on questionnaire data (Bruner et al. 2001). Caution is needed though when interpreting this result from KSNP as in other PAs (Liu 2001). Firstly, KSNP contains a large amount of inaccessible forest and its designation was probably partly based on its unsuitability for other land uses (Pressey 1994). Secondly, there are still patches of forest outside KSNP that can be logged without breaking the laws specifically associated with PAs. These factors in combination suggest that deforestation inside the PA is likely to be at a slower rate than elsewhere. However, logging will still probably take place within KSNP when no other sources of timber or space for farmland are available. If KSNP was effective in preventing the spread of illegal logging, then there would have been no deforestation within the PA and this was clearly not the case.

Even if protection status did correlate with lower levels of deforestation, this is not considered enough to prevent further encroachment because of the other factors that explain deforestation. Flatter areas of land, commonly found at lower elevations, are easier to cultivate and were found to be more susceptible to being converted to agriculture, as also recorded in Mexico (Trejo and Dirzo 2000). Forest at lower elevations is also more accessible (Dirzo and Garcia 1992) and has better quality timber (Jepson et al. 2001). From Thailand and Brazil, flatter land was associated with better quality soil and was more likely to be cleared (Cropper et al. 1997, Pfaff 1997).

In the KS region, deforestation levels tended to be highest around settlements, presumably because villagers preferred to travel short distances to collect timber and clear areas for farmland. However, most of these villages were at lower elevations and so the net effect of this was that low-lying forest was most susceptible to deforestation. This shows the importance of working with local communities to reduce illegal logging. Part of any solution will involve focussed forest protection and the deforestation model can be used to target these resources and predict future trends.

The risk of deforestation coverage indicated that the large patches of forest found in the northeast, east and southwest were most likely to be cleared, although most of the patches of forest that bordered KSNP were also at risk. The deforestation model predicted that forest inside the PA was less prone to logging but it is likely that this will increase as land space outside KSNP disappears. The areas inside KSNP that are most at risk lie in the central section because these areas can be easily accessed as they have public or logging roads inside the border of KSNP. The model predicts that these areas would benefit most from increased patrolling efforts, in conjunction with efforts to encourage local communities to protect the forest.

It is predicted that forest clearance will occur in areas of lowland and hill forest first because these forest types lie outside of KSNP and adjoin farmland. This makes them the most accessible and most threatened. Lowland forest is also vulnerable because it exists as small patches only. As these forest types disappear, submontane and montane forest will become exposed and threatened by the increased human pressures on the forest for natural resources and space for creating farmland. Submontane and montane forest are probably safer because they have a difficult topography and fewer valuable timber species in comparison to lowland forest, most of which was subsequently taken out of the recently gazetted KSNP. In Costa Rica and Mexico, montane forest also had the lowest levels of deforestation due to its poor accessibility and rugged terrain (Dirzo and Garcia 1992, Sanchez-Azofella et al. 2001).

On Sumatra, lowland has been identified as the most threatened forest type (Holmes 2001). These lowland forests, along with hill forests, are capable of supporting higher Sumatran tiger densities than submontane or montane forests (Griffith 1994, see Chapter 9). Nevertheless, these forests are more accessible to humans given their geography and proximity to logging and public roads. It is therefore likely that this forest will have greater human activity and human-related threats near their edges, thereby reducing the quality of this habitat for tiger and tiger prey (Kinnaird et al. 2003, O'Brien et al 2003). The ecological constraints on tigers living in evergreen rainforests means that they require larger areas than tigers in other habitats to maintain viable populations, because low primary productivity at the ground level supports a low density prey base (Eisenberg 1980). The threat posed by forest habitat loss and fragmentation will therefore have a greater impact on tigers in KSNP. This places greater emphasis on a strategy such as the KS-ICDP in mitigating deforestation and producing a village level model that could be feasibly applied across the KS region.

The KS-ICDP did propose participatory village development planning with NGO facilitators, and concession management for biodiversity conservation. However, the project was launched with unrealistic institutional arrangements, inadequate staffing, and no realistic plan to confront the major and immediate threats to the park (Wells et al. 1999). The KS-ICDP did not perform as expected, which suggests that the link between biodiversity conservation and social and economic development of the ICDP villages was weak, was not clearly presented to the villages or understood by the villages. It also suggests that there was not proper monitoring of the communities to check there commitment to conserving biodiversity (Sinclair et al. 2000). Projects varied considerably from village to village. Several villages received hydroelectricity generators that rely on a regular water supply from the rivers. These in turn rely on the water retention capabilities of the forest, thereby requiring an intact forest. Other schemes with less clear links included the disbursement of chickens and goats for animal husbandry. The link between economic development and good forest management is dubious (Wunder 2001). Across Sumatra wealthier farmers were found to clear more forest than less wealthy farmers (Suyanto and Otsuka 2001). This challenges the logic behind the ICDP, which aimed to promote biodiversity conservation through village development.

During the KS-ICDP operation, Indonesia underwent a dramatic transformation that would have exacerbated attempts to lessen deforestation. Decentralization of the natural resource sector had serious consequences: high and unprecedented levels of illegal logging in Sumatra, to which the KS region was not immune. Efforts to establish conservation incentives by investing in development were thwarted by the dispossession of natural resources by powerful interests outside PAs (McCarthy 2000). Decentralisation in Bolivia has brought benefits to many poor rural people in heavily forested areas. These include greater access to forest resources, restricted encroachment by large timber companies and ranchers, and a greater voice in policy making (Contreras and Vargas 2002, Pacheco 2002). Nevertheless, there are major obstacles that could undermine sustainable forest management and use in Indonesia, including weak local technical capacity, limited national support and organisational problems among small-scale loggers.

There are problems associated with bringing communities into conservation forestry schemes in Indonesia because many of the village heads collude with the illegal loggers, granting permits and receiving taxes levied on loggers and logging trucks (McCathy 2000). In addition, many villagers were employed to extract the timber and became financially dependent on this work. The sharp drop and fluctuating price of important cash crops, such as coffee, has caused many farmers to turn to logging the forests (O'Brien and Kinnaird 2003). This problem may exist not because of insecure cash crop prices but because villagers have no real incentives to counter these logging operations because central government is reluctant to grant them legal rights to the forest (Inamdar et al. 1999).

The forest sector in Indonesia, Malaysia, the Philippines and Thailand has involved large-scale corruption and illegal activity flouting regulations that are designed to control logging (Callaham and Buckman 1981). This has led to small scale agriculture expanding simultaneously with logging. In Indonesia logging concessions are often granted on customary forests. This creates land conflict and usually causes greater conversion of forest to agriculture by small scale farmers staking a land claim before the logging company begins (Angelsen 1997).

A new Forestry Law ratified in September 1999 that was supposed to address these issues still failed to recognize the rights of forest-dwelling people. The process by which the new law was drafted came under heavy criticism for its lack of transparency (Sunderlin 1999). Efforts to improve the government's community forestry program and to draft an *adat* decree to secure the rights of traditional communities were developed at The Forest Law Enforcement and Governance (FLEG) East Asia Ministerial Conference (Anon 2001). In 2001, the Indonesian Government issued a Land Reform and Natural Resource Management Act with a mandate to re-arrange land utilization that included the consideration of local *adat* and cultural diversity. The Government also issued a regulation to encourage the involvement of communities in forest management (Ministry of Forestry 2003). Whether this rhetoric becomes reality is difficult to ascertain because of previous poor forestry governance in Indonesia (World Bank 2000). Unless this happens, communities will have less of a vested interest in conserving forest because of their vague and tenuous private property rights.

Any sense of proprietorship generated by traditional *adat* rights claims has been extinguished by the total disregard for these rights, as illustrated by concessions being granted on "government owned" forestlands. The government allowed the communities and individuals to enjoy their *adat* rights until the land was deemed more profitable as a large-scale timber or plantation operation. The subordination of *adat* rights to those of the timber concessions lead to conflict and a race to clear forest. A good example of this is provided by the debacle in Tapan Valley, West Sumatra province.

Tapan Valley is a lowland area that straddles the western border of KSNP where the local community claimed customary rights to forestland. The government who allocated large tracts of this forest to a logging concession did not recognize these rights. The conversion of forest to agriculture increased dramatically once the logging operation commenced (Linkie et al. 2004). The development of logging roads provided increased access to agricultural sites, and may partly explain the increased deforestation rates, but farmers said that they wanted to secure an agricultural livelihood before traditionally managed forest was cleared by the logging operations (WWF 1999). Forest clearing gives the farmers land rights and deforestation presents a title establishment strategy and an investment (Angelsen 1999).

The situation in Tapan Valley is particularly sad because it was an area of good tiger habitat with the highest abundance of tigers recorded from KSNP. The rapid rate of forest habitat clearance and degradation soon reduced this to poor tiger habitat (Linkie et al. 2003). The forest fragmentation patterns in the KS region are predicted to split KSNP into three sections in the near future. This is problematic because larger PAs have the advantage of less pronounced edge effects (Woodroffe and Ginsberg 1998). The next chapter, therefore, investigates if these edge effects caused by forest conversion to farmland will influence crop raiding in the KS region.

Chapter 7

DETERMINANTS OF CROP RAIDING PATTERNS

The *Amorphophallus titanium* growing in a recently cleared farmland (J.Holden)

7.1 INTRODUCTION

Protected area (PA) networks have two main roles: they should represent national biodiversity and should remove this biodiversity from the factors that threaten it (Margules and Pressey 2000). In reality many PAs are under-funded and this makes achieving their protective function difficult. So when wildlife that does not recognize boundaries ventures outside a PA its protection is even less certain. In developing countries, the situation is compounded further because PAs are becoming more isolated by human settlements, agricultural development, and the active elimination of wildlife on these lands (Newmark 1996).

Communities that share their range with wildlife often bear most of the costs of conservation, yet receive few or no benefits or compensation (Kiss 1990). Crop damage caused by raiding wildlife is one of the most prevalent forms of human-wildlife conflict. It occurs wherever wildlife and cultivation coincide. The relative economic losses suffered by the farmers tend to be high because the farmers are poor and are rarely compensated (Rao et al. 2002, Sekhar 1998). This can make communities antagonistic and intolerant towards wildlife, which will undermine and impede conservation strategies (Nyhus et al. 2000). It is therefore necessary to mitigate this form of human-wildlife conflict but, in order to do so more effectively, it is also necessary to determine the spatial factors that predict crop damage (Sitati et al. 2003). It is also necessary to identify the species that cause the greatest amounts of crop damage because farmer's perceptions of the most notorious crop pests may not be correct (Siex and Struhsaker 1999).

To date, most research into human-wildlife conflict arising from crop damage has been conducted in Africa (Naughton-Treves et al. 1999, Smith and Kasiki 2000, Hill et al. 2002). There is a lack of systematic studies of crop damage from Asia, but the extensive clearance of forest for agriculture currently underway on this continent is likely to cause increasing human-wildlife conflict. The situation in the KS region embodies all of these problems and offers the opportunity to investigate crop damage around a large PA. This is a particularly pressing issue across the KS region, because the current and predicted forest fragmentation patterns will increase forest-farmland edge along KSNP and bring humans and wildlife into closer contact.

7.1.1 *Aims and objectives*

This chapter aims to:

- determine farmer's perceptions of wildlife as crop pests;
- determine the main crop pests and their crop preferences; and,
- determine the spatial factors explaining crop raiding patterns.

7.2 METHODS

7.2.1 *Field methods*

The fifty farms previously surveyed in Air Dikit (section 3.2.1) were selected to monitor crop damage over a five month period between November 2001 and March 2002 (Figure 7.1). The selection criteria were that farms should be located within 2 km of the forest edge and that farmers should be willing to participate in this study. From the outset it was explained to farmers that this research was for academic purposes and that no financial benefits, such as compensation from KSNP, would accrue as a result of the research.

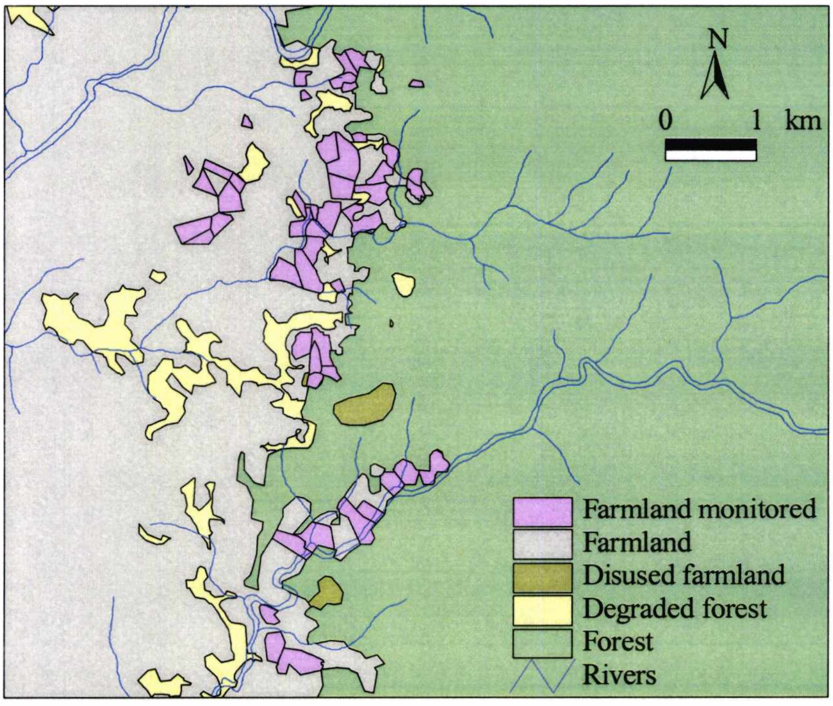


Figure 7.1: Classified land cover map of Air Dikit study site

Crop damage was measured by visiting each of the farms every two days. Farmers were asked to identity recent (1-2 days old) incidents of crop raiding locations. Farms were also searched independently for signs of recent crop damage. When a crop raiding incident was confirmed or discovered, its location was recorded using a GPS. The species responsible was identified by the presence of footprints, faeces, or hair by the damaged crop. Occasionally farmers would directly observe crop raiding. Species identification by farmers was considered reliable, when they were able to correctly name these species from pictures in '*A field guide to the mammals of Borneo*' (Payne and Francis 1985). Direct sightings were always verified where possible through secondary signs. These data allowed calculation of frequency of crop raids by all wildlife species collectively and individually for each farm. Actual crop damage was measured by recording the crop species name, its stage of maturity (seedling, juvenile, fully grown), and the length and width of the area damaged (m^2).

Each farm was also surveyed as part of a wider household survey to obtain information on the crop guarding measures practiced, including guard dogs, noisemakers, guard huts, guns, and fire (Appendix 2). A crop guarding index was constructed from the number of guarding measures in place on each farm. The use of one measure would receive a point, so that no guarding would receive 0 points whereas the full suite of guarding measures would receive 5 points. Finally, the species that farmers thought to be the worst crop pests were obtained from the household surveys. Farmers were asked to rank the top three crop pest species.

7.2.2 GIS methods

Information on the landscape values such as mean elevation, nearest distance to a river, nearest distance to the forest edge, and farm perimeter length, were extracted for each farm from the datasets developed in Chapter 4.

7.2.3 Statistical methods

The crop raiding, socio-economic, and landscape data were imported into SPSS v.11. The continuous data were logarithmically transformed to improve their normality. From this, the total number of forays, the total crop area damaged, and the mean crop area damaged was calculated for wildlife species, both collectively and individually, and for each farm. A Mann-Whitney U test was performed to determine if there was any difference in the amount of crop damage between the two most prolific crop pests.

A stepwise multiple linear regression model was used to determine which combination of landscape and guarding factors best explained crop raiding frequency for wildlife, both collectively and individually. The performance of the model was evaluated by calculating the r^2 value and the presence of spatial auto-correlation in the model was tested by calculating Moran's *I* statistic using Crime-Stat v1.1. Correlation coefficients were used to test for problems with multicollinearity, (correlations between independent variables). This analysis was then repeated to determine which combination of landscape and guarding factors best explained the amount of crop raiding damage for wildlife, both collectively and individually.

7.3 RESULTS

7.3.1 Crop protection

A minority of farmers employed some form of crop protection (30%). The main crop protection strategy was use of a guard dog (24%) followed by a gun (6%) (Figure 7.2).

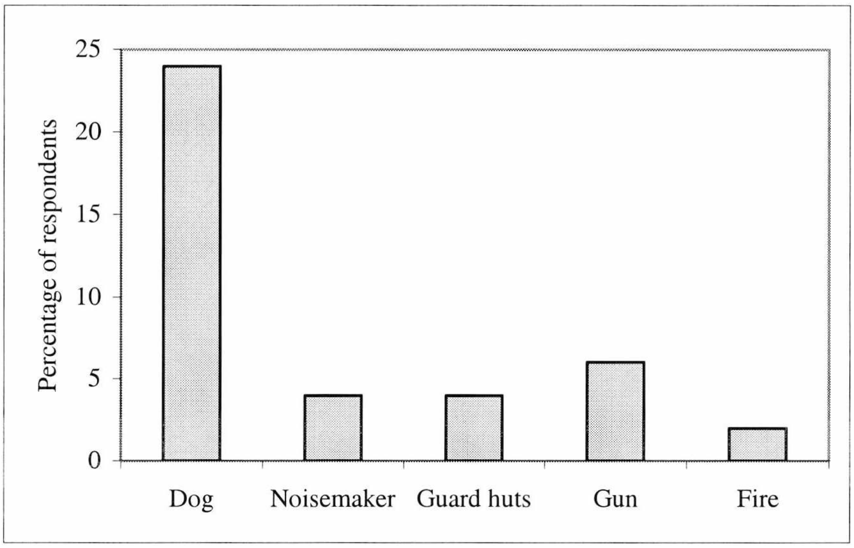


Figure 7.2: Strategies employed by farmers to deter wildlife from crop raiding

7.3.2 Perceived wildlife crop pests

Farmers listed a total of seven problematic wildlife species (Figure 7.3). Most (80%) farmers said that wild boar was the worst crop pest, while pig-tailed macaque was the next worst (75%). Other species like porcupine, bearded pig, muntjac, sambar, and banded langur were regarded as less problematic crop pests.

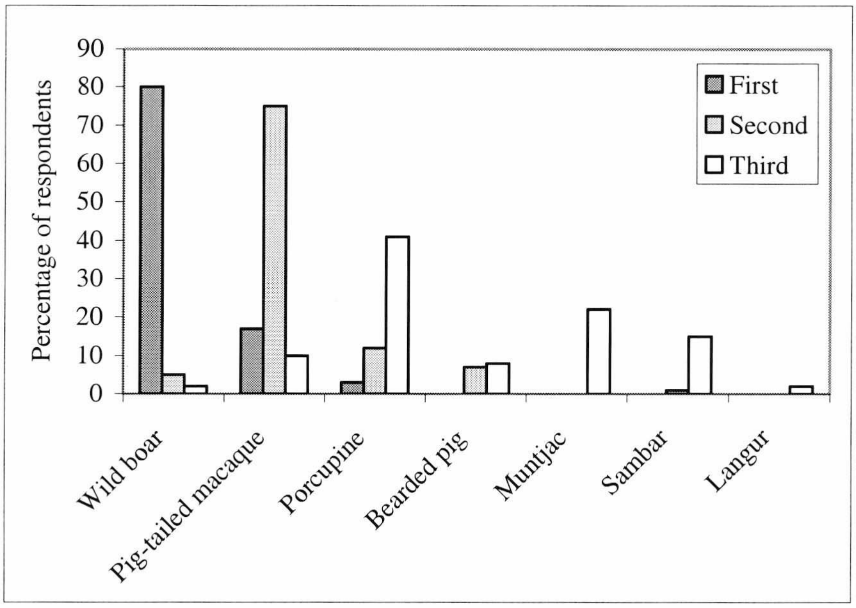


Figure 7.3: Farmer's rankings of the most destructive crop pests

7.3.3 Observed wildlife crop pests

Over five months, a total of 348 independent crop raiding forays were recorded. The crop raiding forays damaged a total crop area of 1420.97 m^2. The mean damage per foray was 4.08 m^2 for all animals. Five different species were recorded as damaging crops, although two species, wild boar and pig-tailed macaque, were actually responsible for 89% of the forays and 99% of the damage (Table 7.1). Wild boar raided crops most frequently (76.4%), followed by pig-tailed macaque (12.6%), but pig-tailed macaque caused significantly more damage (73.1%) than wild boars (25.9%) when crop raiding (n = 310, Z = -4.970, U = 3116.0, $P < 0.0001$; Figures 7.4 and 7.5).

Table 7.1: Frequency and amount of crop damage by animals around KSNP

Animal	Number of events	Percentage of forays	Mean damage (m^2) ± SE	Total area damaged (m^2)	Percentage of total damage caused
Pig-tailed macaque	44	12.6	23.58 ± 1.29	1037.63	73.1
Wild boar	266	76.4	1.39 ± 0.31	368.35	25.9
Porcupine	20	5.7	0.50 ± 0.09	9.97	0.7
Muntjac	16	4.6	0.29 ± 0.17	4.58	0.3
Sambar	2	0.6	0.23 ± 0.02	0.45	0.03

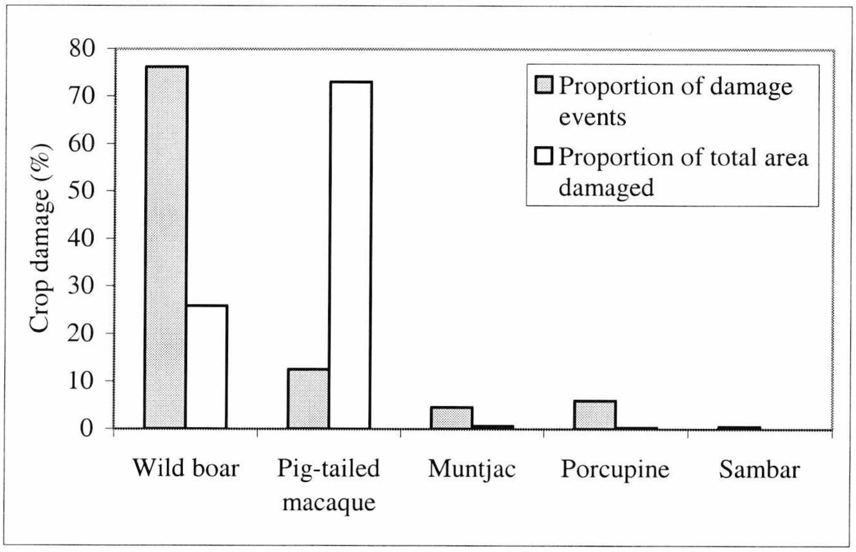


Figure 7.4: Relative contributions by principal species to total number of damage events (n = 348) and total area damaged (n = 1420.97 m^2)

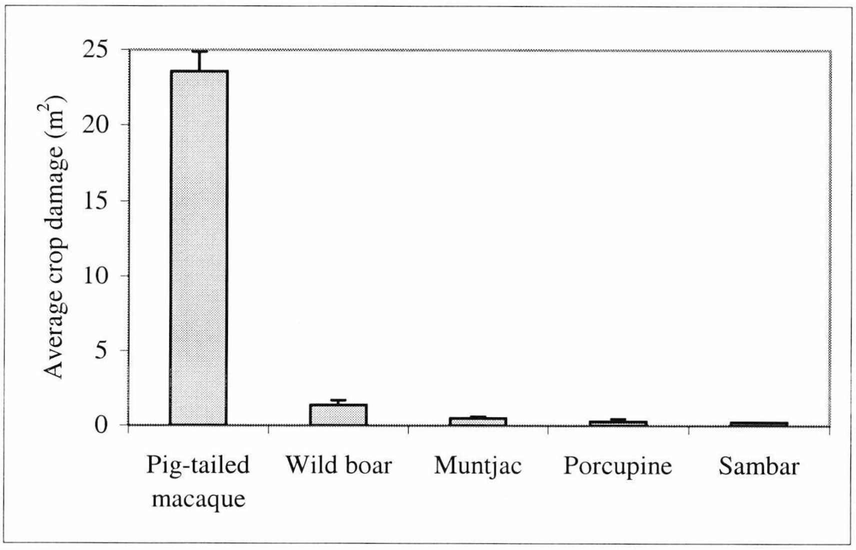


Figure 7.5: Mean ± SE of damage by different crop pests, ranked according to the most destructive species

7.3.4 *Crop preference*

Overall, each problem species differed in their preference for particular crops, although similarities were found between porcupine, muntjac and sambar for chilli and by pig-tailed macaque and wild boar for bananas (Table 7.2). Pig-tailed macaques would almost always eat fully grown crops, which is probably why they caused the most damage. Wild boar were unselective and were found to eat crops at all stages of maturity, while their catholic diet was reflected by their consumption of 21 different crop varieties during this study.

Table 7.2: Preferred crops and parts consumed by animals around KSNP

Animal	Variety of crops eaten	Top two preferred crops	n	Stage of maturity of crops eaten (%)		
				Seedling	Partially grown	Fully grown
Pig-tailed macaque	7	Banana *Musa spp.*	28	0	7.1	92.9
		Eggplant *Solanum melongena*	4	0	0	100
Wild boar	21	Banana *Musa spp.*	82	12.2	36.7	51.1
		Cassava *Manihot esculenta*	46	30.4	65.2	4.4
Porcupine	2	Chilli *Capsicum frutescens*	19	0	36.8	63.2
		Banana *Musa spp.*	1	0	100	0
Muntjac	3	Chilli *Capsicum frutescens*	10	0	40	60
		Water spinach *Ipomonea aquatica*	4	50	0	50
Sambar	1	Chilli *Capsicum frutescens*	2	0	0	100
		n/a				

7.3.5 *Spatial distribution of crop damage*

Of the 50 farms monitored during this study, 40 (80%) were crop raided (Figure 7.6). The intensity of crop raiding and the area damaged by all wildlife was related to $\log_{10}$ distance to forest edge (Table 7.3, Figures 7.7 and 7.8). Wildlife most frequently entered and caused the greatest amount of damage in farms that were closest to the forest edge. The multiple linear regression model of crop raiding intensity explained 54.3% of the original observations and was not affected by spatial autocorrelation (Moran's I = 0.03, $P > 0.1$). The multiple linear regression model of crop raiding damage explained 61.8% of the original observations and was also not affected by spatial autocorrelation (Moran's I = -0.01, $P > 0.1$). There was no multicollinearity in the model, indicating no correlation between the independent variables.

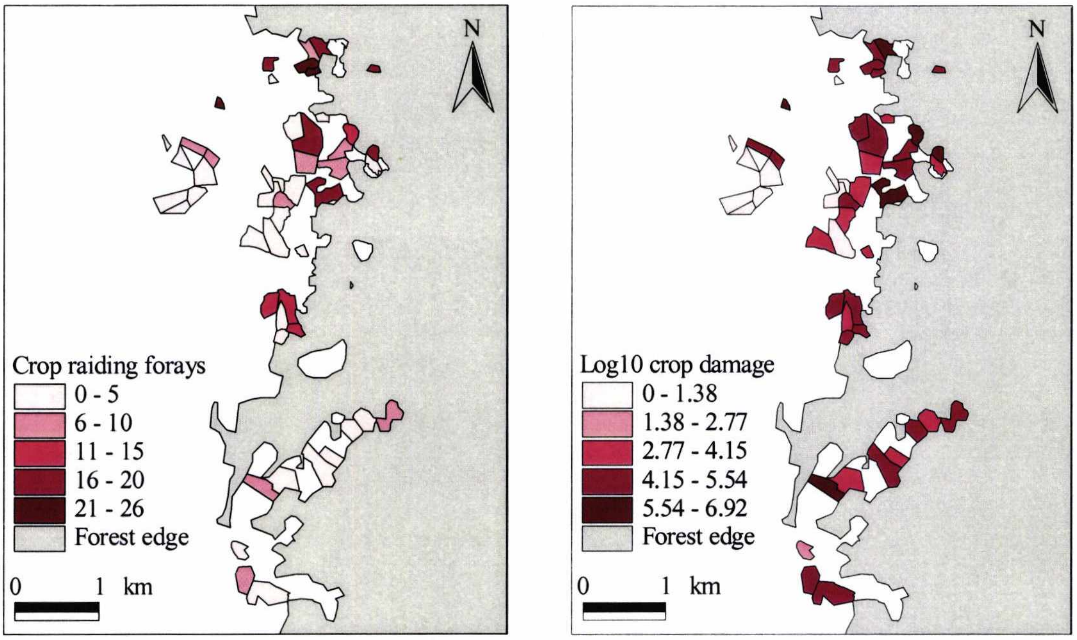


Figure 7.6: Distribution of crop raiding by all wildlife species

Table 7.3: Best multiple linear regression model describing the relationship between the spatial factors and crop raiding intensity and crop area damaged by all wildlife species

Factor	Coefficient (β) ± SE	t	*P*	r^2
Intensity				
Log_{10} distance to forest edge forest	-0.702 ± 0.158	4.437	< 0.0001	0.295
Constant	2.579 ± 0.424	6.087	< 0.0001	
Damage				
Log_{10} distance to forest edge forest	-3.551 ± 0.659	5.390	< 0.0001	0.382
Constant	13.134 ± 1.764	7.445	< 0.0001	

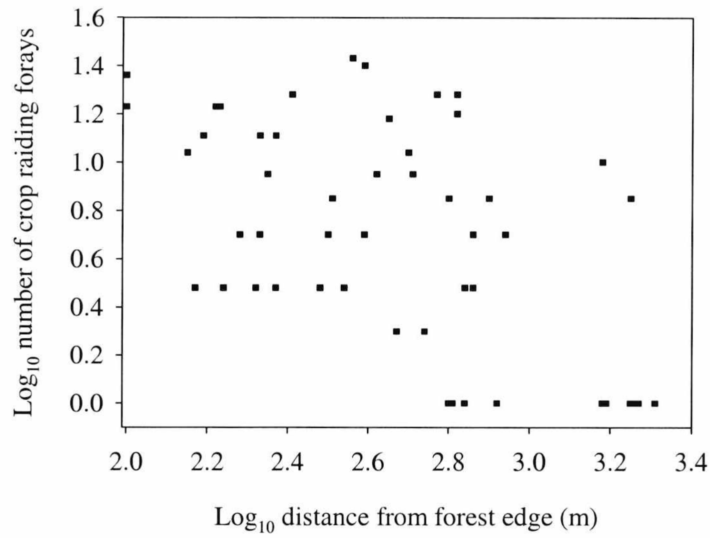


Figure 7.7: The relationship between crop raiding intensity by all wildlife species and distance from the forest edge (r_p= 0.543)

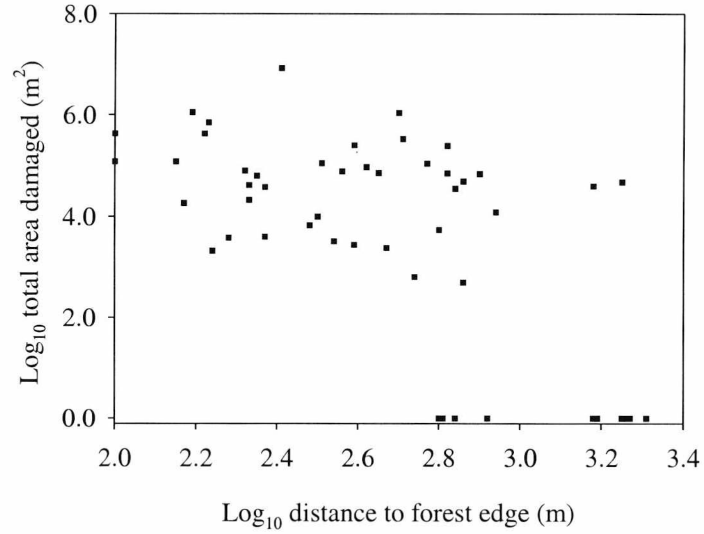


Figure 7.8: The relationship between crop raiding damage by all wildlife species and distance from the forest edge (r_p= 0.618)

Crop raiding by wild boar (n = 266) was widespread and occurred on 38 (76%) of the farms (Figure 7.9). The intensity and amount of crop damage caused by wild boar during crop raiding forays was related to log_{10} distance to forest edge (Table 7.4). Wild boar most frequently entered and caused the greatest amount of damage in farms that were closest to the forest edge. The multiple linear regression model of crop raiding intensity explained 43.9% of the original observations and was not affected by spatial autocorrelation (Moran's I = 0.07, P > 0.1). The multiple linear regression model of crop raiding damage explained 51.8% of the original observations and was also not affected by spatial autocorrelation (Moran's I = 0.04, P > 0.1). There was no multicollinearity in the model, indicating no correlation between the independent variables.

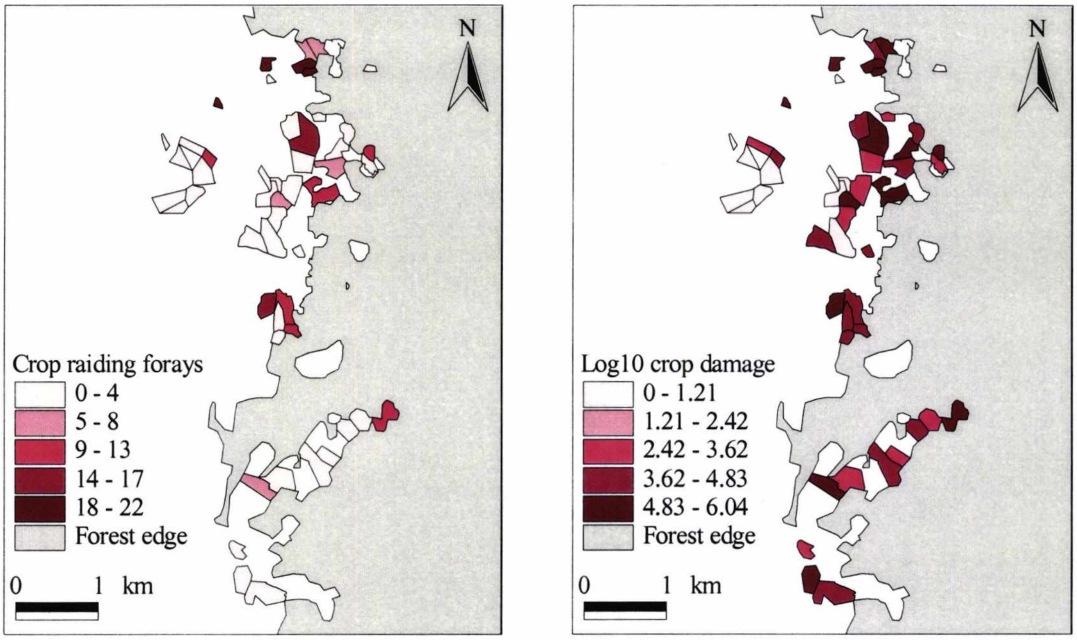


Figure 7.9: Distribution of crop raiding by wild boar

Table 7.4: Best multiple linear regression model describing the relationship between the spatial factors and crop raiding intensity and crop area damaged by wild boar

Factor	Coefficient (β) ± SE	t	P	r^2
Intensity				
Log_{10} distance to forest edge forest	-0.531 ± 0.158	3.350	0.0020	0.193
Constant	2.042 ± 0.424	4.813	< 0.0001	
Damage				
Log_{10} distance to forest edge forest	-2.884 ± 0.694	4.157	< 0.0001	0.269
Constant	11.081 ± 1.858	5.964	< 0.0001	

Crop raiding by pig-tailed macaque (n = 44) was highly localized and occurred on 11 (22%) of the farms (Figure 7.10). The intensity and amount of crop damage caused by pig-tailed macaque during crop raiding forays was related to log_{10} distance to forest edge (Table 7.5). Pig-tailed macaques most frequently entered and caused the greatest amount of damage in farms that were closest to the forest edge. The multiple linear regression model of crop raiding intensity explained 30.7% of the original observations and was not affected by spatial autocorrelation (Moran's $I = 0.02$, $P > 0.1$). The multiple linear regression model of crop raiding damage explained 29.4% of the original observations and was also not affected by spatial autocorrelation (Moran's $I = 0.01$, $P > 0.1$). There was no multicollinearity in the model, indicating no correlation between the independent variables.

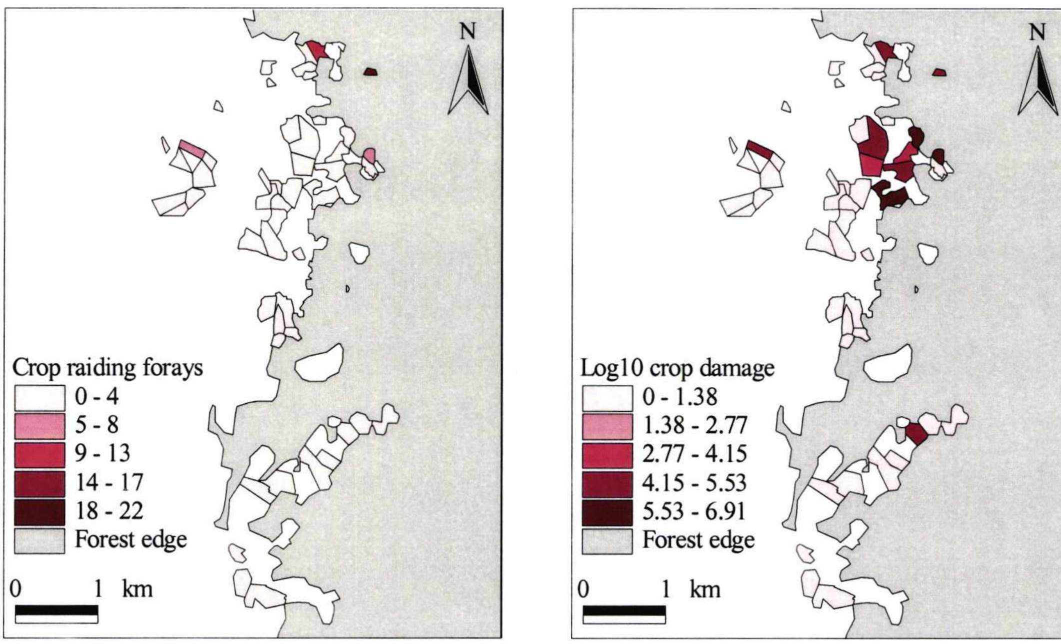


Figure 7.10: Distribution of crop raiding by pig-tailed macaque

Table 7.5: Best multiple linear regression model describing the relationship between the spatial factors and crop raiding intensity and crop area damaged by pig-tailed macaque

Factor	Coefficient (β) ± SE	t	P	r^2
Intensity				
Log_{10} distance to forest edge forest	-0.244 ± 0.929	2.213	0.032	0.094
Constant	0.787 ± 0.295	2.669	0.010	
Damage				
Log_{10} distance to forest edge forest	-1.761 ± 0.837	2.105	0.041	0.086
Constant	5.798 ± 2.240	2.588	0.013	

7.4 DISCUSSION

The crop raiding patterns recorded in this study highlight the vulnerability of farms at the forest edge and the lack of success of the different guarding strategies used. More appropriate protection strategies, such as using natural barriers like thorny shrubs for wild boar and planting crops close to farmhouses and guarding against pig-tailed macaques with slingshots may help to lessen crop raiding frequency and damage. The difference in the perceived and the observed crop pests illustrates the need to investigate community perceptions associated with crop damage, as well as wider issues of human-wildlife conflict, because they need to be clearly understood if they are to guide conservation management decisions.

The farmers in Air Dikit perceived wild boar to be the worst crop pest, although this did not corroborate with my observations on the amount of damage caused. Wild boar and pig-tailed macaques were observed as the most notorious crop raiding species. However, these species exhibited considerable variation between crop raiding intensity and damage. Although wild boar would most frequently enter farms to raid crops, they caused significantly less damage per foray and overall than pig-tailed macaque, which raided less frequently but caused much greater damage.

The propensity of wild boar to raid crops more frequently than pig-tailed macaque may have misled farmers to believe that they were the worst crop pests. This is contrary to research from Uganda where baboons, which caused the most cumulative damage, were considered by farmers as worse crop pests than red-tailed monkeys, which raided most frequently (Naughton-Treves 1997). Wild boar may have received a disproportionate amount of the blame for crop damage because they were more conspicuous as a consequence of their frequent raiding. In Zanzibar, negative attitudes toward red colobus monkeys may have been linked to farmers wrongly blaming these monkeys for banana damage caused by the smaller and less conspicuous Sykes monkey that often intermingles with red colobus. The farmer's main complaint was about the negative impact of coconut consumption by red colobus monkeys. However, quantitative study found that monkeys, which forage on immature coconuts, were actually associated with higher harvests, possibly due to a pruning effect (Siex and Struhsaker 1999).

Wild boar can cause a large amount of collateral damage when crop raiding and this may have influenced farmer's perceptions. Kristiansson (1985) estimated that between 90-95% of crop destruction by wild boar was due to trampling, the remainder being through direct consumption of crops. The Muslim farmers in Air Dikit may have perceived that wild boar offered them no benefits because they do not hunt them for their meat for religious reasons. Although not practised in this study site, sport hunting of wild boar is an important social event among males in other ethnic Muslim groups living adjacent to KSNP. The negative attitudes towards wild boar in this study may then be related to their crop preference.

In this study, crop preferences of wild boar and pig-tailed macaque differed. Both wild boar and pig-tailed macaque ate bananas, but wild boar also ate the more important staple crop cassava, used for its leaves and tubers. In Tanzania, resentment was greater towards bush pigs that ate a 'famine' crop such as cassava (Mascarenhas 1971). In this study crop selection was based on preference because the same crops were continuously available.

The factors that explain crop damage patterns have been found to vary between different species (Hill 1997). In this study, crop damage patterns for all wildlife species both collectively and individually were explained by a single factor: proximity to forest edge. The vulnerability of farms that borders or that is in close proximity to the forest edge has been well documented (Jhala 1993, Naughton-Treves 1998, Hill 2000, Saj et al. 2001). Farmland that is closer to a PA border is also more prone to crop damage, probably because this is acting as a surrogate for proximity to forest edge (Studsrod and Wegge 1995, Sekhar 1998). The patterns of crop raiding by pig-tailed macaques were similar to those of elephants in that they were highly localized and inflicted large losses on farmers (Sitati et al. 2003). Surprisingly the clustering of crop raiding patterns by pig-tailed macaque did not exhibit signs of spatial autocorrelation as might have been expected (Sitati et al. 2003). This may have been due to forest proximity explaining the crop raiding patterns and there being spatial autocorrelation within this factor. With further studies it would be interesting to determine what spatial factors best explain the distributions of these crop raids, because their relationship with proximity to forest edge was weak.

At the current rate of forest clearance and its ongoing fragmentation by agriculture expansion throughout the KS region, an omnivorous and adaptable species such as wild boar may prosper over those species requiring intact forest. Nevertheless, this and other studies have shown that large mammal crop pests still require a forest habitat refuge and do not travel into densely settled agriculture (Else 1991, Naughton-Treves 1998). The human settlements bordering KSNP receive modest benefits from living with the costs of this destructive wildlife. Retribution killings occur in farmland across the KS region because the farmers are intolerant of crop pests, which in turn jeopardize the long term survival of these species, which are also important tiger prey (Hartana and Martyr 2001). The following chapter now investigates the factors that determine the distribution of the key ungulate prey species of tigers and the factors that determine snare trap presence in the KS region.

Chapter 8

TIGER PREY IN HUMAN ALTERED LANDSCAPES

Injured muntjac tiger prey inside KSNP

8.1 INTRODUCTION

The decline of tiger prey populations across Asia is subtle and insidious. Yet it occurs in most tiger habitats and at levels that may pose the most serious threat to the survival of the tiger (Karanth and Stith 1999). Across large parts of the tigers range, a substantial number of areas have been identified that still contain suitable forest habitat but no longer contain tiger prey (Karanth 1991, Wikramanayake et al. 1998). This suggests that less visible threats are active, such as the effects of high human activity near forest edges and the poaching or hunting of tiger prey species. Edge effects are often indirect and poaching is often clandestine, making both of these more difficult to detect.

Habitat loss, fragmentation, and conversion are ubiquitous and severe threats for large mammals. The conversion of forest to farmland removes the natural habitat of tiger prey but it also adds a greater abundance of food for these species by transferring the primary productivity from the forest canopy to the ground level (Eisenberg 1980). These edge environments tend to have higher levels of human activity which may offset any benefits accrued from greater food availability and result in a lower abundance of tiger prey (Griffith and van Schaik 1993).

In South Sumatra, wild boar and sambar deer were found to be abundant at the forest-farmland edge in areas that were less disturbed (O'Brien et al. 2003). Tigers were found to undergo a noticeable shift in habitat preference 3 km from the forest edge (Kinnaird et al. 2003). Similarly, logging roads may create secondary forest that offers more food for tiger prey species than does primary forest. The response of tiger ungulate prey species to habitat conversion in such situations is varied (Davies et al. 2001).

Logging activities and agricultural expansion also present another threat to tigers. Logging roads provide access into the forest to hunt or to set snare traps (Bennett and Robinson 2000). Snare traps may be set by farmers around their farmland to reduce crop damage by tiger prey species, such as wild boar and muntjac. The hunting activities and tolerance of local communities living with this wildlife may also be related to their social and economic backgrounds (Shively 1997, Noss and Cuellar 2001).

To determine the ambiguous response of tiger prey to the different threats associated within human-dominated landscapes requires the monitoring of a species population trends. Programmes to monitor prey base do not receive as much attention as those for tigers, because prey species are not as high profile and the fieldwork is often more labourious (Karanth and Stith 1999, Karanth et al. 2003). In order to conserve tigers, information is needed on how the physical factors and human-related threats interact to determine the distribution of their prey. This information can be used to construct spatially explicit habitat models (SEHMs), which identify vulnerable areas so that they may then be targeted for special protection. Previous studies on large mammals have typically used presence-absence data as the basis for developing a SEHM (Gross et al. 2002, Johnston et al 2004). Whilst the detection of a species can confirm its presence, the non-detection of a species does not confirm its absence. Therefore, biased estimates can arise in logistic regression model parameters from failing to account for these 'false absences' (MacKenzie et al. 2003, Tyre et al. 2003). MacKenzie et al. (2002) recently developed a new method using repeat presence-absence surveys to allow detection probability to be explicitly incorporated into occupancy models. By relating occupancy to factors that are measured at each site, the method of MacKenzie et al. (2002) can be considered as a generalised logistic regression model that incorporates imperfect detection of the species. This method also provided a rigorous estimate of the proportion of habitat occupied which in turn could be incorporated in SEHMs. Thus, this novel approach is particularly salient to developing a SEHM for tiger prey in the KS region.

8.1.1 Aims and objectives

This chapter aims to:

- determine the physical factors and human-related threats that explain the distribution of tiger prey species in the KS region;
- develop a SEHM for tiger prey based on these factors;
- determine how these factors influence the availability of core tiger prey habitat; and,

- determine the socio-economic and physical factors that explain the distribution of snare traps in the KS region.

8.2 METHODS

8.2.1 Field methods

The presence of tigers and their prey species in and around KSNP were obtained from field surveys conducted between 2000 and 2002. The main prey species of tiger were recorded, but only the key ungulate prey of wild boar, sambar, muntjac, and mouse deer were included in the analysis (O'Brien et al. 2003). Bearded pigs were not included in the analysis because they are an unpredictable and migratory prey species, which makes them an irregular quarry (Linkie and Holden 2002). Additional data on tiger prey distributions inside KSNP were provided by the KSNP Tiger Protection Units (KS-TPCUs) from forest patrols. Additional data on tiger prey distributions inside adjoining logging concessions to KSNP were provided by the KS-ICDP from biodiversity surveys. All data collection including those by KS-TPCUs and KS-ICDP staff adhered to a set protocol. A total distance of 4614.3 km was walked through 129 transects with a mean length of 3.58 km, and a range of 1.05-7.53 km. Transects followed pre-existing animal or topographic trails, such as hill and mountain ridges that the focal species and humans would typically use. The location of each transect was recorded using a geographic position system (GPS) unit and compass bearings with 1:50,000 topographic paper maps. On each transect, the presence or absence of tigers (pugmarks and faeces), of their prey species (direct sighting, prints and faeces), and of human-related threats (snare trap set for prey, illegal logging, human disturbance), were all recorded. Only five tiger snare traps were recorded during this study. This did not present a sufficient sample size to confidently test for the effect of tiger poaching: snare traps are easily the most common method used for this activity. Tiger snare traps, which would still be able to trap tiger prey species, were therefore amalgamated with prey snare trap data to form the factor of prey snare trap presence. This factor was used as an indicator for tiger prey poaching. Human disturbance was recorded as the presence of machete marks, footprints, fireplaces, disused camps, litter and direct encounters. Illegal logging was recorded as the presence of unnaturally felled trees.

8.2.2 GIS methods

The 2 km^2 was sampling grid constructed for all forested areas in the KS region. The transect routes were superimposed onto the grid to identify those cells which had been surveyed cells. For each of these cells, information on tiger presence, prey species presence, landscape characteristics, and human-related threats was extracted (Table 8.1). The spatial data for roads, logging roads, settlements, rivers, elevation, slope, PA status, and soil type were taken from the datasets constructed in Chapter 4. The first analysis determined the spatial factors that best explained individual tiger prey distribution. The second analysis determined the spatial factors which best explained all tiger prey distribution. If the second included factors for which there was only partial coverage then the data were reanalysed to include only those factors for which there was complete coverage. This was necessary so that a tiger prey SEHM could be developed for the entire KS region. Finally, the third analysis determined the spatial factors which best explained snare trap presence in the KS Region.

Table 8.1: Spatial information obtained for each sampling cell

Factor	Coverage
Proximity to nearest public road	Complete
Proximity to nearest logging road	Complete
Proximity to nearest settlement	Complete
Proximity to nearest river	Complete
Elevation	Complete
Slope	Complete
PA status	Complete
Soil type	Complete
Prey snare trap recorded	Partial
Human disturbance	Partial
Illegal logging recorded	Partial
Prey species recorded	Partial

The continuous data were logarithmically transformed and along with the categorical data were extracted for each cell and imported into PRESENCE software package (Proteus Wildlife Research Consultants, New Zealand; http://www.proteus.co.nz). For each individual tiger prey species detection (1) and non-detection (0) data were then entered to provide information on the detection history of each sampling cell for each sampling occasion (Figure 8.1). This process was then repeated for all the tiger prey species combined.

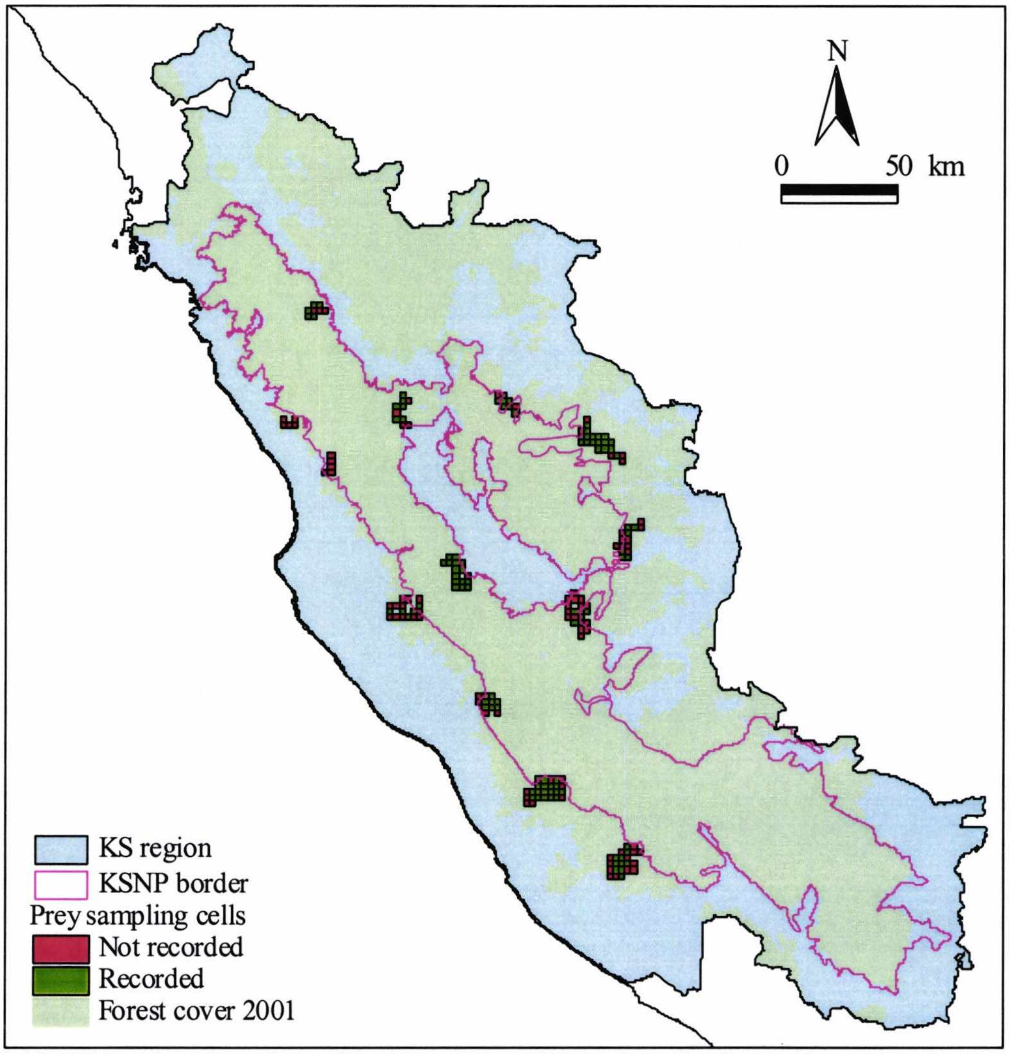


Figure 8.1: Sampling grid containing 200 cells with tiger prey detection and non-detection

8.2.3 Statistical methods

The effect of each variable on the probability of tiger prey presence was first tested by a univariate analysis to determine if any of the factors were interdependent. To estimate the proportion of area occupied by each tiger prey species in the KS region a general likelihood model was constructed that incorporated potential landscape covariates influencing the detection probability of tiger prey species. A multiple logistic regression analysis was performed to determine which of the landscape factors best explained the detection of tiger prey species in the KS region. PRESENCE was used to generate a combination of models that were ranked by their Akaike Information Criterion (AIC) values to determine the most parsimonious model that best fitted the data (Burnham & Anderson 1998). From this, tiger prey species detection and area of occupancy was estimated for the KS region.

The presence of spatial autocorrelation in the model was tested by calculating the Moran statistic of the regression unstandardized residuals using the Crime-Stat software (Levine 2000). The significance of Moran's *I* was examined using a *Z*-test. If spatial autocorrelation was found to be present in the logistic regression model then the analysis was repeated with the inclusion of an autocovariate term to explicitly model this phenomenon (Augustine et al. 1996). The purpose of the autocovariate term was to improve the model fit and reduce the likelihood of including spurious factors in the final model. An inverse Euclidean distance weighted mean of the detection or non-detection of the focal species in the eight surrounding cells of each cell in the grid was used to obtain this term (Sitati et al. 2003). This process was then repeated for all tiger prey species combined to produce the final logistic regression model.

From the final logistic regression model the probability of tiger prey presence (*P*) was determined by,

$$\mathrm{Y} = \beta_0 + \sum \beta_i \mathrm{X}_i$$

where β_0 is the constant coefficient, β_i represents the significant independent variable coefficients, and X_1 represents their associated independent factors. Through incorporating the natural exponential (*e*) into the previous equation the tiger prey SEHM for the KS region was constructed by,

$$P = e^{\mathrm{Y}} / 1 + e^{\mathrm{Y}}$$

This equation was used to assign each 100 m^2 block of forest a value for the probability of tiger prey presence (0 = lowest probability, up to 1 =high probability). These values were used as a surrogate for tiger prey habitat quality. Therefore, values ranged from 0, indicating the poorest quality habitat, to 1, indicating the best quality habitat. The accuracy of the model predictions was independently validated using tiger prey detection and tiger prey non-detection data collected from a single survey in the 100 additional monitoring sites, which were not included in the regression analysis. To minimize potential problems with 'false absences', the sampling effort in these monitoring sites was increased. From these data, 45 cells recorded tiger prey presence. For parity, another 45 cells with no tiger record were then randomly selected from the remaining 55 sites. The SEHM probability value for these sets of cells was extracted. A Mann-Whitney U test was then used to find whether those sites where tiger prey were detected had a higher predicted probability than those sites where tiger prey were not detected. Using the significant factors that explained prey base distribution, a 3D mesh plot was constructed using SigmaPlot v.8 software package (SPSS Inc., Chicago, IL). This showed how the interactions of the significant factors influenced prey habitat preference.

The tiger prey SEHM was reclassified, using the ***RECLASS*** module in Idrisi, to include the proportion of area occupied based on the most suitable habitat available. This allowed the location of core tiger prey habitat with their unique populations to be identified. These core areas were then treated independently. For each core tiger prey area the mean distance from core perimeter to the forest edge was calculated using the ***EXTRACT*** module in Idrisi. The amount of lowland, hill, submontane and montane forest was then calculated.

Finally, a multiple logistic regression was performed to determine which of the landscapes factors best explained snare trap presence in the KS region. Data from each sampling cell were imported into SPSS software package and the best logistic regression model identified. The performance of the model was evaluated by calculating the area under the curve (AUC) of the receiver operating characteristics plot (as in section 3.2.4). The presence of spatial auto-correlation in the model was tested by calculating the Moran statistic of the regression unstandardized residuals using the Crime-Stat software.

8.3 RESULTS

8.3.1 Factors determining distributions of individual prey species

8.3.1.1 Mouse deer

From the 200 sampling cells surveyed, mouse deer were detected at 26 locations. This represented 11 sites from the first occasion, 14 sites from the second occasion and 1 sites from both the first and second occasions. Mouse deer presence across the KS region was related to $\log_{10}$ distance to public roads (Table 8.2). Mouse deer were more likely to occur in areas that were further away from public roads (Figure 8.2). From this, the best logistic regression model (AIC = 191.66) had a mean detection probability of 0.338 (± 0.099) was not affected by spatial autocorrelation (Moran's I = -0.02, $P > 0.1$) and gave an overall estimate for the proportion of sites occupied by mouse deer as 0.603 (± 0.073).

Table 8.2: Best multiple logistic regression model describing the relationship between landscape factors and mouse deer presence in the KS region

Factor	Coefficient (β) ± S.E.	df	*P*
$\log_{10}$ distance to public roads	9.076 ± -5.504	1	< 0.05
Intercept	-33.361 ± -1.509	1	< 0.05

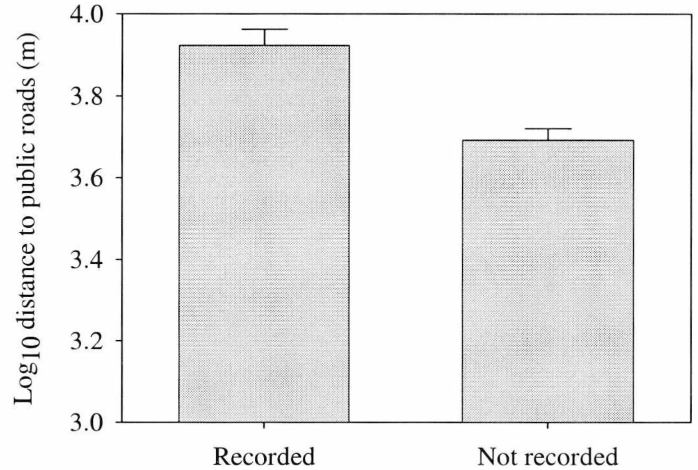


Figure 8.2: Presence of mouse deer relative to mean $\log_{10}$ distance to the nearest public road (with S.E. bars)

8.3.1.2 Muntjac

From the 200 sampling cells surveyed, muntjac were detected at 76 locations. This represented 22 sites from the first occasion, 30 sites from the second occasion and 23 sites from both the first and second occasions. Muntjac presence across the KS region was related to $\log_{10}$ distance to public roads (Table 8.3). Muntjac were more likely to occur in areas that were further away from public roads (Figure 8.3). From this, the best logistic regression model (AIC = 410.62) had a mean detection probability of 0.483 (± 0.014), was not affected by spatial autocorrelation (Moran's $I = 0.01$, $P > 0.1$) and gave an overall estimate for the proportion of sites occupied by muntjac as 0.529 (± 0.061).

Table 8.3: Best multiple logistic regression model describing the relationship between landscape factors and muntjac presence in the KS region

Factor	Coefficient (β) ± S.E.	df	*P*
$\log_{10}$ distance to public roads	4.064 ± -0.680	1	< 0.05
Intercept	-15.107 ± -2.482	1	< 0.05

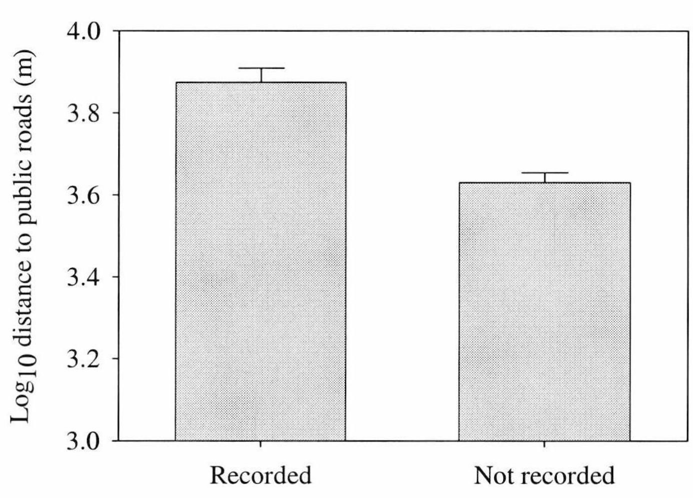


Figure 8.3: Presence of muntjac relative to mean $\log_{10}$ distance to the nearest public road (with S.E. bars)

8.3.1.3 Sambar

From the 200 sampling cells surveyed, sambar were detected at 69 locations. This represented 27 sites from the first occasion, 28 sites from the second occasion and 14 sites from both the first and second occasions. Sambar presence across the KS region was related to $\log_{10}$ distance to public roads and $\log_{10}$ elevation (Table 8.3). Sambar were more likely to occur in areas that were further away from public roads and at higher elevations (Figures 8.4 and 8.5). From this, the best logistic regression model (AIC = 395.41) had a mean detection probability of 0.272 (± 0.017) was not affected by spatial autocorrelation (Moran's $I = 0.01$, $P > 0.1$) and gave an overall estimate for the proportion of sites occupied by sambar as 0.636 (± 0.111).

Table 8.4: Best multiple logistic regression model describing the relationship between landscape factors and sambar presence in the KS region

Factor	Coefficient (β) ± S.E.	df	*P*
Log_{10} distance to public roads	2.266 ± -0.946	1	< 0.05
Log_{10} elevation	3.312 ± -1.519	1	< 0.05
Intercept	-16.342 ± -5.818	1	< 0.05

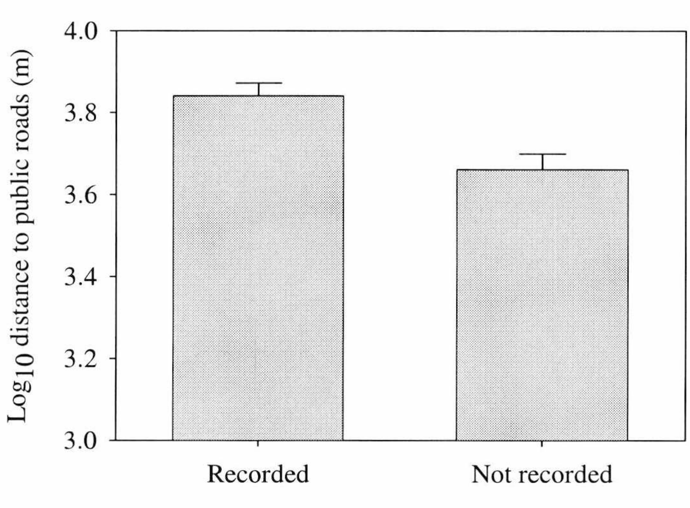


Figure 8.4: Presence of sambar relative to mean $\log_{10}$ distance to the nearest public road (with S.E. bars)

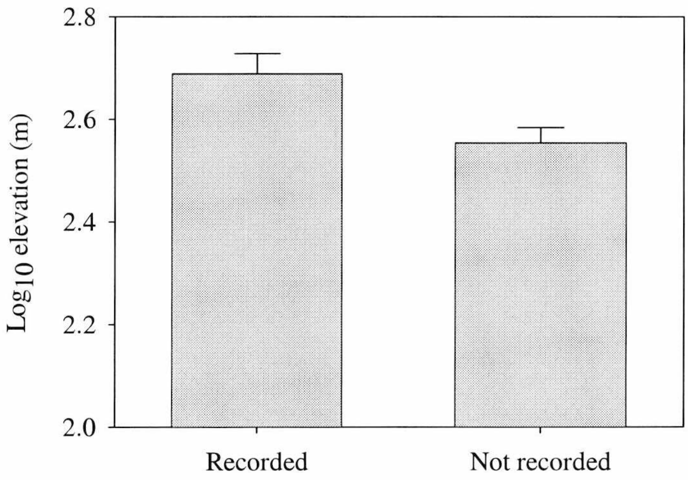


Figure 8.5: Presence of sambar relative to mean $\log_{10}$ elevation (with S.E. bars)

8.3.1.4 Wild Boar

From the 200 sampling cells surveyed, wild boar were detected at 63 locations. This represented 25 sites from the first occasion, 19 sites from the second occasion and 19 sites from both the first and second occasions. Wild boar presence across the KS region was related to $\log_{10}$ distance to public roads and $\log_{10}$ distance to logging roads (Table 8.5). Wild boar were more likely to occur in areas that were further away from public roads and from logging roads (Figures 8.6 and 8.7). From this, the best logistic regression model (AIC = 383.68) had a mean detection probability of 0.437 (± 0.049), was not affected by spatial autocorrelation (Moran's $I = 0.03$, $P > 0.1$) and gave an overall estimate for the proportion of sites occupied by wild boar as 0.490 (± 0.073).

Table 8.5: Best multiple logistic regression model describing the relationship between landscape factors and wild boar presence in the KS region

Factor	Coefficient (β) ± S.E.	df	P
$\log_{10}$ distance to public roads	2.689 ± -0.669	1	< 0.05
$\log_{10}$ distance to logging roads	0.938 ± -0.487	1	< 0.05
Intercept	-13.403 ± -2.592	1	< 0.05

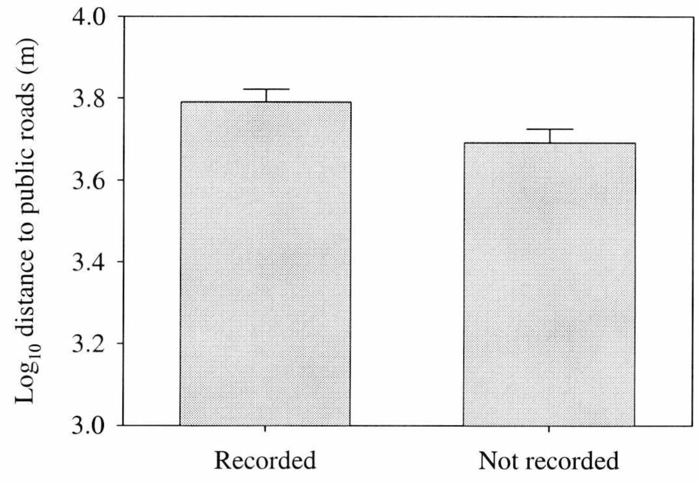


Figure 8.6: Presence of wild boar relative to mean $\log_{10}$ distance to the nearest public road (with S.E. bars)

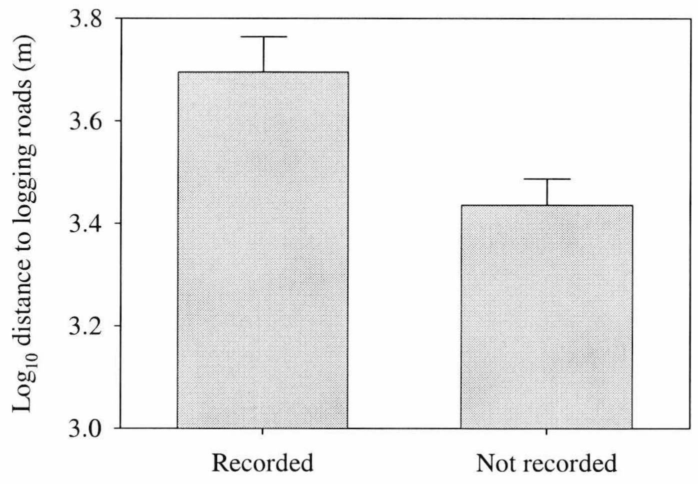


Figure 8.7: Presence of wild boar relative to mean $\log_{10}$ distance to the nearest logging road (with S.E. bars)

8.3.2 Factors determining distribution of tiger prey

From the 200 sampling cells surveyed, all key tiger prey species were detected at 112 locations. This represented 39 sites from the first occasion, 41 sites from the second occasion and 32 sites from both the first and second occasions. Tiger prey presence across the KS region was related to $\log_{10}$ distance to public roads and $\log_{10}$ distance to logging roads (Table 8.6). Tiger prey were more likely to occur in areas that were further away from public roads and logging roads (Figures 8.8 and 8.9). From this, the best logistic regression model (AIC = 514.38) had a mean detection probability of 0.405 (± 0.012), was not affected by spatial autocorrelation (Moran's $I = 0.01$, $P > 0.1$) and gave an overall estimate for the proportion of sites occupied by all tiger prey as 0.791 (± 0.088).`

Table 8.6: Best multiple logistic regression model describing the relationship between landscape factors and tiger prey presence in the KS region

Factor	Coefficient (β) ± S.E.	df	P
Log_{10} distance to public roads	2.742 ± -1.564	1	< 0.05
Log_{10} distance to logging roads	0.651 ± -0.986	1	< 0.05
Intercept	-10.939 ± -2.956	1	< 0.05

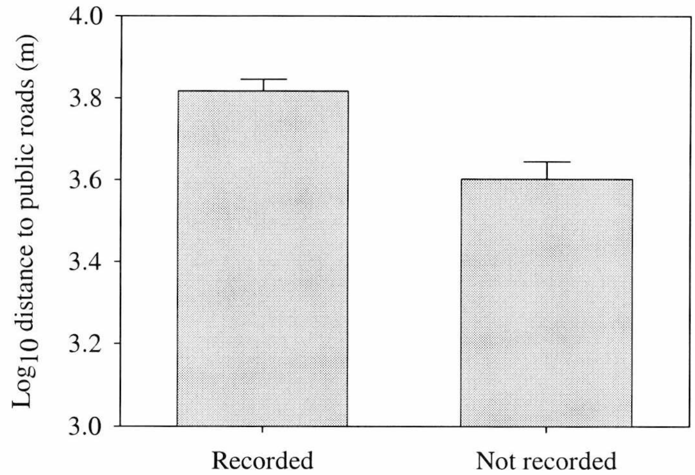


Figure 8.8: Presence of all key tiger prey relative to mean $\log_{10}$ distance to the nearest public road (with S.E. bars)

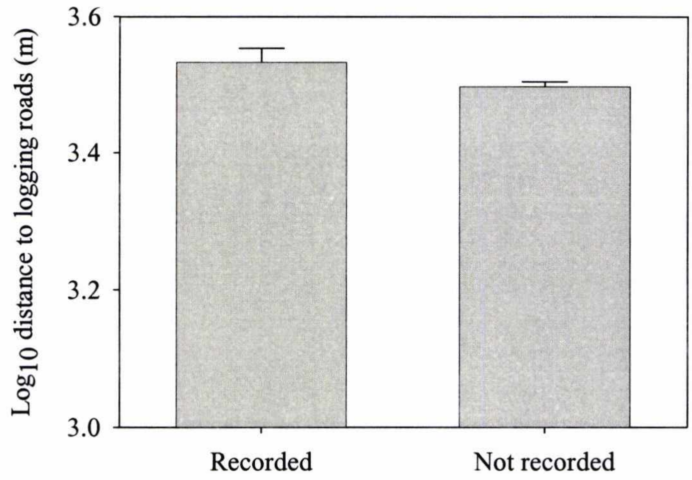


Figure 8.9: Presence of all key tiger prey relative to mean $\log_{10}$ distance to the nearest logging road (with S.E. bars)

8.3.3 Tiger prey habitat preference model

The creation of public roads and of logging roads in and around KSNP had a profound impact on reducing habitat quality for tiger prey species. The influence of public roads had a greater impact on habitat preference of prey species than logging roads (Figure 8.10).

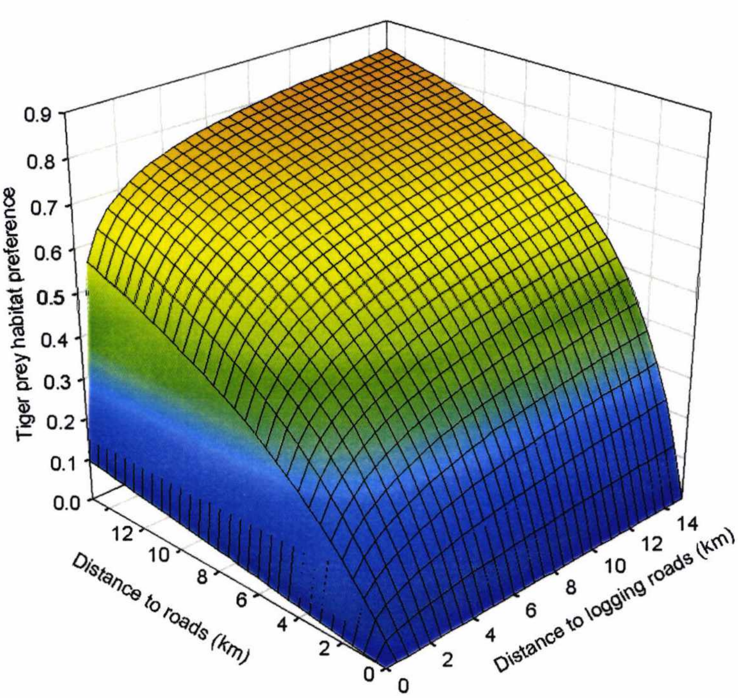


Figure 8.10: The impact of all roads on the presence of tiger prey

Tiger prey species showed a preference for habitat inside KSNP (Figure 8.11). These core areas were remote and inaccessible by the majority of roads in the KS region. The only road close to these areas was a well used asphalt public road running centrally through KSNP that acted as barrier between suitable habitat in the northern and southern sections of KSNP. A similar situation occurs in the eastern section of KSNP where a road divides a block of forest and isolates it from the main section of KSNP. Outside of KSNP the diffuse network of public roads and logging roads renders most of the remaining forest as unsuitable habitat for tiger prey species.

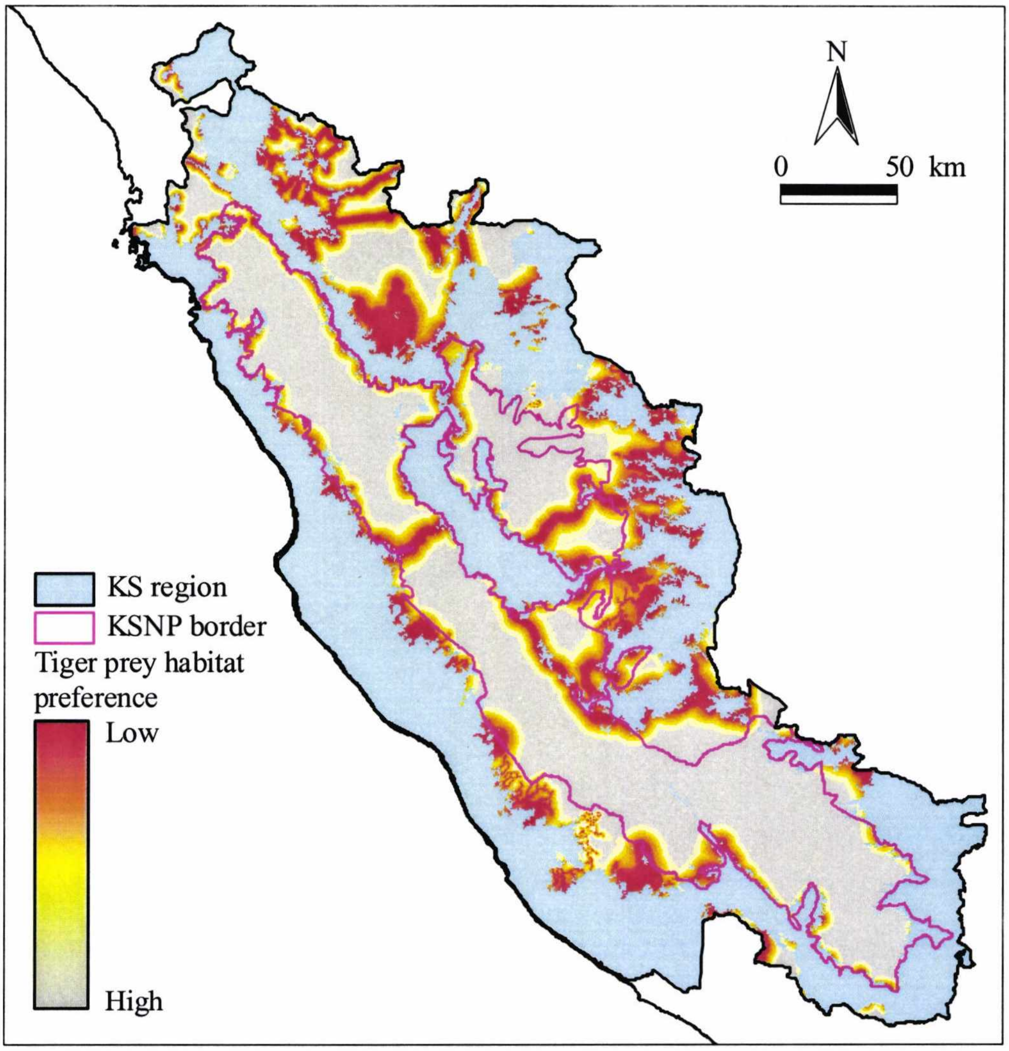


Figure 8.11: Tiger prey habitat preference in the KS region

The tiger prey habitat suitability model predicted that randomly selected sites where prey species were recorded had a value of 0.680 (Figure 8.12). In contrast, the randomly selected sites where prey species were not recorded had a much lower (n = 100, Mann-Whitney U = 141, Z = -7.645, $P < 0.0001$) value of 0.313.

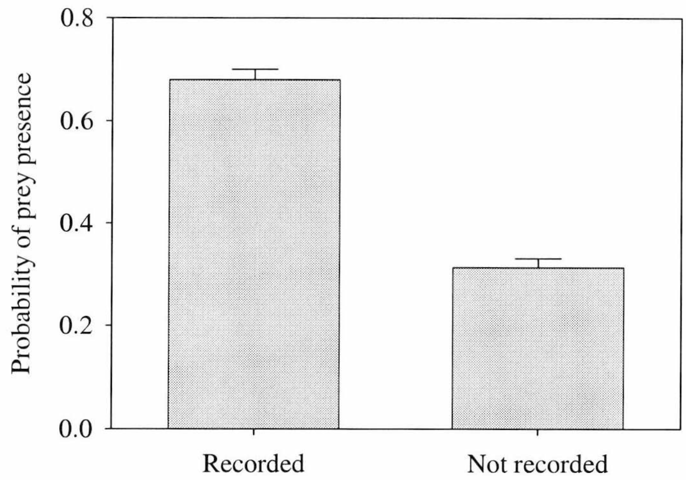


Figure 8.12: Prey preference for forested habitat in the KS region (with S.E. bars)

The final logistic regression model gave an overall estimate for the proportion of sites occupied by all key tiger prey as 0.791 (± 0.088). This represented 79% or 16722.6 km^2 of the 21141.1 km^2 forest in the KS region. Selecting 16722.6 km^2 of the most suitable habitat in the KS region identified seven large patches. However, considering only those areas ≥ 100 km^2, inside or adjoining KSNP, then a total forest area of 13593.7 km^2 for the KS region was obtained from six isolated forest patches (Figure 8.13).

8.3.4 Core tiger prey habitat and prey populations

In the KS region, six core tiger prey areas were identified that had good tiger prey habitat. Based on habitat integrity these areas were assumed to represent six potential tiger prey populations (Figure 8.13).

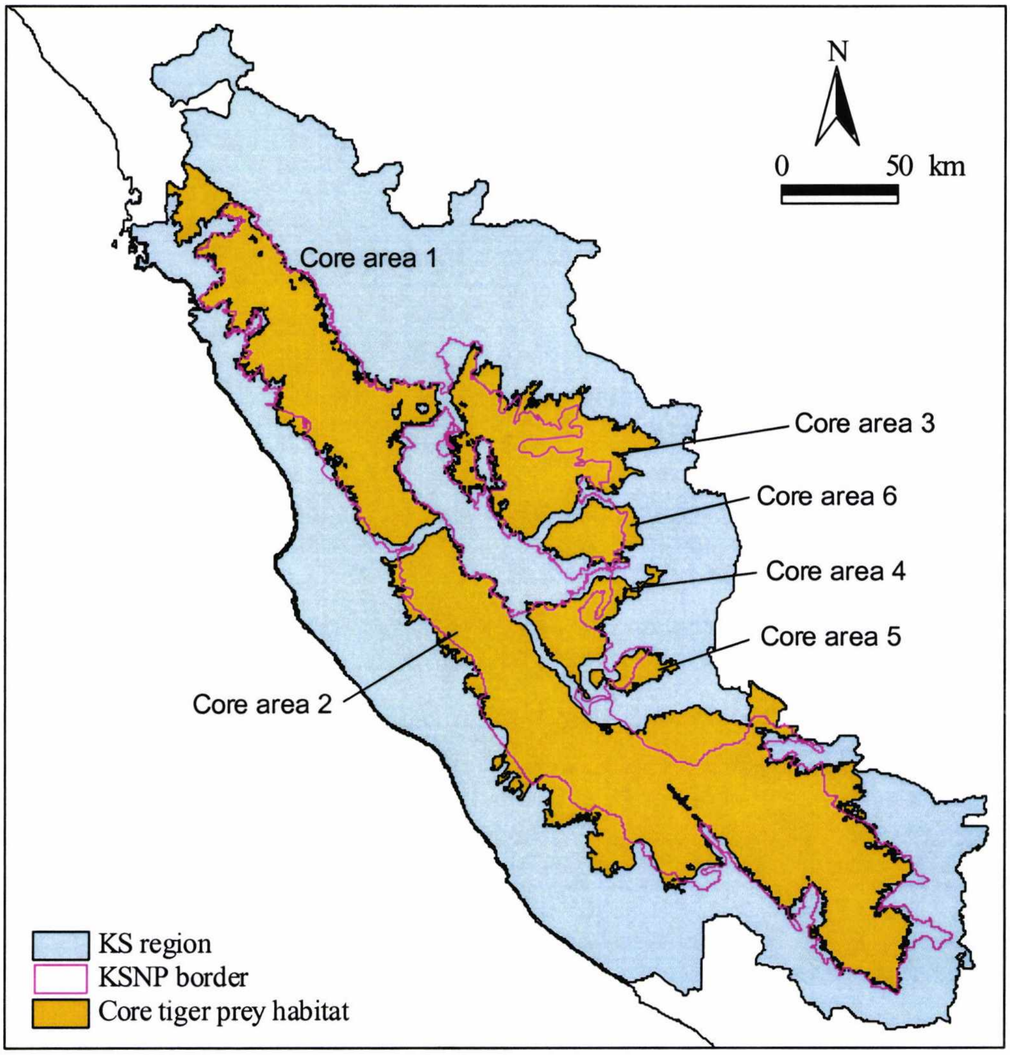


Figure 8.13: Core tiger prey habitat representing six potential tiger prey populations in the KS region

From the six core areas, three emerged as being substantially larger than the others: core 2 represented the largest core habitat (7234.1 km^2), then core 1 (3232.1 km^2), and then core 3 (1941.2 km^2). Core 2 was mainly composed of the better quality lowland and hill forest habitat types that are capable of supporting higher natural prey densities (Table 8.7, Figure 8.14 and 8.15). Core 1, although large, was the poorer quality forest habitat types of submontane and montane. Core 3 was smaller than core 2 and also contained poorer quality tiger prey habitat. Cores 4 and 6 were of a similar size and forest type composition, predominately hill and submontane. Core 5 was the smallest and contained submontane and montane, giving it a low potential as a core tiger prey habitat.

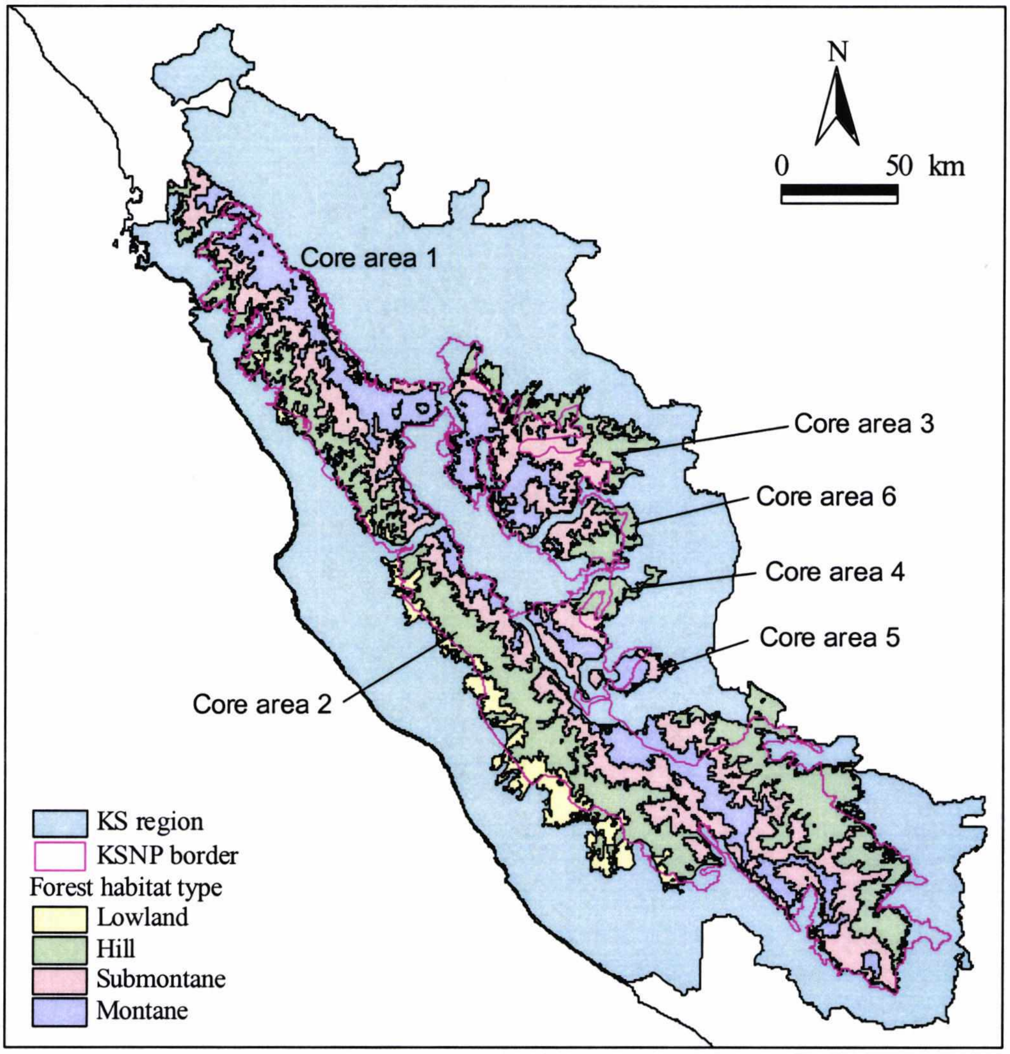


Figure 8.14: Distribution of forest types in core tiger prey habitats in the KS region

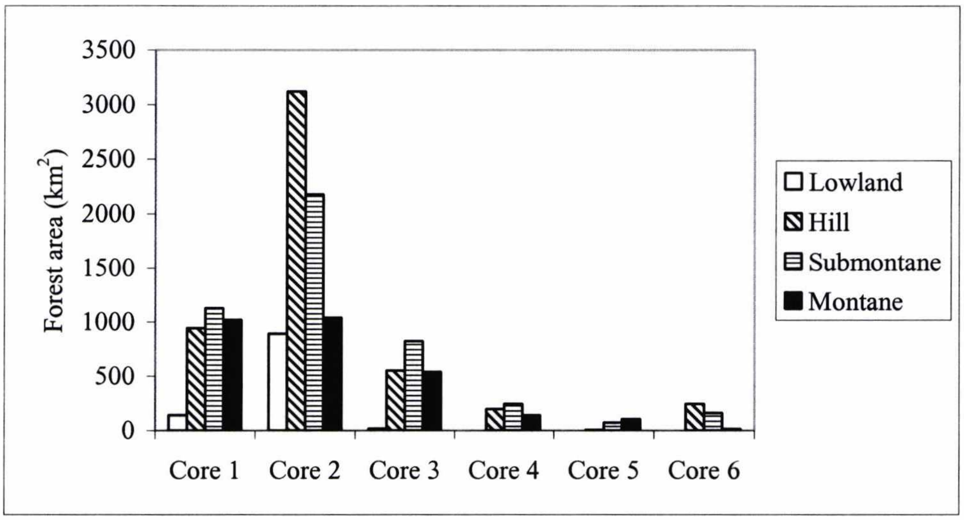


Figure 8.15: Area of forest types in core tiger prey habitats in the KS region

The mean distance from the forest edge to the perimeter of each core area varied between 0.14 km (core 1) and 1.62 km (core 4) due different core proximities to logging roads and public roads and no relation to core size (Table 8.7).

Table 8.7: Tiger prey core habitat characteristics

Core ID	Average distance from core edge to forest edge (km)	Forest (km^2)				
		Total area	Lowland	Hill	Submontane	Montane
1	0.14	3232.1	140.8	944.4	1126.9	1020
2	0.54	7234.1	894	3119.9	2179.4	1040.8
3	0.28	1941.2	17.5	555.8	824.2	543.6
4	1.62	582.3	0	196.2	245.2	140.9
5	0.32	182.6	0	5.2	72.4	105
6	0.79	421.4	0	243.8	162.7	14.9
Total	Mean = 0.61	13593.7	1052.3	5065.3	4610.8	2865.2

8.3.5 Factors determining distribution of snare traps

The location of snare traps set for tiger prey was related to $\log_{10}$ distance to logging roads, to $\log_{10}$ distance to rivers and to transformed indicator of poverty (Table 8.8). Snare traps were more likely to be set in areas that were closer to logging roads, further away from rivers, and closer to less poor villages (Figure 8.16-8.18). The logistic regression model explained 78.1% of the original observations and was not affected by spatial autocorrelation (Moran's $I = 0.03$, $P > 0.1$). The final model had an AUC value of 0.886 indicating an accurate fit.

Table 8.8: Best multiple logistic regression model describing the relationship between landscape factors and tiger prey snare trap presence in the KS region

Factor	Coefficient (β) ± S.E.	df	Wald	*P*
Transformed poverty (satellite)	10.761 ± 5.304	1	4.116	0.042
Log_{10} distance to rivers	2.514 ± 1.157	1	4.721	0.030
Log_{10} distance to logging roads	-1.249 ± 0.437	1	8.175	0.004
Constant	-2.167 ± 2.980	1	0.529	0.467

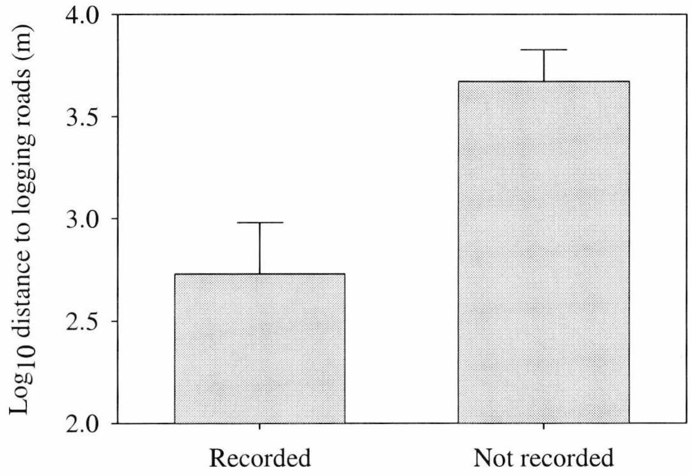


Figure 8.16: Presence of snare traps relative to mean $\log_{10}$ distance to the nearest logging road (S.E. bars)

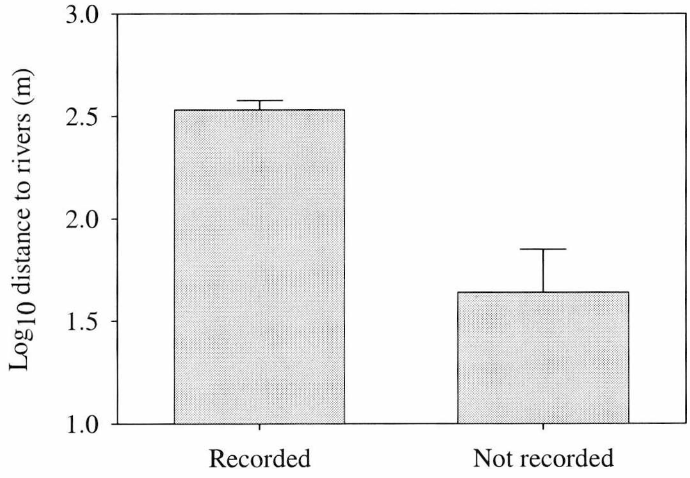


Figure 8.17: Presence of snare traps relative to mean $\log_{10}$ distance to the nearest river (S.E. bars)

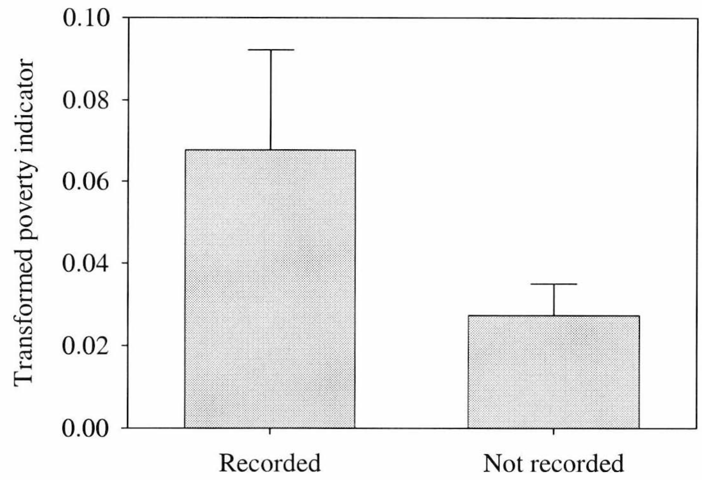


Figure 8.18: Presence of snare traps relative to mean number of arcsine transformed satellite dishes per village households (S.E. bars)

8.4 DISCUSSION

Common themes emerged between the factors that explained individual and collective prey base distribution and snare trap placements in KSNP. A closer proximity to public roads and logging roads was correlated with a lower likelihood of prey presence and a greater likelihood of snare trap presence. The prey habitat preference model highlighted the importance of KSNP as a refuge for tiger prey because better quality habitat for tiger prey occurred inside of KSNP. An asphalt road running through KSNP complicated the matter, which besides reducing habitat quality either side of it may act as a barrier between prey populations in the northern and southern sections of KSNP. A marked reduction in habitat quality, associated with closer proximity to roads, was on average 0.61 km from the forest edge. These edge effects illustrate the pervasive and less apparent affects of habitat loss and fragmentation.

In the KS region, roads created a hostile environment for tiger prey because they are commonly used by humans to access the forest. For the species of deer (mouse deer, muntjac and sambar) an inverse relationship with distance to public roads was recorded. These roads form a peripheral network to KSNP and may therefore be acting as a surrogate for distance to forest-farmland edge. An avoidance of the forest edges was illustrated by the habitat suitability model identifying interior forest as better quality habitat and by the tendency of these species not to crop raid (Chapter 7). This may also explain why sambar showed a preference for higher elevation, because it is located in areas that are further away from the forest edge. These findings concur with those from other studies that showed sambar and muntjac seeking refuge further into the forest away from roads and forest edges with high human activity (Griffith and van Schaik, 1993, Kawanishi 2002, O'Brien et al. unpublished data). In India, human disturbance was found to have significant negative effects on the distribution of chital and sambar (Jonhsingh 1983, Sankar 1994, Mathai 1999, Jathanna 2001). From elsewhere in Asia, sambar and common muntjac have exhibited higher abundances in logged and secondary forest than in primary forest, which typically have greater food availability for terrestrial herbivores. In contrast, yellow muntjac (*Muntiacus atherodes*) and mouse deer, which require closed intact forest, exhibited lower abundances (Duff et al. 1984, Heydon 1994, Heydon and Bulloh 1997, Giao et al. 1998).

Wild boar were found to be more abundant in areas further away from both public and logging roads. Typically described as a forest edge species, the response of wild boar to these edge effects was unusual. In southern Sumatra, wild boar were not encountered at forest edges with high human population densities. At the forest edges with low human population densities, the abundance of wild boar at the immediate edge was low and became highest 7 km into the forest interior (O'Brien et al. unpublished data). Wild boar are a principal agricultural crop pest around KSNP. The damage that these animals are capable of causing is substantial and the economic loss can be high for these low income farmers. This often results in farmers setting snare traps near the forest edge. Snare traps are also set to catch deer for their meat and logging roads provide greater access into the forest for hunters for this.

The poaching pressures on tiger prey base, as identified by the distribution of snare traps, may explain why logging roads had large and negative effects on their distribution in the KS region. Increased access to tropical forests, especially by logging and other roads, has been identified as the single greatest factor correlating with the demise of wildlife populations (Robinson et al. 1999). The construction of roads provides hunters with access to relatively unexploited populations of forest wildlife (Wilkie et al. 2000, Peres and Lake 2003). In Sarawak, ease of access was directly and inversely correlated with the densities of large forest ungulates. In North Sulawesi, the expansion of a highway was correlated with the loss of certain species and diminished populations of others (Clayton and Milner-Gulland 2000; Lee 2000).

Snare trap placement in the KS region was also associated with proximity to rivers, probably because placements tend to be on ridge trails that are far from rivers and where animals such as muntjac and sambar are more likely to be encountered. In contrast to another study, rivers act as access points into the forest and have been shown to reduce large mammal densities in close proximity (Peres and Lake 2003). Whilst the relationship between physical factors and hunting pressures are well documented the relationship with socio-economic factors, in particular poverty, are not (Demmer et al. 2002). The distribution of snare traps in the KS region was associated with richer villages. This is contrary to research from the Philippines that found poor households were more likely to hunt wildlife as a food supplement because they could not produce enough food from farming, which was considered a superior food source (Shively 1997). Attitude studies from communities living near PAs in Africa have found a positive correlation between affluence and conservation attitudes (Infield 1988, Newmark and Leonard 1991). This does not mean that improving the living standards of local communities will increase the viability of wildlife populations within the PAs (Ferraro and Kramer 1997, Newmark and Hough 2000). This makes the findings in this study of particular interest because it is contrary to the KS-ICDP rationale of village development being linked with better protection of biodiversity. The sensitivity of tiger populations to poaching and habitat loss is another concern (Damania et al. 2003). The factors influencing tiger prey distribution have been determined, so it is now important to determine how the same array of factors interact to determine tiger distribution and tiger population viability.

Chapter 9

TIGER RESILIENCE IN A FRAGMENTED LANDSCAPE

Sumatran tiger active during the daytime deep inside KSNP (J. Holden)

9.1 INTRODUCTION

A fundamental requirement for effective management of threatened species is to understand their response to disturbance in human-dominated landscapes. One of the most serious and pervasive threats occurring in these landscapes is that posed by habitat loss and fragmentation (Mace & Balmford 2000, IUCN 2002). These threats bring wildlife and humans into closer contact and leads to competition for space and resources, frequently ending in conflict (Sitati et al. 2003). Large carnivores are particularly susceptible to these threats because they occur at naturally low densities and require large areas that often overlap with those occupied by humans (Woodroffe & Ginsberg 1998, Revilla et al. 2001).

These problems are well illustrated by tigers (*Panthera tigris*), where two Indonesian subspecies are extinct because of widespread forest clearance for agriculture and over-hunting of tiger prey by people (Seidensticker et al. 1987). Whilst it is unlikely that healthy tiger populations can coexist in areas where humans exert strong pressures on natural resources, tigers are be able to tolerate modest levels of habitat disturbance (Karanth & Madhusudan 1997, Karanth & Nichols 2000). This disturbance is most prevalent at the forest edge where it causes a reduction in habitat quality that adversely affects tiger distribution (O'Brien et al. 2003, Griffith & van Schaik 1993). This response may be related to higher poaching pressures on both tiger and their ungulate prey nearer to the forest edge. Poaching is often related to landscape factors such as roads that increase accessibility to the forest (Bennett & Robinson 2000, Linkie 2003). Thus, the position of roads or villages can increase edge effects and can fragment tiger populations and further threaten their viability. This is of particular concern for tiger habitat in Sumatra, Indonesia, because edge effects are expected to increase as illegal logging continues to degrade and fragment forest both inside and outside of PAs (Kinnaird et al. 2003). In order to effectively conserve tigers in Sumatra, it is necessary to determine their current distribution, as well as the location and spatial integrity of suitable habitat (Rushton et al. 2004).

Previous studies on large carnivores such as lynxes and grey wolves used logistic regression modelling to investigate the factors that determine their presence or absence, and used these data within a GIS to construct spatially explicit habitat models (SEHMs) (Mladenoff et al. 1999, Palma et al. 1999, Schadt et al. 2002). In order to reduce the problems associated with 'false absences' a function of detection probability should be included in the model construction (Mackenzie et al. 2002, 2003). From the resulting SEHMs, an important next step for conserving large carnivores is to calculate their population sizes and combine these with spatially explicit population simulation models to determine their viability (Carroll et al. 2003). For conservation planning, the strengths of population viability analyses (PVAs) lie in comparing different management strategies and scenarios (Coulson et al. 2001).

Kerinci Seblat National Park (KSNP) represents one of the largest PAs inhabited by tigers, spanning four provinces in the 40,000 km^2 KS region. Within this area, forest loss has fragmented KSNP in two parts and poaching has severely depleted tiger prey and indeed sections of KSNP to poor quality tiger habitat. To effectively manage tigers in this region, information is needed on the spatial integrity of suitable habitat, the location of tigers, the number and size of tiger subpopulations and the viability of these subpopulations.

9.1.1 Aims and objectives

This chapter aims to:

- determine the human-related threat, physical and prey factors that explain tiger abundance at the forest edge;
- determine the physical factors that explain tiger distribution in the KS region, through construction of a tiger SEHM;
- identify the location of core tiger habitat and tiger subpopulations in the KS region;
- calculate tiger density for lowland, hill, submontane, and montane forest;
- calculate the total number of adult tigers that could be supported by each core habitat area; and,
- investigate the viability and resilience of each of the different tiger populations under different management scenarios.

9.2 METHODS

9.2.1 Field methods

A 2 km^2 sampling grid was constructed for the KS region. The presence, or detection, of tigers in the KS region were obtained from repeated transect surveys in 200 grid cells between 2001 and 2002 (as Section 8.2.1). A further 100 grid cells were surveyed to allow verification of the SEHM once constructed. The presence or absence of the following variables were recorded on each transect: tigers; their prey species; human-related threats; and, human disturbance (as Section 8.2.1).

9.2.2 GIS methods

The 2 km^2 sampling grid constructed for the KS region see Section 8.2.2 was used for two analyses. For each 2 km^2 cell, data were collated on tiger presence, prey species presence, landscape characteristics and human-related threats. To this were added spatial data for roads, logging roads, settlements, rivers, elevation, slope, PA status and soil type, based on the datasets constructed in Chapter 4 (Table 9.1). The first analysis determined the spatial factors that best explained tiger distribution. The second analysis determined the spatial factors with complete coverage that best explained tiger distribution so that a tiger SEHM could be developed for the entire KS region.

Table 9.1: Spatial information compiled for each 2 km^2 sampling cell

Factor	Coverage
Proximity to nearest public road	Complete
Proximity to nearest logging road	Complete
Proximity to nearest settlement	Complete
Proximity to nearest river	Complete
Elevation	Complete
Slope	Complete
PA status	Complete
Soil type	Complete
Prey snare trap recorded	Partial
Human disturbance	Partial
Prey base recorded	Partial

The continuous data were logarithmically transformed and along with the categorical data were extracted for each cell and imported into PRESENCE software package. Tiger detection (1) and non-detection (0) data were then entered to provide information on the detection history of each sampling cell for each sampling occasion.

9.2.3 Statistical methods

The effect of each variable on the probability of tiger presence was first tested by a univariate analysis to determine if any of the factors were interdependent. To estimate the proportion of area occupied by tigers in the KS region a general likelihood model was constructed that incorporated potential landscape covariates influencing the detection probability of tigers. A multiple logistic regression analysis was performed to determine which of the landscape variables best explained the detection of tigers in the KS region. PRESENCE was used to generate a combination of models that were ranked by their Akaike Information Criterion (AIC) values to determine the most parsimonious model that best fitted the data (Burnham & Anderson 1998). From this, tiger detection and area of occupancy was estimated for the KS region. A simulation study was then conducted to investigate the performance of the model. This generated data from the model such that all parameters (number of sampling sites, sampling occasions and detection probabilities) were constant with respect to time and provided an estimate of sampling bias for the logistic regression analysis.

In the spatial analysis it was necessary to test for non-independence caused by spatial auto-correlation because landscape features close to each other tend to have similar characteristics (Koenig 1999). The presence of spatial auto-correlation in the model was tested by calculating Moran's *I* statistic (Cliff and Ord 1981) using the Crime-Stat v1.1 software package.

The statistical analysis was then repeated for only the factors with complete coverage in the KS region so that the factors that best explained tiger distribution could be used to construct a tiger SEHM. From the final logistic regression model the probability of tiger presence (*P*) was determined by:

$$Y = \beta_0 + \sum \beta_i X_i,$$

Where: β_0 is the constant coefficient; β_i represents the significant independent variable coefficients; and, X_1 represents their associated independent variables. Through incorporating the natural exponential (*e*) into the previous equation, the tiger SEHM for the KS region was constructed by:

$$P = e^{Y} / 1 + e^{Y}$$

The accuracy of the model predictions was independently validated using tiger detection and tiger non-detection data collected from a single survey in the 100 additional monitoring sites, which were not included in the regression analysis. To minimize potential problems with 'false absences', the sampling effort in these monitoring sites was increased. From these data, 30 cells recorded tiger presence. For parity, another 30 cells with no tiger record were then randomly selected from the remaining 70 sites. The SEHM probability value for these sets of cells was extracted. A Mann-Whitney U test was then used to find whether those sites where tigers were detected had a higher predicted probability than those sites where tigers were not detected.

Next, the tiger SEHM was reclassified by selecting the most suitable habitat available that corresponded to the estimated proportion of area occupied. This enabled the location of core tiger habitat containing different subpopulations of tigers to be identified. These core areas were then treated independently. For each area the mean distance from the core perimeter to the forest edge was calculated. The amount of lowland, hill, submontane and montane forest was then calculated for each area so that tiger carrying capacity for each of the core areas could be estimated.

Tiger density for each of the forest habitat types was estimated using camera trap data collected in KSNP (Holden 1997). Tiger encounter rates (number of tiger passes/100 days) calculated for each habitat type (Linkie et al. 2003) were converted into a tiger density (adult tigers/100 km^2). This used an equation based on the relationship between relative abundance (Carbone et al. 2001). This method, whilst not as statistically robust as that developed by Karanth & Nichols (1998), does allow a rapid assessment of tiger populations. Unlike capture-recapture studies, a caveat attached to the Carbone et al. (2001) method was that it lacked validation against an independent measure of density (Jenelle et al. 2002). O'Brien et al. (2003) have subsequently helped to show that the relative abundance of tigers is directly related to independently derived estimates of densities. Using the tiger density estimates for lowland, hill, submontane and montane forest the total number of adult tigers that could be supported in each core area was calculated.

9.2.4 PVA method

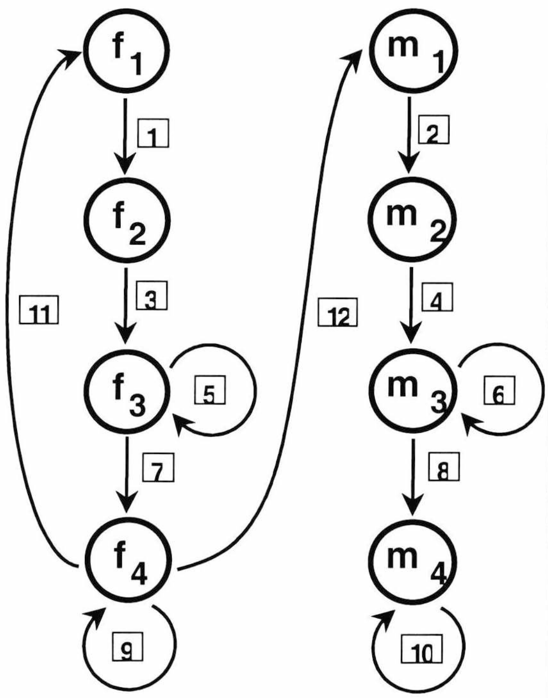
The viability and resilience of the tiger populations in the core areas was studied using the computer program Unified Life Models or ULM (Legendre and Clobert 1995, Ferrière et al. 1996). This software uses a time-discrete stage-structured population model and has been used to model the population dynamics of brown bears *Ursus arctos* (Wielgus et al. 2001), arctic foxes *Alopex lagopus* (Loison et al. 2001), Iberian lynxes *Lynx pardinus* (Bessa-Gomes et al. 2002) and snow leopard *Uncia uncia* (Chapron and Legendre 2002).

9.2.4.1 Life cycle modelling

A variant of the stochastic model proposed by Karanth and Stith (1999) was used in this study. The population was divided into several stages defined by sex, age and breeding status. Tigers were classified as cubs (0-12 months), juveniles (12-24 months), floaters (> 24 months) or territorial breeders (> 36 months) for both sexes (Table 9.2). Transitions between classes are illustrated in Figure 9.1.

Table 9.2: Model classes used in the tiger population life cycle

Age	Females	Males	Class
0-12 months	F1	M1	Cubs
12-24 months	F2	M2	Juveniles
> 24 months	F3	M3	Transients
> 36 months	F4	M4	Breeders



1.Surviving female (f) cubs become juveniles
2.Surviving male (m) cubs become juveniles
3.Surviving f juveniles become transients
4.Surviving m juveniles become transients
5.Surviving f transients that did not find a territory stay transients
6.Surviving m transients that did not find a territory stay transients
7.Surviving f transients that found a territory become breeding f
8.Surviving m transients that found a territory become breeding m
9.Surviving breeding f keep the same status
10.Surviving breeding m keep the same status
11.Surviving breeding f give birth to f cubs
12.Surviving breeding f give birth to m cubs

Figure 9.1: Post-breeding life cycle (each arrow lasts 1 year)

The typical female/male ratio is 3/1 among breeders (Sunquist 1981), the breeding female carrying capacity (K) is thus defined as being equal to $K/3$. The fact that male numbers could fluctuate due to intraspecific competition is ignored because the number of females in a population is usually the limiting resource (Kenney et al. 1995). Floaters are able to find a territory and settle if it is not already occupied. The spatial organization of territories is not modelled based on the assumption that all females are able to reproduce as long as there remains one male in the population. The demographic parameters used for the model compare: an equal sex ratio at birth; a mean litter size of 3 cubs; an interbirth interval of 2.5 years; and, an age of first reproduction of 3 years. The time step in the model is 1 year. For each of the core tiger populations, a ceiling was incorporated into the model so that the population could not exceed this value. The ceiling value for each core area was calculated as in section 9.2.3, but by determining the carrying capacity for the core area plus adjoining forest, regardless of its tiger habitat quality score.

9.2.4.2 Parameters

This study used the same values used by Karanth and Stith (1999). The numerical values of these parameters were based on field data collected for: tigers (Sunquist 1981, Smith et al. 1987, Smith and McDougal 1991, Smith 1993); leopards (Martin and de Meulenaer 1988, Bailey 1993); and, cougars (Lindzey et al. 1992, Laing and Lindzey 1993, Lindzey et al. 1994), see Table 9.3. The model used in this study used demographic stochasticity. Demographic stochasticity is the random realization of probabilistic events (binomial law of a constant mean, the survival). This means that the value of a parameter such as cub survival for example can be fixed at 0.6 so that the probability of every individual cub encountering the event "I survive" is 0.6. For large numbers of individuals (n = 1000) it is possible to check that the mean observed survival is 0.6. For smaller populations, such as 3 tigers there can be 0, 1, 2 or 3 surviving tigers, but this does not mean that survival was not 0.6. Demographic stochasticity was incorporated into the PVA model through the application of Bernoulli trials. For example, a cub survival value of 0.6 was the sum derived from these trials with a mean of 0.6. Trials gave a 0 or 1 to the value, which did not, theoretically, deviate around the mean, but was a small sample size which gave a biased mean. Because this bias was random, Monte Carlo simulations were run to incorporate this randomness. The shape of the curve was thus defined by the parameter value of 0.6.

Table 9.3: Model parameters

	Parameter	
s_1	Cub survival	0.6
s_2	Juvenile survival	0.9
s_{f3}	Transient female survival	0.7
s_{m3}	Transient male survival	0.65
s_{f4}	Breeding female survival	0.9
s_{m4}	Breeding male survival	0.8
f	Litter size (at birth)	1.2

Mortality sources compared: baseline mortality; poaching; and, prey depletion. Baseline mortality accounted for tiger death resulting from intraspecific and interspecific competition, natural disease, starvation and dispersal into unsuitable habitat, as background mortality that occurs among all normal healthy tiger populations (Kotwal and Gopal 1994, Smith 1993, Karanth unpubl. data). Migration or movement between core tiger areas was set at 25%.

Additional increments to natural mortality arise in tiger populations subject to poaching through shooting, poisoning, trapping, snaring and electrocution. In the models, poaching does not compensate mortality due to intraspecific competition and is modelled as being completely additive to natural mortality (Karanth and Stith 1999). Poaching affects all adult sized tigers, and it was therefore considered that poaching would occur in all classes except cubs. The survival of these classes was subsequently lowered through multiplying each class by the probability of it not being poached, 1-*po* (where *po* is the poaching pressure). Anti-poaching measures were incorporated into the model, by specifying that if a population was receiving protection then no tigers were poached.

9.2.4.3 PVA simulations

The tiger subpopulations identified from the core areas in Section 9.2.3 were used to run the following models, in order to:

- determine the viability of each tiger subpopulation;
- determine the viability of each tiger subpopulation with low poaching (1 tiger removed per year);
- determine the viability of each tiger subpopulation with moderate poaching (3 tigers removed per year); and,
- determine the viability of each tiger subpopulation with high poaching (5 tigers removed per year);

The above models were then repeated with different combinations of connectivity between the core areas. Anti-poaching patrols were then incorporated into the models. Using different combinations of connectivity with different levels of poaching, each of the core areas was separately designated as having no poaching, i.e. being the focus of successful anti-poaching measures. Monte Carlo simulations were run with 500 repetitions for a duration of 50 years. A subpopulation qualifies as extinct once all classes are empty.

9.3 RESULTS

9.3.1 Physical and human-related threat factors determining distribution of tigers

From the 200 sampling cells surveyed, tigers were detected at 60 locations. This represented 18 sites from the first occasion, 27 sites from the second occasion and 15 sites from both the first and second occasions. Tiger presence across the KS region was related to presence of prey and to $\log_{10}$ distance to public roads (Table 9.4). Tigers were more likely to occur in areas that contained prey species and that were further away from public roads (Figures 9.2 and 9.3). From this, the best logistic regression model (AIC = 329.69) had a mean tiger detection probability of 0.322 (± 0.012) for the first survey and 0.135 (± 0.002) for the second survey and was not affected by spatial autocorrelation (Moran's I = 0.05, P > 0.1).

Table 9.4: Best multiple logistic regression model describing the relationship between the physical and human-related threat factors and tiger detection in the KS region

Factor	Coefficient (β) ± S.E.	df	*P*
Log_{10} distance to public roads	4.537 ± 1.092	1	< 0.05
Prey presence	2.400 ± 0.686	1	< 0.05
Intercept	-19.204 ± 4.175	1	< 0.05

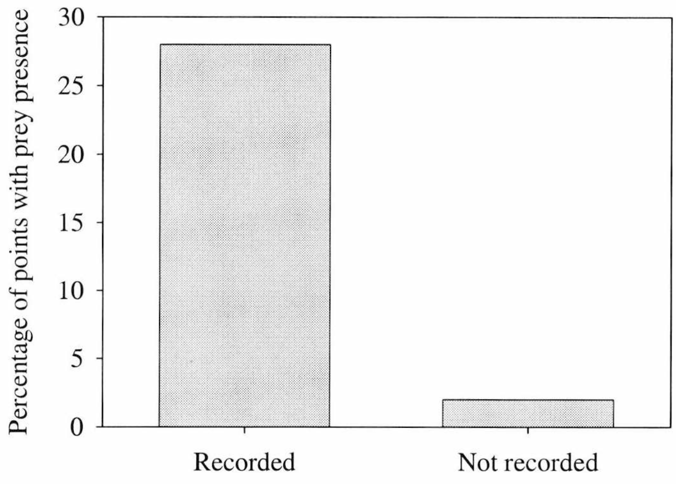


Figure 9.2: Presence of tigers relative to percentage of points with presence of prey

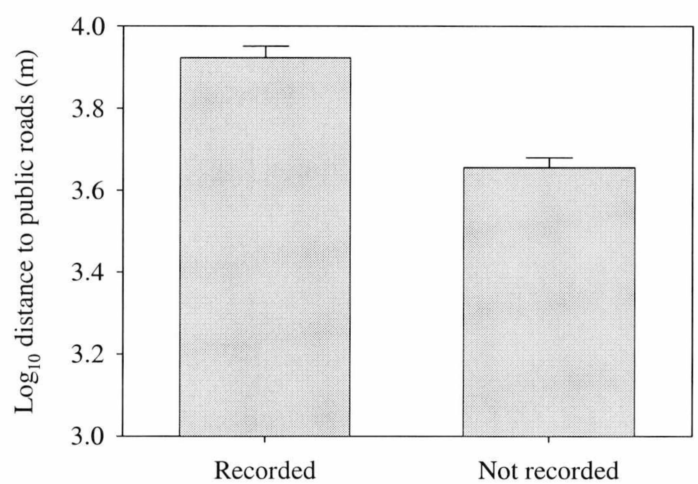


Figure 9.3: Presence of tigers relative to mean log_{10} distance to the nearest public road (with S.E. bars)

9.3.2 Physical factors determining distribution of tigers

Tiger presence across the KS region was related to log_{10} distance to public roads (Table 9.5). Tigers were more likely to occur in areas that were further away from public roads (Figures 9.3). From this, the best logistic regression model (AIC = 342.96) had a mean tiger detection probability was 0.359 (± 0.003) and was not affected by spatial autocorrelation (Moran's I = 0.04, P > 0.1).

Table 9.5: Best multiple logistic regression model describing the relationship between physical factors and tiger detection in the KS region

Factor	Coefficient (β) ± S.E.	df	*P*
Log_{10} distance to public roads	5.965 ± 0.678	1	< 0.05
Intercept	-22.725 ± 2.522	1	< 0.05

9.3.3 Tiger habitat preference model

The tiger SEHM was constructed from a single factor, $\log_{10}$ distance to public roads, and indicates that good quality tiger habitat occurred in three main areas in the KS region. These areas were all predominantly located inside KSNP (Figure 9.4). Although large blocks of forest habitat did occur outside of KSNP, such as in the north, the position of this forest in relation to public roads reduced it to poor quality tiger habitat.

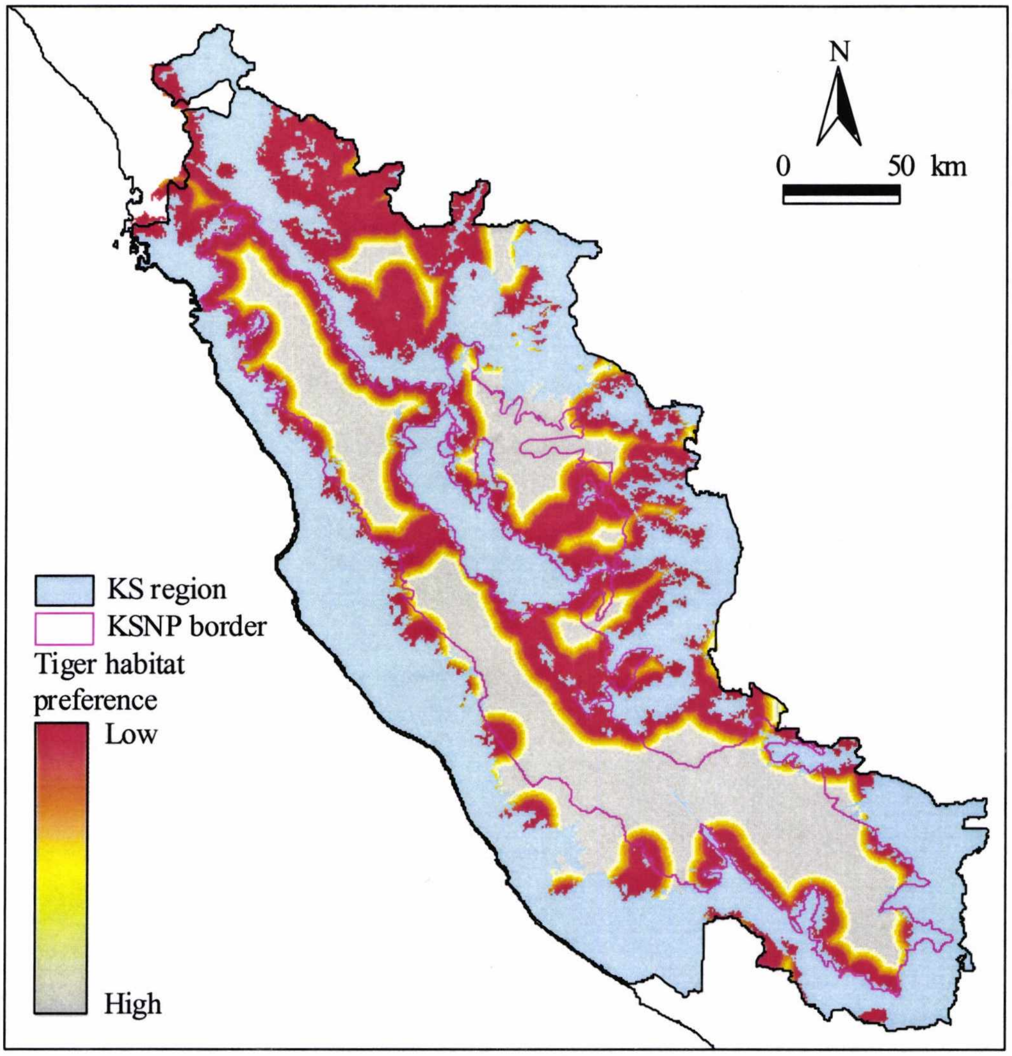


Figure 9.4: Tiger habitat preference in the KS region

The 30 selected monitoring sites with tiger detection had a mean predicted detection value of 0.716 based on the SEHM (Figure 9.5).. In contrast, the 30 selected sites with no tiger detection had a much lower mean predicted detection value of 0.200 (n = 60, Mann-Whitney U = 101.0, Z = -5.160, $P < 0.0001$

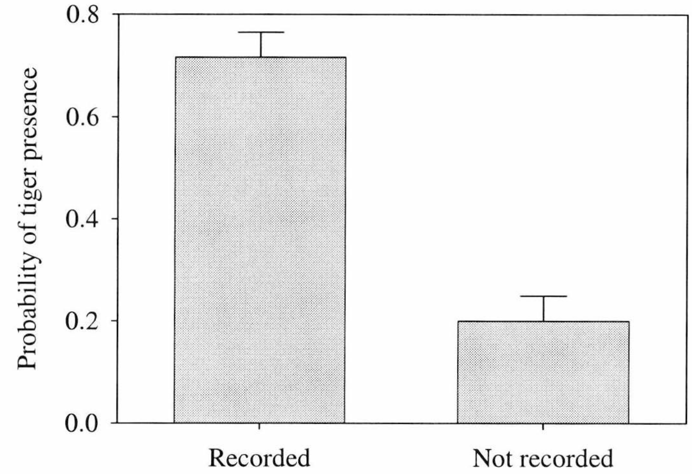


Figure 9.5: Tiger preference for forested habitat in the KS region (with S.E. bars)

9.3.4 Core tiger habitat and tiger populations

The final logistic regression model gave an overall estimate for the proportion of sites occupied by tigers as 0.458 (± 0.069). This represented 45.8% or 9691.1 km^2 of the 21141.1 km^2 forest in the KS region. Selecting 9691.1 km^2 of the most suitable habitat in the KS region identified seven large patches. However, considering those areas ≥ 500 km^2, revealed three isolated patches (Figures 9.6). The three core tiger areas identified as having suitable tiger habitat were assumed to represent three tiger subpopulations based on the degree of habitat integrity (Figure 9.6). The size of each core area differed, as did the mean distance from the perimeter of each core area to the forest edge (Table 9.6).

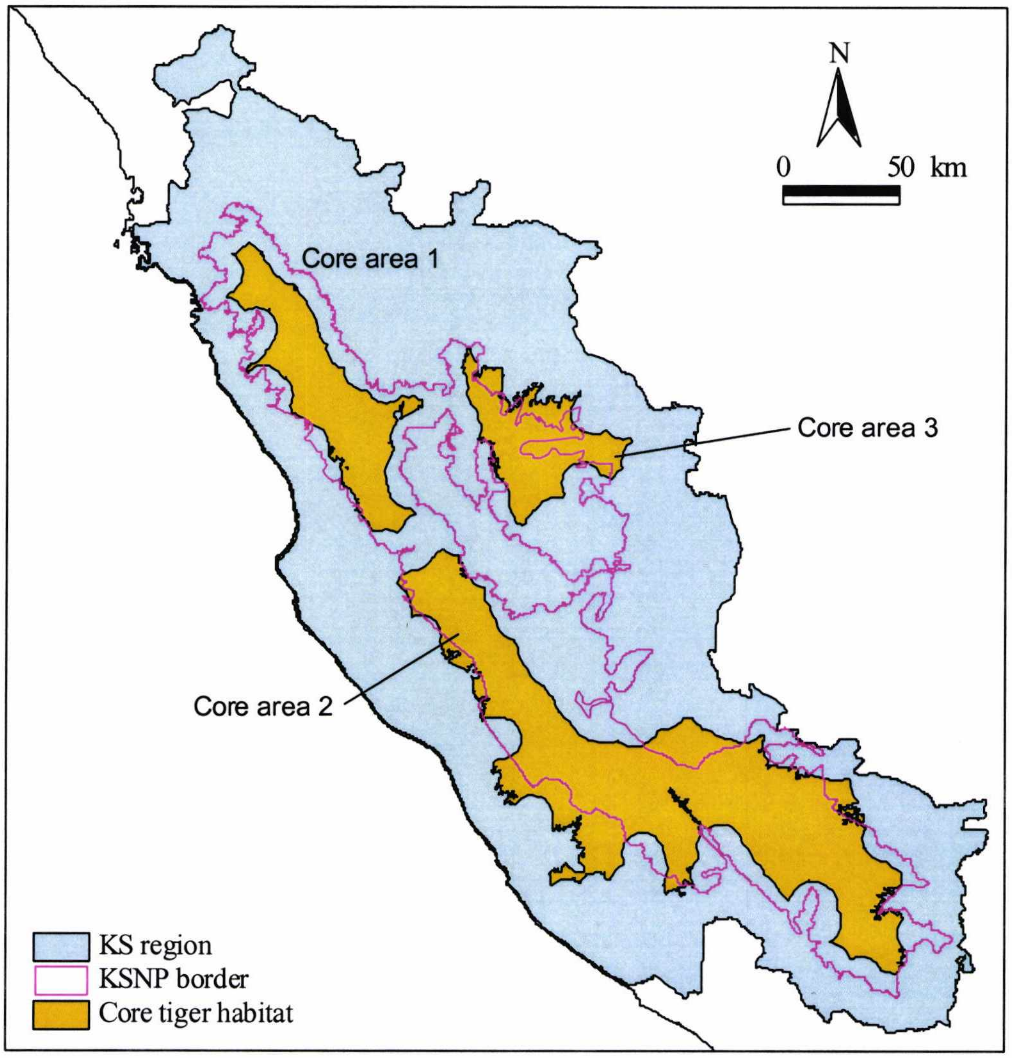


Figure 9.6: Core tiger habitat representing three tiger subpopulations in the KS region

Table 9.6: Core tiger habitat area and distance to forest edge

Core area	Size (km^2)	Distance to forest edge (km)
1	1667.0	1.98
2	5689.2	1.54
3	1219.1	1.41

9.3.5 Tiger density in different forest habitats

Tiger density recorded in the KS region was related to elevation and the associated habitat types (Table 9.7).

Table 9.7: Number of individuals, demography, encounter rates and density of tigers from photo-trapping across different tiger habitat types in the KS region.

Location	Total trap hours	Forest type	Elevation (m)	Individual tigers (adult males / adult females / cubs)	Encounter rate (days/tiger photo)	Density (tigers/100 km^2)
Tapan Valley	31000	Lowland	125-400	10 (3/4/3)	38	3.40
Tandai	50000	Degraded/ hill	500-900	2 (1/1/0)	74.4	1.77
Sipurak	28000	Submontane	600-1000	3 (2/1/0)	97.3	1.36
Mount Tujuh	23000	Montane	1800-2400	1 (1/0/0)	479	0.29

9.3.6 Tiger habitat and subpopulation abundance in core areas

Based on the distribution of different forest types in the three core areas, core 2 not only had the largest area but also mainly comprised better quality lowland and hill forest tiger habitat (Table 9.8, Figures 9.7 and 9.8). Core 1, although relatively large, was predominantly poorer quality submotane and montane forest tiger habitat. Core 3, the smallest core area also contained poorer quality tiger habitat. The different combinations of size and forest types meant that each core area had different carrying capacities for tigers (Table 9.8).

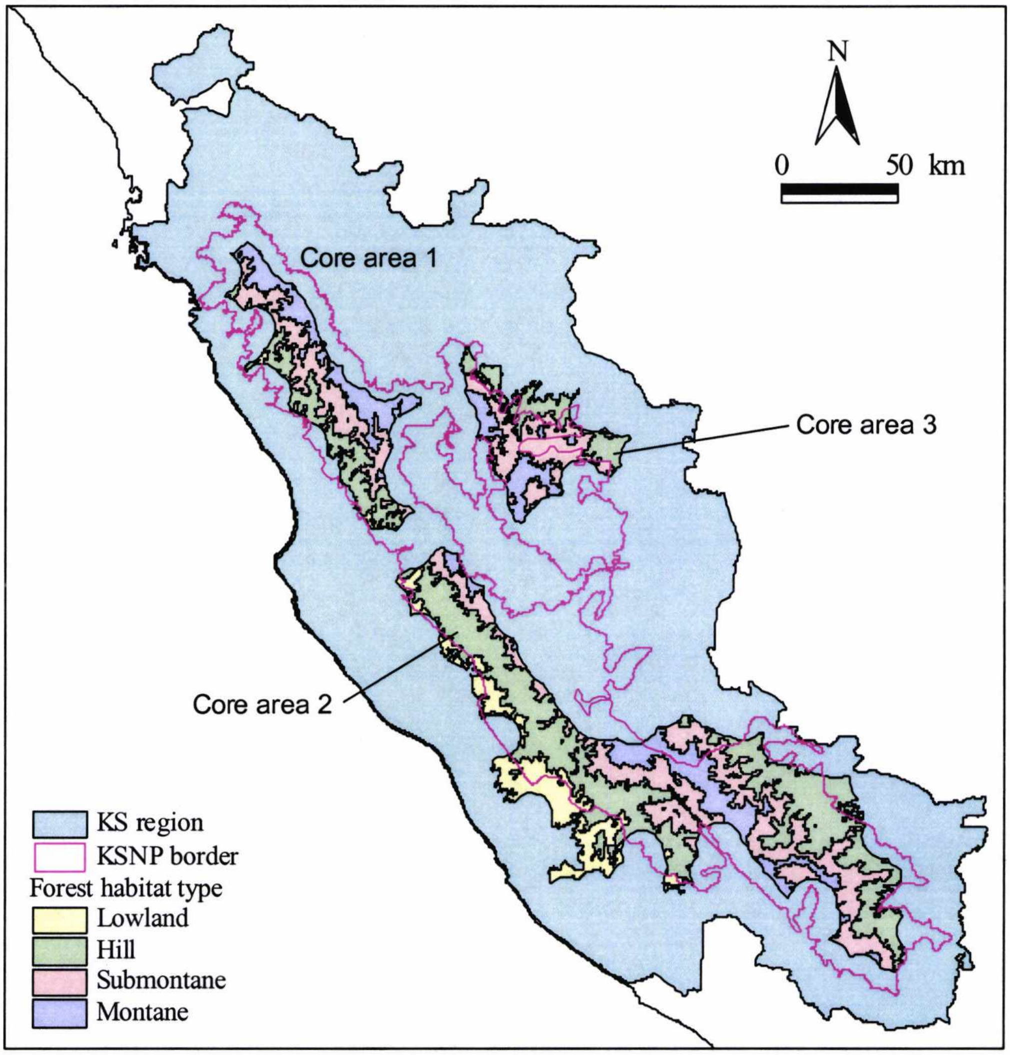


Figure 9.7: Distribution of forest types in core tiger habitats in the KS region

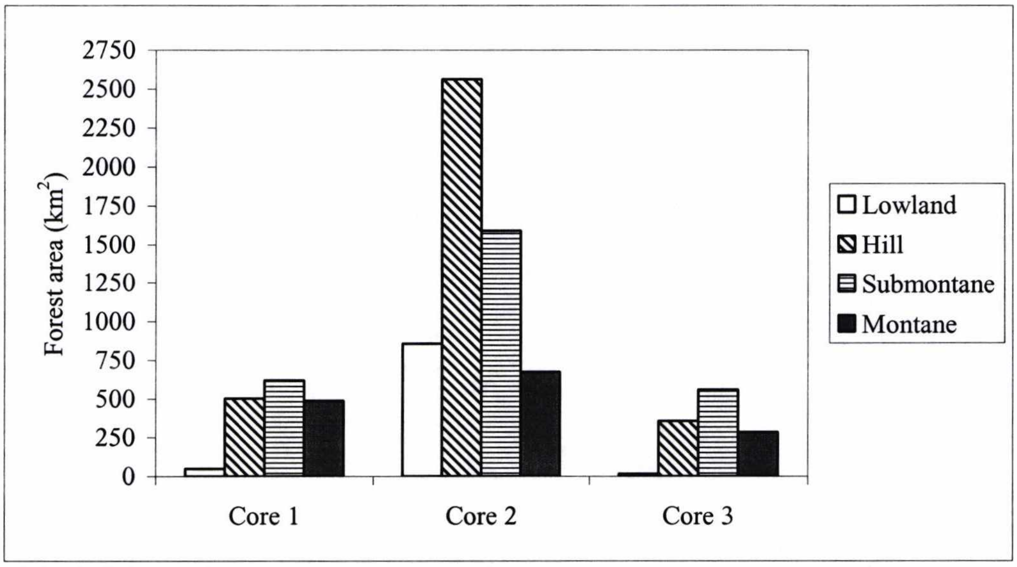


Figure 9.8: Area of forest types in core tiger habitats in the KS region

Core 2 could hold the largest population of 98 adult tigers, while core 1 could hold 20 adult tigers, and core 3 could hold 15 adult tigers. These population estimates were equivalent to an average adult tiger density of 1.20 tigers/100km^2 (core 1), 1.72 tigers/100km^2 (core 2), and 1.23 tigers/100km^2 (core 3). The total number of tigers estimated to be present over the three core areas was 133.

Table 9.8: Habitat characteristics and estimated numbers of tigers in each core area

	Core 1		Core 2		Core3	
Forest type	Area (km^2)	Estimated number of tigers	Area (km^2)	Estimated number of tigers	Area (km^2)	Estimated number of tigers
Lowland	49.3	1.7	856.5	29.1	17.1	0.6
Hill	505.1	8.9	2563.4	45.4	357.5	6.3
Submontane	622.1	8.5	1593.3	21.7	559.4	7.6
Montane	490.5	1.4	676.0	2.0	285.1	0.8
Total*	1667.0	20	5689.2	98	1219.1	15

*Number rounded down

9.3.7 Tiger population viability and resilience in the core areas

From estimating the tiger subpopulation sizes for all forest in and around each core area the ceiling adult tiger subpopulations used in the final PVA were set at 40 (core 1), 158 (core 2), and 37 (core 3). When each core tiger subpopulation was considered in isolation and with no poaching pressure, none was predicted to reach extinction within 50 years (Figure 9.9). However, when poaching levels were set at $\geq$ 3 tigers removed per year the subpopulations in core areas 1 and 3 faced almost certain extinction. In contrast, the subpopulation in core area 2 was large enough to withstand up to 5 tigers per year being poached.

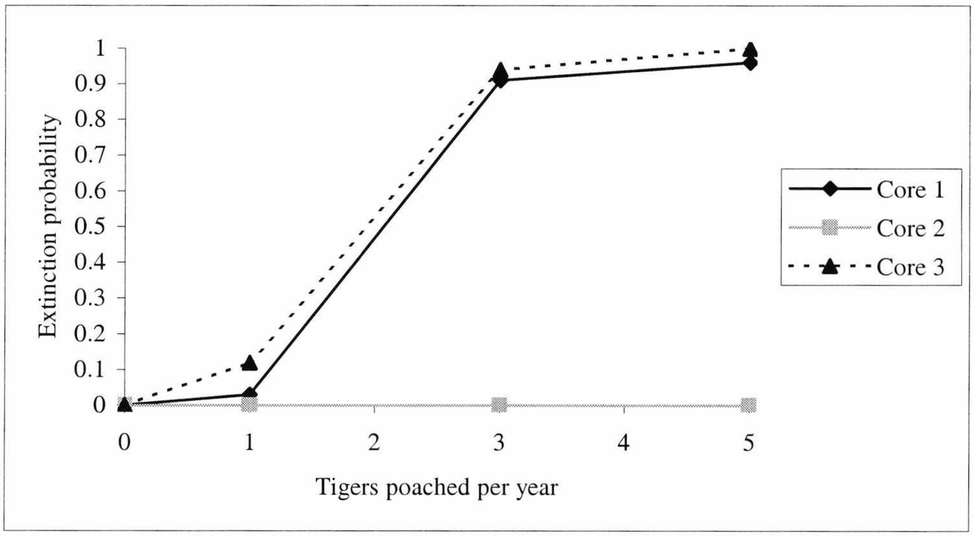


Figure 9.9: Extinction probabilities over 50 years for isolated tiger subpopulations with different poaching pressures

Connecting the core areas increased the likelihood of survival of each tiger subpopulation (Figure 9.10). Connecting core area 2 directly to core area 1, with a 25% dispersal rate, and indirectly to core area 3 [(1+3):(1+2)] greatly lowered the extinction probability of the tiger subpopulation. In core areas 1 and 3 with connectivity to core area 2, the tiger subpopulation in core area 1 was predicted not to reach extinction even if 3 tigers per year were poached. This illustrates the importance of core area 2 as a sufficiently large source population able to replenish depleted tiger populations in the two other core areas. In contrast, connectivity only between the smaller subpopulations in core areas 1 and 3 was not sufficient to ensure the survival of these populations if poaching levels remained high. With connectivity maintained between core areas 1 and 3, and with poaching levels increased to 5 tigers per year, core area 1 became more threatened, although still remained viable, whereas core area 3 was predicted to reach extinction.

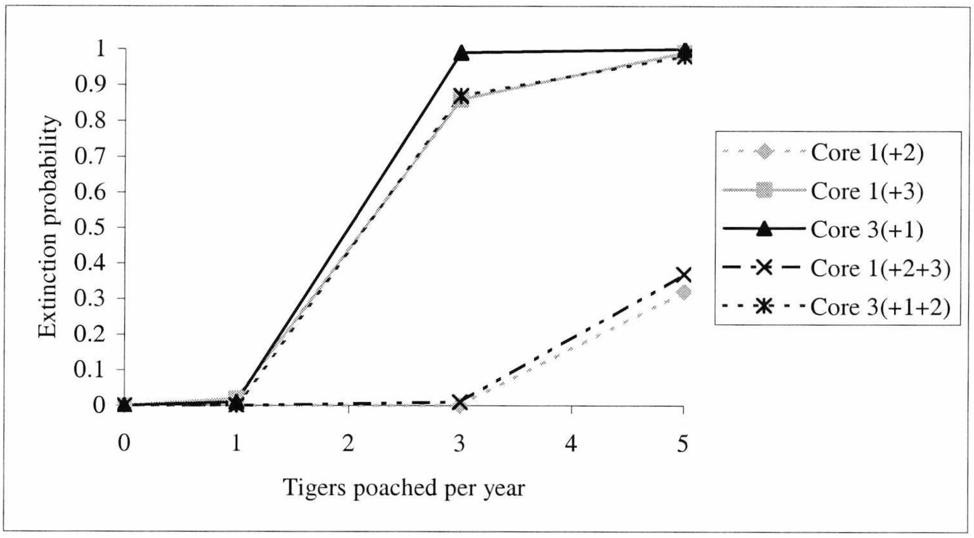


Figure 9.10: Extinction probabilities over 50 years for tiger subpopulations with different poaching pressures and connectivity

The introduction of anti-poaching strategies that targeted one core area predicted a marked decrease in extinction probability of tigers in the connecting core areas (Figure 9.11). If anti-poaching successfully eliminated the removal of tigers from core area 2, then the tiger subpopulation connected in core area 1 was predicted to be much less likely to reach extinction even if 5 tigers per year were poached ($P = 0.12$) than if there were no anti-poaching measures ($P = 0.32$). However, the most substantial change in tiger population viability was predicted to occur as a result of successful anti-poaching measures in core areas 1 and 3. If subjected to poaching levels of ≥ 3 tigers per year both these subpopulations were not viable. However, if core area 1 became the focus of anti-poaching measures, then it was predicted that the tiger population in core area 3 was no longer certain of extinction ($P = 0.53$) as previously predicted ($P = 0.99$). Equally, if core area 3 became the focus of anti-poaching measures, then the extinction probability in core area 1 ($P = 0.34$) was lower than if without these measures ($P = 0.86$). An improved situation was also predicted if core areas 1 and 3 were connected directly or indirectly to core area 2 (Figure 9.11). Extinction probabilities for core areas 1 and 3 were lowered considerably if a connecting core area 2 became the focus of anti-poaching patrols.

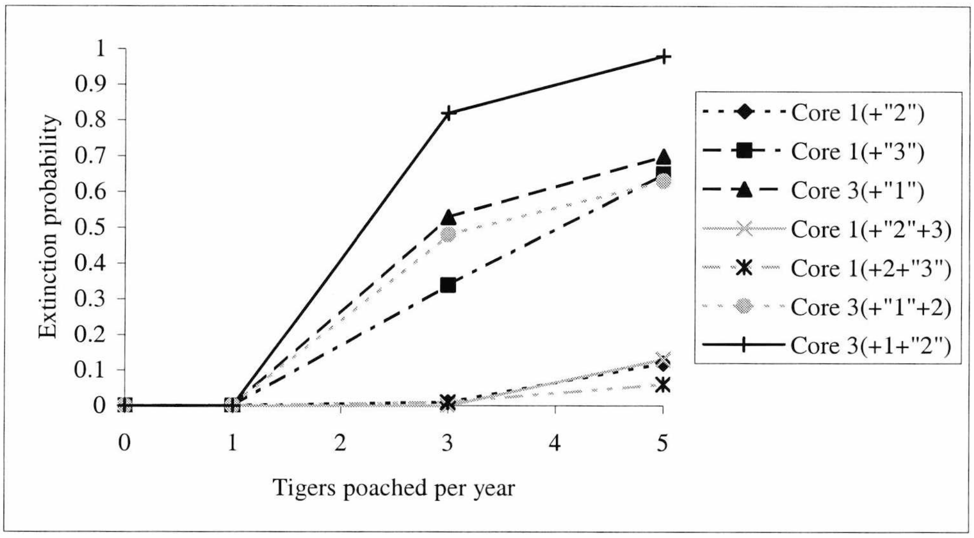


Figure 9.11: Extinction probabilities over 50 years for tiger subpopulations with different poaching pressures, connectivity (+), and anti-poaching measures focussed on specific core areas ("#")

9.4 DISCUSSION

This study is the first of its kind to develop a SEHM that incorporates an estimate of detection probability and proportion of area occupied. It is also the first to use a SEHM as the basis for assessing the role of different management strategies for protecting a large carnivore. In the KS region, the habitat model identified the negative affect of public roads on tiger distribution and habitat quality that resulted in the isolation of three core tiger areas. Linking the habitat model with the PVA showed the effectiveness of focussing anti-poaching strategies at the two smaller core areas and maintaining connectivity between the largest core area with the smaller adjacent core area to greatly improve the long term persistence of tigers in KSNP. Such an approach is important for the continued survival of large carnivores because it can be used to support the prioritisation of areas for conservation management and reintroduction.

Before drawing any conclusions from this study it necessary to discuss the caveats with the likelihood-based modelling framework used. Ideally, the sampling protocol follows a robust mark-recapture design and conducts five surveys or five sampling occasions (Pollock 1982, Pollock et al. 1990). A possible problem with the survey design in the KS region was that it included only two sampling occasions. MacKenzie et al. (2002, 2003) conducted simulation studies to test model performance under scenarios with a differing number of sampling occasions, sampling sites and detection probabilities. They found that increasing these factors improved the accuracy and precision of the predicted area of occupancy. From our study, we reduced the potential biases associated with a low number of sampling occasions by including a large number of sampling sites, which were found to have reasonable detection probabilities of 0.330-0.384 (MacKenzie et al. 2002). Furthermore, a simulation study conducted on these data from the KS region indicated that proportion of area occupied may be biased by approximately 0.025 which is relatively small in comparison to the magnitude of the standard errors. However recent research (MacKenzie, unpublished manuscript) would suggest more precise estimates of the proportion of area occupied by tigers could be achieved with same level of field effort by conducting more repeat surveys at fewer grid cells (e.g., 5 surveys are each of 80 grid cells).

The validity of this method was particularly important as it gave greater confidence in the PVA predictions. Difficulties associated with PVA modeling frequently arise because of a lack of requisite data. In order to minimize such difficulties, we evaluated relative rather than absolute extinction risk, with projections over short time period, and with stochasticity modeled by true probability sampling rather than truncating numbers (Burgman and Possingham 2000). The purpose of the PVA predictions was as part of a decision support tool, as opposed to a decision making tool, which has greater advocacy (Possingham et al. 1993, Starfield 1997).

Tigers in KSNP showed a preference for areas further away from public roads. This was probably due to three main reasons: tiger prey were avoiding these roads (section 8.3.3); human activity was higher nearer these roads; and hunting pressure on ungulates is increases nearer to roads (Auzel and Wilke 2000, Chin 2002). In Russia, the threat of human access to tiger habitat from public roads and logging roads decreased the survivorship and reproductive success of tigers (Kerley et al. 2002). The position of roads confined suitable tiger habitat within the borders of KSNP. The impact of roads in KSNP was extensive and produced a marked reduction in habitat quality, correlated with tiger abundance, of 1.64 km from the forest edge. This is a greater distance than that recorded for tiger prey (0.61 km) and highlights the greater vulnerability of top carnivores to edge effects. A notable absence of tigers was recorded up to 3 km from the forest edge in BBSNP in South Sumatra (Kinnaird et al. 2003). The more prominent edge effects arise because BBSNP is smaller than KSNP and therefore less able to buffer the much higher local community demands upon forest resources in and around BBSNP.

Tiger distribution in the KS region was determined by the distribution of their prey, as would be expected (Sunquist 1981, Karanth 1995, Carbone and Gittleman 2002). Poaching was identified as posing a principal threat to tiger prey in the KS region and this indirectly threatens tigers. Much of the hunting of large forest mammals in tropical Asia is unsustainable (Robinson and Bennett 2000). The over-hunting of tiger prey by humans has led to widespread declines over the tigers range (Karanth 1991, Rabinowitz 1993). Prey base decline has a strong and adverse effect on tiger population dynamics and represents a major driving force behind the decline of tigers (Karanth and Stith 1999). Due to the ecological constraints of living in tropical evergreen forests, tigers occur at particularly low densities in KSNP, which makes them even more sensitive to the effects of prey base decline. The survival of tigers in KSNP is therefore dependent on sound management strategies and practices, such as those aimed at connecting and protecting the three core tiger populations.

In the southern section of KSNP, core area 2 was identified with an estimated 98 adult tigers. This population was deemed viable even when 5 tigers a year were poached. In the northern and eastern sections of KSNP core area 1 and 3 were identified with much smaller estimated adult populations (20 and 15 tigers, respectively). A minimum population of between 50-100 individuals has been recommended for tigers and other large mammals (Seidensticker 1986; Shaffer 1987, Allen et al. 2001). The small population sizes in cores areas 1 and 3 should therefore be particularly vulnerable to extinction through stochastic processes (Soulé 1980, Kenney et al. 1995). Nevertheless, Karanth and Stith (1999) found that even isolated tiger populations containing only six breeding females could still remain viable after 100 years, as was similarly the case in KSNP.

The PVA model predicted that the two smaller tiger populations could both remain viable if well protected, low poaching of 1 tiger per year. However, the PVA model also highlighted the relative vulnerability of the two smaller tiger populations, because both were predicted to reach extinction within 50 years if 3 tigers a year or more were poached. Connecting the two smaller tiger populations to the largest population was predicted to greatly increase their survival, because the latter acted as an important source population that would counteract the effects of poaching from the smaller populations. This presents a strong argument for ensuring the predicted habitat fragmentation between core area 1 and 2 does not isolate these populations (Chapter 6). It also emphasizes the merits of a well coordinated anti-poaching forest patrol strategy aimed at core areas 1 and 3.

The detrimental effects of roads on large carnivores have been widely documented across their ranges (Noss et al. 1996). The effect of roads on large carnivores varies between and within species, and therefore has important implications for their management. For example, grey wolves were found to avoid frequently used roads (Thurber et al. 1994). Female Florida panthers did not cross roads, which thus represented a barrier to their range, whereas male Florida panthers did cross roads and suffered high road kill mortalities as a consequence (Maehr 1997). The salient findings from this study should serve as a caveat when considering the construction of new roads in the KS region. For example, local government in West Sumatra and Bengkulu provinces are keen on developing the area around KSNP for tree crops and plantation estates by supporting roads construction through and around KSNP. To ensure the protection of tigers, the most precautionary strategy would be to veto any such proposal. Expanding the small estimated tiger adult populations in core areas 1 and 3 could be facilitated by habitat restoration between the areas. The establishment of wildlife corridors is not a novel idea in Sumatra. Elephant and tiger corridors reconnecting four forest blocks in PAs are planned for a *c.* 1880 km^2 area, Tesso Nilo, northeast of KSNP. The aim of this project is to join and protect remaining forest through re-zoning and improved law enforcement. The benefits to local communities would be through reduced conflicts with elephants and tigers, increased agricultural yields, and new employment opportunities (WWF 2003). Taking a landscape approach in the KS region would require the cooperation of the local communities that adjoin KSNP, but is feasible (Chapter 3). The final chapter discusses these issues and the other pertinent issues for tiger conservation management in the KS region in light of the findings from this thesis and other studies.

Chapter 10

RESEARCH AND CONSERVATION MANAGEMENT OF TIGERS

KS-Tiger Protection and Conservation Unit patrolling inside the national park (TNKS)

10.1 INTRODUCTION

This thesis provides new information on the specific threats relating to tiger habitat, tiger prey species, and directly to tigers in the Kerinci Seblat (KS) region. The findings are also significant for the conservation of tigers across their range and for large carnivores in general. I will now summarize and set in context the major findings of this study and use this information to guide and develop more salient and focussed management strategies.

10.2 MAIN RESEARCH FINDINGS

The main research findings in the six data analysis chapters in this study are summarized below.

10.2.1 Pioneer farming around KSNP

If tigers, their prey, and their forest habitat are to be effectively protected, Chapter 3 showed that it is crucial to work in cooperation with the local communities. Interview surveys with a pioneer farming community living adjacent to Kerinci Seblat National Park (KSNP) showed that most farmers had positive attitudes towards tigers and their conservation. Furthermore, most farmers thought that conserving tigers in KSNP was important. However, farmer's attitudes may also be linked to the benefits received from KSNP, such as collecting non-timber forest products (NTFPs). Although such activities are prohibited inside KSNP, most farmers were unsure about the restrictions associated with KSNP. These pioneer farmers represent some of the poorest people in Indonesia. The supplementary income they receive from the sale of NTFPs is important and gives the farmers a vested interested in maintaining the existing forest cover. However, cutting down the forest for farmland still provides the subsistence way of life. Most farmers thought that the greatest limitation to their agricultural success was from crop raiding by wildlife. Most farmers also thought that deforestation would adversely affect them, through increased soil erosion and flooding. The farmers also thought that deforestation would adversely affect tigers and their prey, leading to population decreases. The views of these farmers informed the rest of the research, which took the approach of developing a Geographic Information System (GIS), as fully explained in Chapter 4.

10.2.2 Forests and the forest sector

The KS region experienced a mean annual deforestation rate of 0.96%, based on a GIS analysis of remotely sensed data on the KS region from 1995 to 2001 (Table 10.1). Each district in the KS region contained on average just over 50% forest in 2001. The largest districts of Solok and Bengkulu Utara contained the largest areas of forest in 1995, but also lost the largest areas of forest. The KSNP itself experienced a much lower mean annual deforestation rate. Inside the logging concessions (HPHs) and estate crop plantations mean annual deforestation rates were some of the highest within the KS region (Table 10.1). Hill forest was the most abundant and most cleared forest type in the KS region. The mean deforestation rate of hill forest was less than that of lowland forest, which underwent the most rapid loss (Table 10.1).

Table 10.1: Summary of deforestation within different localities of the KS region

Locality	Area of forest (km^2) in 1995	Area cleared (km^2) from 1995-2001	Mean deforestation rate (%/yr)
KS region	22327.1	1278.4	0.96
Solok	4166.0	314.7	1.27
Bengkulu Utara	3552.8	194.9	0.92
KSNP	12657.7	207.5	0.28
HPHs	4805.9	681.5	2.96
Estate crop plantations	498.5	204.2	5.91
Hill forest	9693.1	588.0	1.01
Lowland forest	2355.3	368.3	2.61
Bengkulu Utara	1568.9	184.0	1.96

The low rates of deforestation inside KSNP are encouraging for conservationists, especially when compared with the mean annual deforestation rate of 2% recorded from inside Bukit Barisan Selatan NP (BBSNP) (Kinnaird et al. 2003). However, the results from KSNP should be interpreted cautiously because, unlike BBSNP, there are still reasonably large blocks of forest outside the PA, which act as a buffer, as opposed to the PA status of KSNP offering enhanced protection. Lowland forest in

Sumatra is predicted to be logged most rapidly and indeed, to be completed cleared by 2005 (Holmes 2001). This makes a special case for focussing protection in Bengkulu Utara district. Although it was identified as containing over 70% of the lowland forest in the KS region, the loss of lowland forests in this district (1.96%/yr) was much higher than overall forest loss in the region. On a positive note, however, this forest was found to mainly occur inside HPHs, making it possible for a partnership of NGOs to buy out the lease from the logging companies and protect these areas privately.

In Sumatra, Birdlife International and the RSPB are currently trying to purchase the lease from a HPH concession to run it as a management concession (Birdlife 2004). The concession is comprised of approximately 600-800 km^2 lowland dry forest, of which 25-30 % is still good forest with the remaining being in various stages of degradation. An associated problem with buying such a lease is that by law a HPH concession must be logged in part. However, a new law now permits the use of a HPH concession for restoration. The legal entity of this area must be determined.

*10.2.3 Mapping and predicting deforestation**

Areas that were at lower elevations, nearer to settlements and public roads, on flatter terrain, and outside of KSNP were more likely to be cut down, based on an analysis of physical factors that best predict the likelihood of deforestation. The analysis also predicted that the large patches of forest occurring outside of KSNP in the northeast, east and southwest of the KS region were most at risk from deforestation. Forest within KSNP is generally less at risk, but areas in the central section close to the asphalt road were found to be most susceptible. The predicted pattern of deforestation suggested that areas of forest loss would steadily decline but, as deforestation progresses, the mean annual rates of deforestation would increase more rapidly. The small amount of lowland forest was predicted to decline steadily from the first stage of deforestation and is predicted to disappear much more quickly than the other forest types. Hill forest was the next most susceptible to clearance, followed by submontane forest and lastly by montane forest.

The predicted pattern of deforestation suggests that forest losses will steadily decline until deforestation stages 4 ($P > 0.6 - 0.7$) when the losses will increase more rapidly until stage 8 ($P > 0.2 - 0.3$), when only a small amount of forest will remain.

Increasingly larger areas of forest would be lost from stages 1 to 9. There was no significant difference in the proportion of forest loss between ICDP and non-ICDP villages, indicating that the KS-ICDP did not meet its conservation objectives. Unsurprisingly, deforestation in villages occupied by a HPH was significantly higher than those villages without HPHs. However, within villages occupied by HPHs, analysis of deforestation using socio-economic and physical factors showed that deforestation was positively related to the proportion of a village occupied by a HPH and negatively related to slope. This suggests that empowering local communities with secure property rights would have the greatest effect on reducing forest loss. The habitat loss and fragmentation patterns predicted for the KS region will split KSNP into three sections, which in turn will have serious consequences for tigers and their prey.

10.2.4 Determinants of crop raiding patterns

Interviews surveys showed that a minority (30%) of farmers employed a guarding strategy against crop pests, and their main strategy was to own a guard dog, as shown by Chapter 7. An overwhelming majority of farmers perceived wild boar to be the worst crop pest (80%), followed by pig-tailed macaque (75%). By monitoring crop raiding over 5 months, a total of 348 independent crop raiding forays by five species of wildlife were recorded. Wild boar indeed raided crops most frequently (76.4%), followed by pig-tailed macaque (23.6%), but pig-tailed macaque caused significantly more damage (73.1%) than wild boars (25.9%) when crop raiding. Both these species primarily raided bananas, but pig-tailed macaque favoured fully matured crops, whereas wild boar were unselective. The spatial distribution of crop damage showed that all wildlife, and wild boar and pig-tailed macaque individually, most frequently entered and caused the greatest amount of damage in farms that were closest to the forest edge.

10.2.5 Tiger prey in human altered landscapes

A GIS based study of the distributions of tiger prey species in Chapter 8 showed that mouse deer, muntjac, sambar and wild boar were more likely to occur in areas that were further away from public roads. In addition, sambar were more likely to occur at higher elevations and wild boar were more likely to occur in areas that were further away from logging roads. Taken overall, tiger prey were more likely to occur in areas that were further away from logging roads and public roads. Tiger prey was most likely to prefer habitats inside KSNP, and to occur in six core areas in the interior of KSNP that were remote and inaccessible by the majority of the roads. The network of public roads and logging roads in the KS region had a profound impact on reducing habitat quality for tiger prey by an average of 0.61 km from the forest edge. Snare traps were more likely to be set in areas that were closer to logging roads and further away from rivers. In addition, snare traps were more likely to be found near richer villages, further challenging the logic behind the KS-ICDP village development strategy.

10.2.6 Tiger resilience in a fragmented landscape

Tigers were more likely to occur in areas that contained prey species and that were further away from public roads, as shown by a GIS based study in Chapter 9. The logistic regression model constructed for tiger habitat suitability had one highly significant factor: proximity to public roads. Good quality tiger habitat was represented in three main areas in the KS region, that were all largely inside KSNP. The sizes and mean distances from the forest edge to the perimeter of each core area varied (Table 10.2). Core areas 1 and 3 were both quite large but mainly comprised submontane and montane forest and consequently supported much smaller adult tiger subpopulations.

Table 10.2: Core tiger habitat distance to forest edge, area and estimated tiger subpopulation

Core area	Distance to forest edge	Size (km^2)	Estimated number of tigers
1	1.98	1667.0	20
2	1.54	5689.2	98
3	1.41	1219.1	15

A time-discrete stage-structured population model showed that these three tiger populations inside KSNP would remain viable if all were well protected. However, in a scenario where poaching removed 3 tigers per year from each area, then the two smaller populations were predicted to go extinct within 50 years. Connecting the two smaller tiger populations to the largest population was predicted to ensure their survival, because the latter acted as a source population that would offset the effects of poaching. This presents a strong argument for ensuring the predicted habitat fragmentation in KSNP does not isolate populations from core areas 1 and 2. It also emphasizes the merits of an effective and well coordinated forest patrol strategy.

The competition between farmers and wildlife over space and resources identified in this thesis epitomizes the current threats facing tiger across the KS region and across all tiger range states. The research findings suggest that the principal threat of deforestation not only adversely affected tigers and their prey, through habitat loss, but also the farmers who were involved in the logging, through loss of NTFP and crop raiding by wildlife. Forest fragmentation and disturbance was shown to determine the distribution of tiger prey species and then tigers. The severity of these predicted patterns of forest fragmentation was shown to cause the extinction of tigers in the smaller areas of KSNP if they were exposed to low levels of poaching. It is therefore vital to maintain connectivity between the large forest blocks. I now consider the conservation measures that could be used to lessen this threat.

10.3 CONSERVATION MANAGEMENT CONSIDERATIONS AND RESEARCH

To lessen deforestation in the KS region, small scale subsistence farmers need to be offered realistic alternatives to cutting down the forest.

10.3.1 Alternative agriculture

The majority of farmers living near KSNP cultivate sun tolerant coffee that is grown without an overstorey and involves the clearance of forest. Shade grown coffee schemes require a forest canopy and therefore offers an attractive alternative for conservation organizations to promote (Conservation International 2000, Rainforest Alliance 2000). In February 2004, Verde Ventures (managed by Conservation International) invested US$200,000 in finance for shade grown coffee produced in Aceh, northern Sumatra, by ForesTrade partner, the Gayo Organic Coffee Farmers' Cooperative Association (Conservation International 2004). In Central America, poor farmers have accrued modest benefits from switching to shade varieties of coffee instead of growing sun coffee (Pagiola and Ruthenberg 2002). A price premium is paid to producers because the financial returns per hectare for sun coffee are much greater than for shade coffee. This may be acceptable because mammalian and avian diversity in shade coffee stands tends to be greater than in sun coffee stands, but it is still substantially lower when compared to the biodiversity in natural forests (Gallina et al. 1996, Roberts et al. 2000, Petit and Petit 2003). However, shade coffee, although a benign form of agriculture, cannot provide the ecosystem services of the natural forest. It may therefore be better to pay communities to maintain intact forests. This can be achieved through indirect or direct payments.

10.3.2 Indirect versus direct conservation payments

It is difficult for KSNP to generate funds through tourism or wildlife hunting that are sufficient to finance itself and compensate communities for their loss of access to resources or opportunities. The ICDP concept may therefore have seemed appropriate because it offered effective biodiversity conservation, increased local community participation in conservation and development, and economic development for the rural poor. As this study suggests, however, the indirect payments offered by an ICDP do not provide realistic alternatives to clearing the forest (Sayer et al. 2000). One of the difficulties of implementing the KS-ICDP was its complexity and the ambiguous incentives designed for preserving biodiversity. Instead direct payments to individuals or communities for conservation performance may be a simpler and more cost effective approach than ICDPs (Ferraro and Kiss 2002, Ferraro and Simpson 2002). If developed countries want a continuation of the ecosystem services provided by tropical forests, and the existence values such as providing tigers with a habitat, then they must make direct payments to compensate developing countries for lost opportunity costs, e.g. from timber extraction. An interesting approach that holds much promise is being tried in Costa Rica.

In Costa Rica, local, national, and international beneficiaries of ecosystem services pay landowners through the National Forestry Financial Fund (*Fondo Nacional de Financiamiento Forestal*, or FONAFIFO) (Castro et al. 1998). Support from the World Bank (US$33.9 million loan) and GEF (US$8 million grant) for FONAFIFO is to be used in part to provide financial incentives to small and medium-sized landowners to conserve primary forests, encourage sustainable management of secondary forests, and promote reforestation efforts throughout Costa Rica. Whilst the initial signs are promising, there are a number of weaknesses that include a lack of recognition of environmental values and the need for improved monitoring through field control (Anon. 2002[b]). This market-oriented approach must also heed caution because farmers may become dependent on these payments or demand more money. Farmers may sell off the land to an illegal logging mafia that offers more money or threatens to log without consent. However, in such circumstances indirect payments would not even work either. In situations where communities have weak property rights, payments would not be to the government but would be to the individuals, which is a labourious process that would require close monitoring (Angelsen and Wunder 2003).

Another advantage of direct payments, relevant to the KS situation, is that they create a local stake in ecosystems because they strengthen the links between individual well-being, individual actions, and habitat conservation (Ferraro 2001). An alternative and more binding way to increase the local stake in the ecosystem would be to actually provide individuals with secure land tenure rights instead of surrogates. From the perspective of KSNP, this would be a sensible approach because land insecurity emerged as the pertinent issue associated with deforestation.

10.3.3 Land tenure rights

A positive relationship has been reported previously between effective property rights and loss of forest cover in the tropics (Saxena 1988, Dorner and Thiesenhusen 1992, Southgate et al. 1991, Southgate 1992, Angelsen 1996, Pichón 1997, Nelson et al. 2001, Ochoa-Gaona 2001). In Ecuador, the national government recognizes the rights of some communities to govern their local affairs, in which the communities are empowered and assigned land ownership rights. This common land is then distributed between community members and treated as private property, with the only proviso being that they must use the land and they must not sell it. This land adjoins forest and in the parts of the forest that have not been allocated to individuals, outsiders have caused significant forest degradation, removing up to 70% of the forest cover (Gibson and Becker 2000). Although there has been a recent change in Indonesian law whereby the Government recognizes *adat*, the situation remains similar to before because the national government is unwilling to grant legal property rights to local communities or where these do exist only protects them weakly (Inamdar et al. 1999, BAPPENAS 2003).

Clearly defined land tenure rights still do not ensure good forest management because forest management regimes may differ within a community (Gibson and Becker 2000, Banana and Gombya-Ssembajjwe 2000). The short term financial benefits of clearing the forest may seem most appealing to poor communities and undermine long-term local livelihoods. Land titles should therefore have a few clear stipulations to safeguard the forest from such actions. They should not be overly restrictive by denying certain resource uses, such as collecting NTFPs or fuel wood, but they should deny certain resource uses, such as clearing watershed areas, and clearing inside PAs. Local level institutions are needed that are capable of producing explicit rules, monitoring and enforcing these rules, and resolving disputes (Berkes 1989, Berkes and Folke 1998). To make enforcement more efficient, a consensual agreement of the rules is required. Monitoring local management and guardianship of habitats can be achieved through mapping forest integrity at a fine scale with more recent satellite images and using the method outlined in Chapter 5. This will help to identify areas of forest in the villages adjoining KSNP, identifying areas undergoing disproportionately high levels of clearance compared to the background deforestation rate. These areas can then be targeted for special protection and a more detailed assessment of habitat maintenance can be conducted through forest patrols.

10.3.4 Law enforcement

Law enforcement is an essential component of a good PA, especially in a country with poor governance. Inadequate law enforcement in KSNP was a major obstacle to the successful implementation of the ICDP. The extent to which the effective enforcement of laws and regulations is a basic requirement for successful ICDPs is little appreciated (Wells et al. 1999). In KSNP, there are large profits to be gained from illicit use of forest resources, such as growing cinnamon inside the national park. Developing effective agreements that limit land use may therefore be difficult, which in turn makes enforcement even more important (Muttulingam and Shen 1999).

For law enforcement to be effective, forest patrols need to be focused at key locations rather than spread too thinly over a large PA (Leader-Williams and Albon 1988). In KSNP likewise, the number of rangers per patrol was also related to patrol effectiveness and success (Linkie et al. 2003). Therefore, law enforcement needs to target the priority areas in the KS region.

Logging roads and public roads were identified as very important factors in this study. Between them, they explained forest loss, snare trap location, and the distribution of tigers and their prey. Unambiguous law enforcement measures are needed to lessen the impact of these factors. It would be sensible to follow the actions of the Malaysian Director of Forests, who in April 1999 decreed that logging companies must close the entrance to logging roads once a block has finished being logged, so that vehicle access was prevented. Nevertheless, these roads will still be accessible on foot, so it would be rational to identify which routes could be guarded to minimise uncontrolled access to KSNP, because law enforcement staff cannot properly cover the entire area. Although these roads can pose a serious threat, they can also provide the opportunity to increase the ability of forest patrols and to provide enhanced law enforcement cover (Madhusudan and Karanth 2000). Where enforcement is strong and detection rates are high, this can be effective (Milner-Gulland and Leader-Williams 1992, Leader-Williams and Milner-Gulland 1993, Dinerstein et al. 1999).

Strong enforcement in KSNP would also require an augmentation of forest patrols. KSNP forest police staff need to become more involved with TPCUs through training and coordination of patrol activities. This would to allow the northern and southern sections of KSNP to be better patrolled, conflict areas to be patrolled more frequently, and the provision of cover for TPCUs when they need to pursue wildlife traders outside KSNP (Linkie et al. 2003). Involvement of army personnel that are not normally based in the provinces surrounding KSNP has proved to be successful in halting illegal logging activities around KSNP (Linkie and Sibarani 2002). Given that army personnel have been complicit in illegal logging activities, this option would have to be backed by strong support from provincial government, kept independent by rotating army personnel between provinces, and be monitored by an NGO watchdog. In Paguyaman Reserve, on the Indonesian island of Sulawesi, the deployment of six special forces soldiers on 24 hour patrol resulted in the cessation of illegal logging, poaching and slash-and-burn clearance within the reserve (Clayton 2004). Results of operations against illegal loggers included the confiscation of $90m^3$ (seven lorry loads) of top quality illegal timber at the reserve, the confiscation of a timber, chain-saws and axes from other offenders and a two-month jail sentence for one chain-saw owner.

Forest patrols targeting illicit activities pose inherent risks to patrol staff. These patrols involve hard physical exercise, so for patrols to be more effective, patrol staff need to have high morale and motivation. This could be achieved by establishing a scheme that rewards success in making arrests and confiscating equipment such as chainsaws and snare traps. In India, WWF awards and honours diligent forest rangers and those committed to wild tiger conservation. In Khao Yai NP, Thailand, where NP guards have been killed, tough policing and punishment successfully deterred resource extraction from the centre of the park but resulted in villagers undertaking complex avoidance activities to reduce their chances of being caught (Alber and Grinspoon 1997). An integral part of conservation success and PA viability is community support (Hannah 1992, Ite 1996). To bolster community support for tough law enforcement and tiger conservation a community education programme should compliment the work of the forest rangers.

10.3.5 Community education and outreach programme

At present KSNP does not have a formal education programme geared towards tiger conservation, although the KS-ICDP did run biodiversity workshops for school children. In KSNP, forest rangers could meet with schoolchildren and give talks on tiger conservation, why they protect tigers and why they protect the tiger's forest habitat. These talks could include photographs of the effects of deforestation: massive floods that destroyed crops and asphalt trade roads in the KS region during the 2000 monsoons. These floods resulted in frequent electricity power cuts, loss of fuel supplies as delivery tankers were unable to reach isolated towns, and economic costs from lost crops. Educational activities need to be fun by combining play with real conservation messages. As they become better educated, the children could convey important messages about tiger conservation to their parents who are most likely farmers, as this is the main source of employment. This in turn could stimulate conversations about tigers in the region. Another approach to stimulating parental interest through their children might be to hold painting and poetry competitions so that school children work on their entries at home. Tiger quizzes between schools in the region is another viable method that can be fun, informative, and stimulate more interest in tigers. Prizes containing tiger and forest logos might help to raise awareness about tigers.

Community outreach programmes could be undertaken to convey the conservation message directly to adults. Such programmes could aim to develop good relationships between local communities and forest patrol staff, educate local communities about the benefits from wildlife conservation, and discourage local communities from poaching or committing illegal activities (WildAid 2002, WWF 2002). Community education programmes can promote stewardship of a PA, if communities realize the indirect benefits received from preserving forest. Villagers living around Khao Yai NP who were targeted for an education programme, said they recognized that the increase in their drought and flood problems was due to deforestation. In turn, this created social pressures on those involved in illegal logging (Albers and Grinspoon 1997). The community outreach programmes should initially gain information from the communities bordering KSNP to identify community concerns, problems, possible solutions, and which communities should be targeted for additional programme activities. However, community outreach programmes are not without criticism because their activities are often expensive, their conservation benefits are ambiguous, and they have little prospect of generating income to cover their costs, indicating that the direct payments approach, as previously discussed, might be more germane (Inamdar et al. 1999).

When people live in close proximity to large carnivores, conflict is inevitable (Madhusudan and Mishra 2003). Guarding and behavioural measures can be adopted by humans to minimize this conflict. One of the outputs of this education programme should be to produce and distribute guidelines within a simple pictorial booklet showing villagers how to minimize conflict with tigers through modification of human behaviour and livestock management practices. For example, 98% of livestock depredations by tigers in South India occurred during the post harvest seasons (Madhusan 2003). These were periods when there would be substantially less human activity in farmland during the day and less guarding of crops during the night. However, conflict will always be present and, when appropriate, villagers could be compensated for loss of life, injury, or loss of livestock.

10.3.6 Tiger compensation schemes

There are no fully-fledged compensation schemes currently operating in KSNP or Sumatra more generally so firstly there needs to be an investigation into the feasibility of setting up a compensation scheme for villagers that suffer loss from sharing a landscape with tigers. An unambiguous protocol would be necessary to promptly reimburse communities suffering loss of life and livelihood from tigers. If compensation schemes do not exist, then the chances of local communities feeling animosity towards tigers will increase, making retribution a more justified response. Human-tiger conflict compensation schemes already exist elsewhere so lessons can be learned from their merits and shortfalls.

10.3.6.1 Livestock depredation

In Bhadra Tiger Reserve, South India, a government compensation scheme set up to reimburse owners of livestock killed by tigers and leopards failed to satisfactorily compensate (Madhusan 2003). The incentives to claim for compensation were weak: the Forest Department compensated only 3% of the total losses incurred by villagers; the average final payment was only 27% of the original request; the processing time took on average 180 days until payment.

In Northern India, WWF-India offered a compensation scheme to augment the one provided by the Government of Uttar Pradesh (Talwar 1999). The previous government scheme had comparable problems to the one reported from South India. The WWF scheme provided additional incentives for immediate information on the occurrence of livestock depredation, and immediate compensatory payment for the full value of the animal.

An important factor in human-tiger conflict resolution is response time, because inaction by the relevant authority can lead to villagers removing a problem tiger or opportunistic poaching. A typical method of retribution is to lace the killed livestock with poison. Therefore, in Uttar Pradesh, the person who reported the occurrence of a kill (not necessarily the livestock owner) received a reward based on how quickly they contacted park officials. Funds were provided by WWF TCP. If reported within 24 hours of occurrence a Rs 300 reward was given, within 24 to 48 hours a Rs 200 reward and with 48 to 72 a Rs 100 reward. No compensation is paid after 72 hours because it is unlikely that there will be anything left of the carcass to poison.

The success of the WWF-India programme can be evaluated from its results in Corbett Tiger Reserve and Dudhwa Tiger Reserve. Before the compensation scheme was implemented in 1998, nine tigers were poisoned in a period of a little over 2 months in both the parks and their adjoining forests. Since then there has been only one reported case of poisoning.

A problem with compensation schemes is that they are open to abuse. To guard against fraudulent claims checks need to be built in. The livestock carcass therefore has to be found and inspected and if it is reasonably fresh, it is easy to find out whether the animal has been killed by a tiger or has died of a disease or old age. Evidence of tiger pugmarks should be noted. Again the reporting and response needs to be rapid so that all available evidence can be assessed, e.g. the carcass has not already been consumed, and whether or not the location of the kill is inside a prohibited area. Ownership of the livestock should be established with the village elders. The livestock should then be removed and destroyed. There may be some false claims, but as long as these are only a few and all owners of genuine tiger kills are compensated then this is acceptable. One problem identified in Bhadra Tiger Reserve was that villagers were required to produce evidence of land ownership, when in many cases they did not have official property right documents. In the KS region this would obviously present a problem. To overcome this it might be acceptable to consult the village head who would be able to provide evidence of their land status within the village. This would be an unofficial document and would be open to abuse so requires close scrutiny.

Wells et al. (1992) caution against well developed compensation schemes because they may attract outsiders and create additional problems. This could be curbed through the allocation of secure property rights to indigenous villagers. Sariska Tiger Reserve (STR), in Rajasthan, central India, provides a contrast to the previous Tiger Reserves. In STR, there are no compensation schemes for villagers that suffer livestock loss to tigers or leopards or crop damage from tiger prey, such as wild boar and nilgai. Yet, villages still have positive attitudes towards STR and are tolerant towards tigers because they receive benefits from the reserve in the form of fuelwood collection and fodder for livestock. These attitudes are further strengthened through the benign religious and cultural beliefs of villagers (Sekhar 1998, but see

Madhusudan and Karanth 2002). In Nepal, villagers had positive attitudes towards Chitwan NP where they had access to, and use of, grasses (Studsrød and Wegge 1995). These were considered as a form of compensation for living in the vicinity of Chitwan NP and suffering from wildlife conflict (Lehmkuhl et al. 1988). There are problems with this situation because communities may become heavily dependent on resources within the PA and over exploit it (Straede and Helles 2000). Indeed, such resources can actually provide more forage for tiger prey species, such as chital (Moe and Wegge 1997). Furthermore, it may not send out a good sign for law enforcement in a PA because it may lead to a change in perceptions that other resource use activities are permissible.

10.3.6.2 Human injury and loss of life

The Malaysian government has set up a trust fund for people who have been attacked by elephants, tigers or wild boars. The trust fund was set up as a token of sympathy to victims of wildlife attacks and is in response to the dramatic increase in attacks by tigers from 7.7% in 1998 to 41% in 2001. The 'Wildlife Attack Victims Assistance Fund' will pay up to 5,000 Malaysian ringgits (approximately US$1,316) to victims injured by tiger, elephant, and wild boar. Double that amount will be paid to the relatives of those who are killed. It does not compensate for loss of livestock or property damage resulting from wildlife attacks. The increase in attacks was apparently due to the loss of habitats and a bigger tiger and elephant population as the result of better protection from the authorities. To determine how the tiger population is responding to law enforcement activities, a well designed monitoring programme needs to be established.

10.3.7 Monitoring and evaluating tiger management

If the ultimate aim of KSNP is to conserve tigers in the wild, then the success of all strategies can be assessed by monitoring tiger population trends. Population trends that are stable or increasing would result from successful management regimes. The final step therefore is to evaluate KS-management strategies in accomplishing this objective. KSNP will therefore need to monitor tiger and prey species population trends using reliable methodologies (Miquelle 2001). A list of key monitoring areas will be drawn up from consultation with KSNP management, Fauna & Flora International and provincial universities. In areas identified as a high priority, monitoring will involve the use of camera traps to determine estimates of absolute tiger densities (Karanth and Nichols 1998, 2002). For wider surveys across KSNP, monitoring will use detection-non detection surveys across the four different forest habitat types. Each individual study site will be divided into 2 km^2 grid cells. Between 30 and 40 cells will be selected and monitored for tiger and prey sign during the rainy season, when signs are easiest to detect. Each cell will be surveyed five times during the course of one season to reduce problems with 'false absences', i.e. recording an animal as absent when in fact it has not been detected (MacKenzie et al. 2002, 2003). To cover key areas across the four provinces that KSNP spans it would be useful to collaborate with provincial universities to maintain sustained effort.

10.4 LARGE CARNIVORES AND HUMANS

The Sumatran tiger is a charismatic species and is used to promote wildlife conservation in KSNP, but the Sumatran tiger in KSNP remains under threat and its status does not show signs of improving. Large carnivores are often used as focal species (indicators, umbrellas, or flagships) in strategies either aimed at conserving carnivores, the wider biodiversity that occupies their habitats, or both (Linnell et al. 2000, Leader-Williams and Dublin, 2001). This helps to generate conservation funding for these species and also those that occur 'under their umbrella'. If the current global extinction crisis is to be averted, then using the Sumatran tiger to attract international donor funds to protect the incredibly rich biodiversity in the country that is currently experiencing the highest rates of deforestation in the world is an imperative.

References

Achard, F., Eva, H.D., Stibig, H.J., Mayaux, P., Gallego, J., Richards, T. and Malingreau, J.P. 2002. Determination of deforestation rates of the world's humid tropical forests. Science 297: 999-1002.

Albers, H. and Grinspoon, E. 1997. A comparison of the enforcement of access restrictions between Xishuangbanna Nature Reserve (China) and Khao Yai National Park (Thailand). Environmental Conservation 24: 351-362.

Allen, C.R., Pearlstine, L.G. and Kitchens, W.M. 2001. Modelling viable mammal populations in gap analyses. Biological Conservation 99: 135-144.

Alston, L.J., Libecap, G.D. and Schneider, R. 1995. Property rights and the preconditions for markets: the case of the Amazon frontier. Journal of Institutional and Theoretical Economics 151: 89–107.

Ananta, A., Anwar, E.N. and Miranti, R. 2000. Age-sex pattern of migrants and movers: a multilevel analysis on an Indonesian data set. Asian Metacentre Research Paper Series Number 1. Asian Metacentre, University of Singapore, Singapore.

Angelsen, A. 1996. Deforestation: population or market driven? Different approaches in modeling of agricultural expansion. CMI working paper number 9. Chr. Michelsen Institute, Bergen.

Angelsen, A. 1997. The poverty-environment thesis: was Brundtland wrong? Forum for development studies 1: 135-154.

Angelsen, A. 1999. Agricultural expansion and deforestation modelling: the impact of population, market forces, and property rights. Journal of Development Economics 58: 185-218.

Angelson, A. and Resosudarmo, D.P. 1999. Krismon, farmers and forests: the effects of the economic crisis on farmers' livelihoods and forest use in the outer islands of Indonesia. CIFOR Bogor, Indonesia.

Angelsen, A. and Wunder, S. 2003. Exploring the forest - poverty link: key concepts, issues and research implications. CIFOR occasional paper number 40. CIFOR Bogor, Indonesia.

Anon. 2001. A summary report of the Forest Law Enforcement and Governance East Asia ministerial conference. Sustainable Development 60: 1-15.

Anon. 2002[a]. Conserving Kerinci Seblat National Park: some encouraging results, but still much to do. Unpublished Report, ICDP Component A, Jaakko-Poyry Consulting, Sungai Penuh, Indonesia.

Anon. 2002[b]. International expert meeting on forest landscape restoration. Sustainable Development, 71: 2-8.

Armitage, D. 2002. Socio-institutional dynamics and the political ecology of mangrove forest conservation in Central Sulawesi, Indonesia. Global Environmental Change-Human and Policy Dimensions 12: 203-217.

Armitage, D.R. 2003. Traditional agroecological knowledge, adaptive management and the socio-politics of conservation in Central Sulawesi, Indonesia. Environmental Conservation 30: 79-90.

Ashenafi, Z.T. 2001. Common property resource management of an Afro-alpine Habitat; supporting a population of the critically endangered Ethiopian wolf (*Canis simensis*). Unpublished Ph.D. thesis. University of Kent at Canterbury, UK.

Augustine, N.H., Mugglestone, M.A. and Buckland, S.T. 1996. An autologistic model for the spatial distribution of wildlife. Journal of Applied Ecology 33: 339–347.

Aumeeruddy, Y. 1992. Agroforestry in the Kerinci valley: a support to buffer zone management for Kerinci Seblat National Park-Sumatra, Indonesia. Institut de Botanique, Montpellier and PHPA, Bogor, Indonesia.

Auzel, P. and Wilkie, D.S. 2000. Wildlife Use in Northern Congo: Hunting in A Commercial Logging Concession. In: J.G. Robinson and E.L. Bennett (eds.) Hunting for Sustainability in Tropical Forests. Columbia University Press. New York.

Badola, R. 1998. Attitudes of local people towards conservation and alternatives to forest resources: a case study from the lower Himalayas. Biodiversity and Conservation 7: 1245-1259.

Bailey, T.N. 1993. The African leopard: ecology and behavior of a solitary felid. Columbia University Press, New York.

Bakels, J. 1994. The Tiger by the Tail: On Tigers, Ancestors and Nature Spirits in Kerinci. In: R.L. Tilson, K. Soemarna, W. Ramono, S. Lusli, K. Traylor-Holzer, and U.S. Seal. (eds), Sumatran Tiger Population and Habitat Viability Analysis Report. Indonesian Directorate of Forest Protection and Nature Conservation and IUCN/SSC Conservation Breeding Specialist Group, Apple Valley, Minnesota.

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R.E., Jenkins, M., Jefferiss, P., Jessamy, V., Madden, J., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J., Trumper, K. and Turner, R.K. 2002. Ecology - Economic reasons for conserving wild nature. Science 297: 950-953.

Banana, A.Y. and Gombya-Ssembajjwe, W. 2000. Successful forest management: the importance of security of tenure and rule enforcement in Ugandan forest. In: C.C. Gibson, M.A. McKean and E. Ostrom (eds), People and Forests: Communities, Institutions, and Governance (Politics, Science and the Environment). The MIT Press, Cambridge, MA.

BAPPENAS-ADB. 1999. Causes, extent, impact and costs of 1997/1998 fires and drought. Final Report, Annex 1 and 2. Planning for Fire Prevention and Drought

Management Project. Asian Development Bank TA 2999-INO. National Development Planning Agency (BAPPENAS) and Asian Development Bank, Jakarta.

BAPPENAS. 2003. Indonesian biodiversity strategy and action plan 2003-2020. National document, National Development Planning Agency (BAPPENAS), Jakarta.

Barber, C.V. 1997. "The Case Study of Indonesia." Project on Environmental Scarcities, State Capacity, and Civil Violence, University of Toronto and the American Academy of Arts and Sciences.

Barbier, E.B. 1997. The economic determinants of land degradation in developing countries. Philosophical Transactions of the Royal Society Series B 352: 891-899.

Barbier, E.B., Bockstael, N., Burgess, J.C. and Strand, I. 1993. The timber trade and tropical deforestation in Indonesia. LEEC Paper DP 93-01. London Environmental Economics Centre. London.

Barr, C. 2001. Banking on sustainability: A critical assessment of structural adjustment in Indonesia's forest and estate crop industries. CIFOR and WWF-International, Macroeconomics Program Office. Bogor, Indonesia.

Bawa, K.S. and Dayanandan, S. 1997. Socioeconomic factors and tropical deforestation. Nature 386: 562 563.

Bennett, C. 2002. Responsibility, accountability and village unity in village governance. In: C.J.P. Colfer and I.A.P. Resosudarmo (eds) Which way forward? Forest, policy and people in Indonesia. Resource for the future, Washington D.C.

Bennett, E.L. and Robinson, J.G. 2000. Hunting of wildlife in tropical forests - implications for biodiversity and forest peoples. Toward Environmentally and Socially Sustainable Development: Environment Department papers; no. 76 (Biodiversity series - Impact studies). The World Bank, Washington D.C.

Berkes, F. 1989. Common property resources: ecology and community-based sustainable development. Bellhaven Press, London.

Berkes, F. and Folke, C. 1998. Linking social and ecological systems: management practices and social mechanisms for building resilience. Cambridge University Press, New York.

Bessa-Gomes, C., Fernandes, M., Abreu, P., Castro, L., Ceia, H., Pinto, B. and Elisabete, A. 2002. Le lynx ibérique au Portugal : diverses approches face à un scénario de pré-extinction. In: G. Chapron and F. Moutou (eds), L'étude et la conservation des carnivores. SFEPM, Paris.

Birdlife. 2004. Indonesia clears the way for rainforest rescue. Birdlife International press release, 22-06-2004, Birdlife International, Cambridge.

Borner, M. 1978. Status and conservation of the Sumatran tiger. Carnivore 1: 97-102.

Boyce, M.S. and McDonald, L.L. 1999. Relating populations to habitats using resource selection functions. Trends in Ecology and Evolution 14: 268-272.

Brashares, J.S., Arcese, P. and Sam, M.K. 2001. Human demography and reserve size predict wildlife extinction in West Africa. Proceedings of the Royal Society of London Series B-Biological Sciences 268: 2473-2478.

Brown, K. and Pearce, D. 1994. The causes of tropical deforestation: the economic and statistical analysis of factors giving rise to the loss of tropical forests. University College London Press. London.

Bruner, A.G., Gullison, R.E., Rice, R.E. and da Fonseca, G.A.B. 2001. Effectiveness of parks in protecting tropical biodiversity. Science 291: 125-128.

Burgman, M. and Possingham, H.P. 2000. Population viability analysis for conservation: the good, the bad and the undescribed. In: A.G. Young and G.M. Clarke (eds), Genetics, demography and viability of fragmented populations. Cambridge University Press, London.

Burnham, K.P. and Anderson, D.R. 1998. Model selection and inference. Springer-Verlag, New York, USA.

Callaham, R. and Buckman, R. 1981. Some perspectives of forestry in the Philippines, Indonesia, Malaysia and Thailand. US Department of Agriculture/Forest Service, Washington, D.C.

Carbone, C., Christie, S., Conforti, K., Coulson, T., Franklin, N., Ginsberg, J.R., Griffiths, M., Holden, J., Kawanishi, K., Kinnaird, M., Laidlaw, R., Lynam, A., Macdonald, D.W., Martyr, D., McDougal, C., Nath, L., O'Brien, T., Seidensticker, J., Smith, D.J.L., Sunquist, M., Tilson, R. and Wan Shahruddin, W.N. 2001. The use of photographic rates to estimate densities of tigers and other cryptic mammals. Animal Conservation 4: 75-79.

Carbone, C. and Gittleman, J.L. 2002. A common rule for the scaling of carnivore density. Science 295: 2273-2276.

Carroll, C., Noss, R.F. and Paquet, P.C. 2001. Carnivores as focal species for conservation planning in the Rocky Mountain region. Ecological Applications 11: 961-980.

Carroll, C., Noss, R.F., Paquet, P.C. and Schumaker, N.H. 2003. Use of population viability analysis and reserve selection algorithms in regional conservation plans. Ecological Applications 13: 1773–1789.

Casson, A. 2000. The hesitant boom: Indonesia's oil palm sub-sector in an era of economic crisis and political change. Occasional Paper No. 29, CIFOR, Bogor, Indonesia.

Casson, A. 2003. Oil palm, soybeans and critical habitat loss. A Review prepared for WWF Forest Conversion Initiative, WWF

Castro, R., Gamez, L., Olson, N. and Tattenbach, F. 1998. The Costa Rican Experience with Market Instruments to Mitigate Climate Change and Conserve Biodiversity. FUNDECOR, MINAE, and the World Bank, San Jose, Costa Rica.

Caughley, G. 1994. Directions in conservation biology. Journal of Animal Ecology 63: 215-244.

Chapron, G. and Legendre, S. 2002. Some insights into snow leopard (*Uncia uncia*) demography by using stage-structured population models. In: T.M. McCarthy and J. Weltzin (eds), Contributed Papers to the Snow Leopard Survival Strategy Summit. International Snow Leopard Trust, Seattle, Washington D.C.

Chin, C.L.M. 2002. Hunting Patterns and Wildlife Densities in Primary and Production Forests in the Upper Baram, Sarawak. Unpublished M.Sc. thesis, University of Malaysia Sarawak, Malaysia.

Circle-Indonesia. 2002. Labour market conditions in Indonesia. Research paper, Center for Industrial Relation Research and Labor Education, Jakarta.

Clark, T.W., Peyton Curlee, A. and Reading, R.P. 1996. Crafting effective solutions to the large carnivore conservation problem. Conservation Biology 10: 940-948.

Clayton, L. 2004. Conservation of the Paguyaman Forest in Northern Sulawesi, Indonesia. Unpublished report to the Darwin Initiative for the Survival of Species, London.

Clayton, L.M. and Milner-Gulland, E.J. 2000. The trade in wildlife in North Sulawesi, Indonesia. In: J.G. Robinson and E.L. Bennett (eds), Hunting for Sustainability in tropical forests, pp.473-496. Columbia University Press, New York.

Cliff, A.D. and Ord, J.K. 1981. Spatial Processes – Models and Applications. Pion, London, UK.

Colchester, M. 1997. Salvaging nature: indigenous peoples and protected areas. In: K.B. Ghimire, and M.P. Pimbert (eds), Social Change and Conservation. Earthscan Publications, London.

Conservation International. 2000. Conservation coffee. CI, Washington D.C. http://www.celb.org/pressreleases/CCColombia.pdf [accessed 20th June 2003].

Conservation International. 2004. Forestrade Indonesia. http://www.conservation.org/xp/verdeventures/portfolio/forestrade_indonesia.xml [accessed 11th June 2004].

Contreras, A. and Vargas, M.T. 2002. Social, Environmental and Economic Dimensions of Forest Policy Reforms in Bolivia. Forest Trends and CIFOR, Washington D.C. and Bogor.

Contreras-Hermosilla, A. 2000. The underlying causes of forest decline. CIFOR Occasional Paper No. 30, CIFOR, Bogor, Indonesia.

Coulson, T., Mace, G.M., Hudson, E.J. and Possingham, H. 2001. The use and abuse of population viability analysis. Trends in Ecology and Evolution 16: 219-221.

Cropper, M. and Griffiths, C. 1994. The interaction of population-growth and environmental-quality. American Economic Review 84: 250-254.

Cropper, M., Griffiths, C. and Mani, M. 1997. Roads, population pressures, and deforestation in Thailand, 1976-89. Policy Research Working Paper 1726. The World Bank, Washington D.C.

Crooks, K.R. 2002. Relative sensitivities of mammalian carnivores to habitat fragmentation. Conservation Biology 16: 488-502.

Curran, S.R. and Agardy, T. 2002. Common property systems, migration and coastal ecosystems. Ambio 31: 303-305.

Dale, M.R.T. 1999. Spatial pattern analysis in plant ecology. Cambridge University Press, Cambridge, UK.

Damania, R., Stringer, R., Karanth, K.U. and Stith, B. 2003. The Economics of Protecting Tiger Populations: Linking Household Behavior to Poaching and Prey Depletion. Land Economics 79: 198-216.

Davies, G., Heydon, M., Leader-Williams, N., MacKinnon, J.R. and Newing, H.S. 2001. Large terrestrial mammals and logging. In: R.A. Fimbel, A. Grajal and J. Robinson (eds), Wildlife Logging Interactions in Tropical Forests. Columbia University Press, Columbia.

Demmer, J., Godoy, R., Wilkie, D., Overman, H., Taimur, M., Fernando, K., Gupta, R., McSweeney, K., Brokaw, N., Sriram, S. and Price, T. 2002. Do levels of income explain differences in game abundance? An empirical test in two Honduran villages. Biodiversity and Conservation 11: 1845-1868.

Departemen Kehutanan [Forestry Department]. 1995. Rencana Pengelolaan Taman Nasional Kerenci Seblat 1995 – 2019 [Kerinci Seblat National Park Management Plan]. Data, Proyeski, Dan Analisis book II. Indonesia.

Dinerstein, E. and Wikramanayake, E.D. 1993. Beyond 'hotspots': how to prioritise investments in biodiversity in the Indo-Pacific region. Conservation Biology 7: 53-67.

Dinerstein, E., Wikramanayake, E., Robinson, J., Karanth, U., Rabinowitz, A., Olson, D., Mathew, T., Hedao, P. and Connor, M. 1996. A framework for identifying high priority areas for the conservation of free-ranging tigers. Report to National Fish and Wildlife Foundation. Word Wildlife Fund-U.S., Washington D.C.

Dinerstein, E., Rijal, A., Bookbinder, M., Kattel, B. and Rajuria, A. 1999. Tigers as neighbours: efforts to promote local guardianship of endangered species in lowland Nepal. In: J. Seidensticker, S. Christie and P. Jackson (eds), Riding the Tiger: Tiger Conservation in Human-Dominated Landscapes, Cambridge University Press, Cambridge, UK.

Dirzo, R. and Garcia, M.C. 1992. Rates of deforestation in Los Tuxtlas, a neotropical area in Southeast Mexico. Conservation Biology 6: 84-90.

Dorner, P. and Thiesenhusen, W.C. 1992. Land tenure and deforestation: Interactions and environmental implications. Discussion Paper No. 34, United Nations Research Institute for Social Development (UNRISD), Geneva.

Duff, A.B., Hall, R.A. and Marsh, C.W. 1984. A survey of wildlife in and around a commercial tree plantation in Sabah. The Malaysian Forester 47: 197-213.

Eastman, J.R. 1999. Guide to GIS and image processing. Clark University, U.S.

EIA. 1999. Indonesia-UK Tropical Forest Management Programme (ITFMP), "Roundwood Supply and Demand in the Forest Sector in Indonesia", December 1999.

EIA-Telapak. 2003. Above the Law: Corruption, Collusion, Nepotism and the Fate of Indonesia's Forests. The Environmental Investigation Agency and Telapak report. London and Bogor.

Eisenberg, J.F. 1980. The density and biomass of tropical mammals. In: M. Soulé and B.A. Wilcox (eds), Conservation Biology: an Evolutionary-Ecological Perspective. Sinauer Associates, Sunderland, Massachusetts.

Else, J.G. 1991. Nonhuman primates as pests. In: H.O. Box (ed.), Primate responses to environmental change. Chapman and Hall, London.

European Commission. 2002. Indonesia country strategy paper 2002-2006. European Commission, Brussels.

FAO. 1990. Situation and Outlook of the Forestry Sector in Indonesia. Volume 1: issues, findings and opportunities. Ministry of Forestry, Government of Indonesia; Food and Agriculture Organization of the United Nations, Jakarta.

Faust, T. and Tilson, R. 1994. Estimating how many tigers are in Sumatra: a beginning. In: R.L. Tilson, K. Soemarna, W. Ramono, S. Lusli, K. Traylor-Holzer and U.S. Seal. (eds), Sumatran Tiger Population and Habitat Viability Analysis Report. Indonesian Directorate of Forest Protection and Nature Conservation and IUCN/SSC Conservation Breeding Specialist Group, Apple Valley, Minnesota.

Ferraro, P.J. 2001. Global habitat protection: Limitations of development interventions and a role for conservation performance payments. Conservation Biology 15: 990-1000.

Ferraro, P.J. and Kramer, R.A. 1997. Compensation and Economic Incentives: reducing pressures on protected areas. In: R. Kramer, C. van Schaik, and J. Johnson (eds), Last Stand: protected areas and the defence of tropical biodiversity. Oxford University Press, New York.

Ferraro, P.J. and Simpson, R.D. 2002. The cost-effectiveness of conservation payments. Land Economics 78: 339-353.

Ferraro, P.J. and Kiss, A. 2002. Ecology - Direct payments to conserve biodiversity. Science 298: 1718-1719.

Ferrière, R., Sarrazin, F., Legendre, S. and Baron, J.-P. 1996. Matrix population models applied to viability analysis and conservation: theory and practice using the ULM software. Acta Oecologica 17: 629-656.

Fiallo, E. and Jacobson, S.K. 1995. Local communities and protected areas: Attitudes of rural residents toward conservation and Machalilla National Park, Ecuador. Environmental Conservation 22: 241-249.

Folt, C.L. and Burns, C.W. 1999. Biological drivers of zooplankton patchiness. Trends in Ecology and Evolution 14: 300-305.

Franklin, N., Bastoni, Sriyanto, Siswomartono, D., Manansang, J. and Tilson, R. 1999. Last of the Indonesian tigers: a cause for optimism. In: J. Seidensticker, S.

Christie and P. Jackson (eds), Riding the Tiger: Tiger Conservation in Human-Dominated Landscapes, Cambridge University Press, Cambridge, UK.

FWI/GFW. 2002. The State of the Forest: Indonesia. Forest Watch Indonesia and Global Forest Watch, Bogor, Indonesia and Washington D.C.

Gallina, S., Mandujano, S. and Gonzalez-Romero, A. 1996. Conservation of mammalian biodiversity in coffee plantations of Central Veracruz, Mexico. Agroforestry Systems 33: 13-27.

Giao, P., Tuoc, D., Dung, V., Wikramanayake, E., Amato, G., Arctander, P. and MacKinnon, J. 1998. Description of *Muntiacus truongsonensis*, a new species of muntjac (Artiodactyla: Muntiacidae) from central Vietnam, and implications for conservation. Animal Conservation 1: 61-68.

Gibson, C.C. and Becker, C.D. 2000. A lack of institutional demand: why a strong local community in Western Ecuador fails to protect its forest. In: C.C. Gibson, M.A. McKean and E. Ostrom (eds), People and Forests: Communities, Institutions, and Governance (Politics, Science and the Environment). The MIT Press, Cambridge, MA.

Gillison, A., Liswanti N. and Arief Rachman, I. 1996. Rapid Ecological Assessment, Kerinci Seblat National Park buffer zone. Preliminary Report on Plant Ecology and Overview of Biodiversity Assessment. Working paper No. 14, CIFOR, Bogor, Indonesia.

Ginsberg, J.R. 2001. Setting priorities for carnivore conservation: what makes carnivores different? In: J.L. Gittleman (ed.), Carnivore Conservation. Cambridge University Press, Cambridge.

Ginsberg, J.R., Mace, G.M. and Albon, S. 1995. Local extinction in a small and declining population - wild dogs in the Serengeti. Proceedings of the Royal Society of London Series B-Biological Sciences 262: 221-228.

de Gloria, S.D. 1984. Spectral variability of Landsat-4 Thematic Mapper and Multispectral Scanner data for selected crop and forest cover types. IEEE Transactions on Geoscience and RemoteSensing 2: 303-311.

Green, G.M. and Sussman, R.W. 1990. Deforestation history of the eastern rain forest of Madagascar from satellite images. Science 248: 212-215.

Griffith, M. and van Schaik, C.P. 1993. The impact of human traffic on the abundance and activity periods of Sumatran rain forest wildlife. Conservation Biology 7: 623-626.

Griffiths, M. 1994. Population density of Sumatran tigers in Gunung Leuser National Park. In: R.L. Tilson, K. Soemarna, W. Ramono, S. Lusli, K. Traylor-Holzer and U.S. Seal. (eds), Sumatran Tiger Population and Habitat Viability Analysis Report. Indonesian Directorate of Forest Protection and Nature Conservation and IUCN/SSC Conservation Breeding Specialist Group, Apple Valley, Minnesota.

Gross, J.E., Kneeland, M.C., Reed, D.F. and Reich, R.M. 2002. GIS-based habitat models for mountain goats. Journal of Mammalogy 83: 218-228.

Guisan, A., Edwards, T.C. and Hastie, T. 2002. Generalized linear and generalized additive models in studies of species distributions: setting the scene. Ecological Modelling 157: 89-100.

Hall, S. 1985. The decline of the Italian wolf. Science 856: 90.

Hannah, L. 1992. African people, African parks: an evaluation of development incentives as a means of improving protected area conservation in Africa. USAID Biodiversity Support Program, Conservation International. Washington D.C.

Hartana, A.T. and Martyr, D. 2001 Pelestarian Harimau Sumatera Kerinci-Seblat [Kerinci Seblat Sumatran tiger protection programme]. Unpublished report. Kerinci Seblat National Park and Fauna & Flora International. Sungai Penuh, Indonesia.

Hastie, T.J. and Tibshirani, R.J. 1990. Generalized Additive Models. Chapman and Hall. London.

Heydon, M.J. and Bulloh, P. 1997. Mousedeer densities in a tropical rainforest: The impact of selective logging. Journal of Applied Ecology 34: 484-496.

Heydon, M.J. 1994. The ecology and management of rain-forest ungulates in Sabah, Malaysia: implications of forest disturbance. Unpublished Report, Institute of South-east Asian Biology, University of Aberdeen, Aberdeen, Scotland.

Hill, C.M. 1997. Crop-raiding by wild animals: the farmers' perspective in an agricultural community in western Uganda. International Journal of Pest Management 43: 77-84.

Hill, C.M. 2000. Conflict of interest between people and baboons: Crop raiding in Uganda. International Journal of Primatology 21: 299-315.

Hill, C., Osborn, F. and Plumptre, A.J. 2002. Human-wildlife conflict: identifying the problems and possible solutions. Albertine Rift Technical Report Series Vol. 1. Wildlife Conservation Society, New York.

Holden, J. 2001. Small cats in Kerinci Seblat National Park, Sumatra, Indonesia. Cat News 35: 11-14.

Holden, J. 2002[a]. Mammal List For Kerinci Seblat National Park (KSNP). http://www.kerinci.org/mammals_list.html [accessed 10 August 2003].

Holden, J. 2002[b]. Bird List in Kerinci Seblat National Park (TNKS). http://www.kerinci.org/bird_list.html [accessed 10 August 2003].

Holden, J., Yanuar, A. and Martyr, D.J. 2003. The Asian Tapir in Kerinci Seblat National Park, Sumatra: evidence collected through photo-trapping. Oryx 37: 34-40.

Holmes, D. 2001. Deforestation in Indonesia: A review of the situation in Sumatra, Kalimantan, and Sulawesi. The World Bank, Jakarta.

Holmes, D. 2002. Indonesia: Where have all the forests gone? Discussion paper. The World Bank, Jakarta.

Hooge, P.N. and Eichenlaub, B. 2000. Animal Movement extension to ArcView v.2.0. Alaska Science Center, Biological Science Office, U.S. Geological Survey, Anchorage, USA.

Horler, D.N.H. and Ahern, F.J. 1986. Forestry information content of Thematic Mapper data. International Journal of Remote Sensing 7: 405-428.

Ickes, K. and Williamson, G.B. 2000. Edge effects and ecological processes: are they on the same scale? Trends in Evolution and Ecology 15: 373.

IFFN. 2002. Fire Situation in Indonesia. International Forest Fire News (IFFN) report 26: 37-45.

Inamdar, A., de Jode, H., Lindsay, K. and Cobb, S. 1999. Capitalizing on Nature: Protected Area Management. Science 283: 1856-1857.

Infield, M. 1988. Attitudes of a rural community towards conservation and a local conservation area in Natal, South Africa. Biological Conservation 45: 21-46.

Ite, U.E. 1996. Community perceptions of the Cross River National Park, Nigeria. Environmental Conservation 23: 351-357.

IUCN. 2002. The 2002 IUCN Red List of Threatened Species. http://www.redlist.org [accessed 10 August 2003].

Jackson, P. 1999. The tiger in human consciousness. In: J. Seidensticker, S. Christie and P. Jackson (eds), Riding the Tiger: Tiger Conservation in Human-Dominated Landscapes, Cambridge University Press, Cambridge, UK.

Jakarta Post. 2003[1]. Haze covers Pekanbaru as rainy season nears end. The Jakarta Post, March 07. Indonesia.

Jakarta Post. 2003[2]. Thick haze threatens Riau, Aceh, Kalimantan. The Jakarta Post, June 07. Indonesia.

Jathanna, D. 2001. Density, Biomass and Habitat Occupancy of Ungulates in Bhadra Tiger Reserve, Karnataka, India. Final report submitted to Save the Tiger Fund of the National Fish and Wildlife Foundation, Washington, DC and the Exxon Mobil Corporation. Wildlife Institute of India, Bangalore, Dehradun, India.

Jennelle, C.S., Runge, M.C. and Mackenzie, D.I. 2002. The use of photographic rates to densities of tigers and other cryptic animals: a comment on misleading conclusions. Animal Conservation 5: 119-120.

Jepson, P., Jarvie, J.K., MacKinnon, K. and Monk, K.A. 2001. The end for Indonesia's lowland forests? Science 292: 859-861.

Jepson, P. and Whittaker, R.J. 2002. Ecoregions in context: a critique with special reference to Indonesia. Conservation Biology 16: 42-57.

Jhala, Y.V. 1993. Damage to sorghum crop by blackbuck. International Journal of Pest Management 39: 23-27.

Johnsingh, A.J.T. 1983. Large mammalian prey-predators in Bandipur. Journal of the Bombay Natural History Society 80: 1-57.

Johnson, C.J., Seip, D.R. and Boyce, M.S. 2004. A quantitative approach to conservation planning: using resource selection functions to map the distribution of mountain caribou at multiple spatial scales. Journal of Applied Ecology 41: 238-251.

Kaimowitz, D. 2003. The flawed wisdom of sustainable forest management. CIFOR news online.

(http://www.cifor.cgiar.org/scripts/default.asp?ref=publications/newsonline/32/wisdom.htm) [accessed 10 August 2003].

Kaimowitz, D. and Angelsen, A. 1998. Economic models of tropical deforestation: a review. CIFOR, Bogor, Indonesia.

Kaimowitz, D. and Ahmad, M. 2003. Sustaining RI forests needs strong measures. The Jakarta Post, January 20. Indonesia.

Karanth, K.U. 1991. Ecology and the management of the tiger in tropical Asia. In: N. Maruyama, B. Bobek, Y. Ono, W. Reglin, L. Bartos and R. Ratcliffe (eds), Wildlife Conservation: Present Trends and Perspectivesfor the 21st Century. Japan Wildlife Research Centre, Tokyo.

Karanth, K.U. 1995. Estimating tiger *Panthera tigris* populations from camera-trap data using capture-recapture models. Biological Conservation 71: 333-338.

Karanth, K.U. and Sunquist, M.E. 1995. Prey selection by tiger, leopard and dhole in tropical forests. Journal of Animal Ecology 64: 439-450.

Karanth, K.U. and Madhusudan, M.D. 1997. Avoiding paper tigers and saving real tigers: response to Saberwal. Conservation Biology 11: 818-820.

Karanth, K.S. and Nichols, J.D. 1998. Estimation of tiger densities in India using photographic captures and recaptures. Ecology 79: 2852-2862.

Karanth, K.S. and Stith, B.M. 1999. Prey depletion and tiger population viability. In: J. Seidensticker, S. Christie and P. Jackson (eds), Riding the Tiger: Tiger Conservation in Human-Dominated Landscapes, Cambridge University Press, Cambridge, UK.

Karanth, K.U. and Nichols, J.D. 2000. Ecological Status and Conservation of Tigers in India. Final Technical Report to the Division of International Conservation, US

Fish and Wildlife Service, Washington DC and Wildlife Conservation Society, New York. Centre for Wildlife Studies, Banglore, India.

Karanth, K.U. and Nichols, J.D. 2002. Monitoring tigers and their prey: A manual for researchers, managers and conservationists in tropical Asia. Centre for Wildlife Studies, India.

Karanth, K.U., Nichols, J.D., Seidenstricker, J., Dinerstein, E., Smith, J.L.D., McDougal, C., Johnsingh, A.J.T., Chundawat, R.S. and Thapar, V. 2003. Science deficiency in conservation practice: the monitoring of tiger populations in India. Animal Conservation 6: 141-146.

Kawanishi, K. 2002. Population status of tigers (*Panthera tigris*) in a primary rainforest of Peninsular Malaysia. Unpublished Ph.D. thesis. University of Florida. Florida.

Kenney, J.S., Smith, J.L.D., Starfield, A.M. and McDougal, C.W. 1995. The long-term effects of tiger poaching on population viability. Conservation Biology 9: 1127-1133.

Kerley, L.L., Goodrich, J.M., Miquelle, D.G., Smirnov, E.N., Quigley, H.B. and Hornocker, M.G. 2003. Effects of roads and human disturbance on Amur tigers. Conservation Biology 16: 97-108.

Khushwah, R.B.S. 1990. The past and present problem of managing tigers with regard to past stock lifting and man killing and later killing of tigers. Unpublished Dipl. Thesis. Wildlife Institute of India, India. In: K. Nowell and P. Jackson (eds), 1996, Wild Cats. Status Survey and Conservation Action Plan. IUCN/SSC Cat Specialist Group. IUCN, Gland, Switzerland.

Kinnaird, M.F., Sanderson, E.W., O'Brien, T.G., Wibisono, H.T. and Woolmer, G. 2003. Deforestation trends in a tropical landscape and implications for endangered large mammals. Conservation Biology 17: 245-257.

Kiss, A. 1990. Living with Wildlife. Wildlife Resource Management and Local Participation in Africa. World Bank Technical Paper Number 130, Africa Technical Department Series, Washington D.C.

Kompass. 1999. 17th November print, cited in Holmes, D. 2002. Indonesia: Where have all the forests gone? Discussion paper. The World Bank, Jakarta.

Kotwal, P.C. and Gopal, R. 1994. Food habits of tigers in Kanha National Park. Journal of Tropical Forestry 10: 139-143.

Kristiansson, H. 1985. Crop damage by wild boars in Central Sweden. Proceedings of the XVIIth Congress of the International Union of Game Biologists, Brussels.

Laing, S.P and Lindzey, F.G. 1993. Patterns of replacement of resident cougars in southern Utah. Journal of Mammology 74: 1056-1058.

Lambin, E. 1997. Modelling and monitoring land-cover change processes in tropical regions. Progress in Physical Geography 21: 375-393.

Lande, R. 1988. Genetics and demography in biological conservation. Science 241: 1455-1460.

Lauer, D.T., Morain, S.A. and Salomonson, V.V. 1997. The Landsat program: Its origins, evolution, and impacts. Photogrammetric Engineering and Remote Sensing 63: 831-838.

Laumonier, Y. 1994. The vegetation and tree flora of Kerinci-Seblat National Park, Sumatera. Tropical Biodiversity 2: 232-251.

Laurance, W.F. 1999. Reflections on the tropical deforestation crisis. Biological Conservation 91: 109-119.

Laurance, W.F. 2000. Cut and run: the dramatic rise of transnational logging in the tropics. Trends in Evolution and Ecology 15: 433-434.

Laurance, W.F., Albernaz, A.K.M., Schroth, G., Fearnside, P.M., Bergen, S., Venticinque, E.M. and da Costa, C. 2002. Predictors of deforestation in the Brazilian Amazon. Journal of Biogeography 29: 737-748.

Leader-Williams, N. and Albon, S.D. 1988. Allocation of resources for conservation. Nature 336: 533-535.

Leader-Williams, N. and Milner-Gulland, E.J. 1993. Policies for the enforcement of wildlife laws - the balance between detection and penalties in Luangwa Valley, Zambia. Conservation Biology 7: 611-617.

Leader-Williams, N. and Dublin, H. 2000. Charismatic megafauna as 'flagship species'. In: A. Entwistle and N. Dunstone (eds), Priorities for the Conservation of Mammalian Diversity: Has the Panda had its Day? Cambridge University Press, Cambridge, UK.

Lebar, F.M. 1972. Ethnic groups of insular Southeast Asia. Volume 1: Indonesia, Andaman Islands, and Madagascar. Human Relations Area Files Press, New Haven.

Lee, R.J. 2000. Impact of subsistence hunting in North Sulawesi, Indonesia, and conservation options. In: J.G. Robinson and E.L. Bennett (eds), Hunting for Sustainability in Tropical Forests, pp. 455-472. Columbia University Press, New York.

Legendre, S and Clobert, J. 1995. ULM: Unified Life Models, a software for conservation and evolutionary biologists. Journal of Applied Statistics 22: 817-834.

Lehmkuhl, J.F., Upreti, R.K. and Sharma, U.R. 1988. National-parks and local development - grasses and people in Royal Chitwan National Park, Nepal. Environmental Conservation 15: 143-148.

Levine, N. 2000. CrimeStat: A spatial statistics program for the analysis of crime incident locations. Ned Levine and Associates and the National Institute of Justice, Washington D.C.

Lewington, A. 1997. Atlas of the Rainforests. Raintree Steck-Vaughn Publishers, Austin, Texas.

Lillesand, T.M. and Kiefer, R.W. 1994. Remote sensing and image interpretation. John Willey and Sons, New York.

Lindzey, F.G., Vansickle, W.D., Laing, S.P. and Mecham, C.S. 1992. Cougar population response to manipulation in Southern Utah. Wildlife Society Bulletin 20: 224-227.

Lindzey, F.G., Vansickle, W.D., Ackerman, B.B., Barnhurst, D., Hemker, T.P. and Laing, S.P. 1994. Cougar population-dynamics in southern Utah. Journal of Wildlife Management 58: 619-624.

Linkie, M. 1999. Using a Geographic Information System (GIS) to assess the importance of lowland rainforest habitat in Tapan Valley, Kerinci Seblat National Park (KSNP): the tiger as a case study. Unpublished M.Sc. dissertation. University of Kent at Canterbury, UK.

Linkie, M. and Holden, J. 2002. The enigma surrounding tigers in Kerinci Seblat National Park. Cat News 36: 2-3.

Linkie, M. and Sibarani, P. 2002. Is it that difficult to stop illegal logging? The Jakarta Post, September 24. Indonesia.

Linkie, M., Martyr, D.J., Holden, J., Yanuar, A., Sugardjito, J., Hartana, A. and Leader-Williams, N. 2003. Habitat loss and poaching threaten the Sumatran tiger in Kerinci Seblat National Park, Sumatra. Oryx 37: 41-48.

Linkie, M., Smith, R.J. and Leader-Williams, N. 2004. Mapping and predicting deforestation patterns in the lowlands of Sumatra. Biodiversity and Conservation 13: 1809-1818.

Linnell, J.D.C., Swenson, J.E. and Andersen, R. 2000. Conservation of biodiversity in Scandinavian boreal forests: large carnivores as flagships, umbrellas, indicators, or keystones? Biodiversity and Conservation 9: 857-868.

Liu, J.G. 2001. Ecological Degradation in Protected Areas: The Case of Wolong Nature Reserve for Giant Pandas. Science 292: 98-101.

Liu, J.G., Daily, G.C., Ehrlich, P.R. and Luck, G.W. 2003. Effects of household dynamics on resource consumption and biodiversity. Nature 421: 530-533.

Loison, A. Strand, O. and Linnell, J.D.C. 2001. Effect of temporal variation in reproduction on models of population viability: a case study for remnant arctic fox (*Alopex lagopus*) populations in Scandinavia. Biological Conservation 97: 347-359.

Lynch, O.J. and Talbot, K. 1995. Balancing Acts: Community-Based Forest Management and National Law in Asia and the Pacific. World Resources Institute, Washington D.C.

Mace, G.M. and Balmford, A. 2000. Patterns and processes in contemporary mammalian extinction. In: A. Entwistle and N. Dunstone (eds), Priorities for the conservation of mammalian diversity: has the panda had its day? Cambridge University Press, Cambridge, UK.

MacKenzie, D.I., Nichols, J.D., Hines, J.E., Knutson, M.G. and Franklin A.B. 2003. Estimating site occupancy, colonization, and local extinction when a species is detected imperfectly. Ecology 84: 2200-2207.

MacKenzie, D.I., Nichols, J.D., Lachman, G.B., Droege, S., Royle, J.A. and Langtimm, C.A. 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83: 2248-2255.

MacKinnon, K. 2001. Integrated conservation and development projects – can they work? Parks 11.2:1-5.

MacKinnon, K., Mishra, H. and Mott, J. 1999. Reconciling the needs of conservation and local communities: Global Environmental Facility support for tiger conservation in India. In: J. Seidensticker, S. Christie and P. Jackson (eds), Riding the Tiger: Tiger Conservation in Human-Dominated Landscapes, Cambridge University Press, Cambridge, UK.

McCarthy, J.F. 2000. 'Wild logging': the rise and fall of logging networks and biodiversity conservation projects on Sumatra's rainforest frontier. Occasional Paper No. 31. CIFOR, Bogor, Indonesia.

McCracken, S.D., Brondizio, E.S., Nelson, D., Moran, E.F., Siqueira, A.D. and Rodriguez-Pedraza, C. 1999. Remote sensing and GIS at farm property level: Demography and deforestation in the Brazilian Amazon. Photogrammetric Engineering and Remote Sensing 65: 1311-1320.

Madhusan, M.D. 2003. Living amidst large wildlife: livestock and crop depredation by large mammals in the interior villages of Bhadra Tiger Reserve, South India. Environmental Management 31: 466-475.

Madhusudan, M.D. and Mishra, C. 2003. Why big, fierce animals are threatened: conserving large mammals in densely populated landscapes. In: M. Rangarajan and V. Saberwal (eds), Battles over nature: the science and politics of conservation in India. Permanent Black, New Delhi.

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

Madhusudan, M.D. and Karanth, K.U. 2000. Hunting for an answer: is local hunting compatible with wildlife conservation in India? In: J.G. Robinson and E.L. Bennett (eds), Hunting for sustainability in tropical forests. Columbia University Press, New York.

Maehr, D.S. 1997. The Florida panther: life and death of a vanishing carnivore. Island Press, Washington D.C.

Margules, C.R. and Pressey, R.L. 2000. Systematic conservation planning. Nature 405: 243-253.

Martin, R.B. and de Meulenaer, T. 1988. Survey of the status of the leopard (*Panthera pardus*) in sub-Saharan Africa. CITES Secretariat, Lausanne.

Mascarenhas, A. 1971. Agricultural vermin in Tanzania. In: S.H. Ominde (ed.), Studies in East African Geography and Development. University of California Press, Berkeley.

Mathai, M. 1999. Habitat occupancy across anthropogenic disturbances by sympatric ungulate species in Panna Tiger Reserve, M.P. M.Sc. dissertation. Saurashtra University, Rajkot.

Mazák, V. 1979. Der Tiger. Ziemsen, Wittenberg, Lutherstadt.

Mazák, V. 1981. *Panthera tigris*. Mammalian Species 152: 1-8.

Mazák, V. 1996. Der Tiger. Magdeburg, Westrap Wissenschaften.

Mertens, B. and Lambin, E.F. 1997. Spatial modelling of deforestation in southern Cameroon. Applied Geography 17: 143-162.

Millington, A.C., Walsh, S.J. and Osborne, P.E. 2001. GIS and Remote Sensing Applications in Biogeography and Ecology. The Kluwer International Series in Engineering and Computer Science. Kluwer Academic Publishers.

Milner-Gulland, E.J. and Leader-Williams, N. 1992. A model of incentives for the illegal exploitation of black rhinos and elephants: poaching pays in Luangwa Valley, Zambia. Journal of Applied Ecology 29: 388-401.

Ministry of Forestry. 2003. Report to stakeholders: current condition of forestry sector development. Statement on progress towards implementing sustainable forest management at the The Consultative Group on Indonesia (CGI) mid year meeting, 2 June 2003, "Mid-Year Review Meeting", Jakarta.

Miquelle, D.G., Smirnov, E.N., Merrill, T.W., Myslenkov, A.E., Quigley, H.B., Hornocker, M.G. and Schleyer, B. 1999[a]. Heirarchical spatial analysis of Amur tiger relationships to habitat and prey. In: J. Seidensticker, S. Christie and P. Jackson (eds), Riding the Tiger: Tiger Conservation in Human-Dominated Landscapes, Cambridge University Press, Cambridge, UK.

Miquelle, D.G., Merrill, T.W., Dunishenko, Y.M., Smirnov, E.N., Quigley, H.B., Pikunov, D.G. and Hornocker, M.G. 1999[b]. A habitat protection plan for the Amur tiger: developing political and ecological criteria for a viable land-use plan. In: J. Seidensticker, S. Christie and P. Jackson (eds), Riding the Tiger: Tiger Conservation in Human-Dominated Landscapes, Cambridge University Press, Cambridge, UK.

Miquelle, D.G. 2001. Counting Tigers in the Russian Far East: "How many are there?" versus "Is there a change?". Cat News 34: 7-9.

Mishra, H.R. 1982. Balancing human needs and conservation in Nepal's Royal Chitwan National Park. Ambio 11: 246-251.

Moe, S.R. and Wegge, P. 1997. The effects of cutting and burning on grass quality and axis deer (*Axis axis*) use of grassland in lowland Nepal. Journal of Tropical Ecology 13: 279-292.

Morrison, M.L., Marcot, B.G. and Mannan, R.W. 1992. Wildlife-Habitat Relationships - Concepts and Applications. The University of Wisconsin Press, Madison, WI.

Murcia C. 1995. Edge effects in fragmented forests - implications for conservation. Trends in Ecology and Evolution 10: 58-62.

Muttulingam, S. and Shen, S. 1999. Integrated Conservation Development Plan going forward with an ICDP. On-line Report. The World Bank.

http://lnweb18.worldbank.org/ESSD/essdext.nsf/48DocByUnid/C123ADBB8A7B1FA185256B8900566E04/$FILE/ICDPConditions.pdf [accessed 20 December 2002].

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B. and Kent, J. 2000. Biodiversity hotspots for conservation priorities. Nature 403: 853-858.

Naughton-Treves, L. 1997. Farming the forest edge: Vulnerable places and people around Kibale National Park, Uganda. The Geographical Review 87: 27-47.

Naughton-Treves L. 1998. Predicting patterns of crop damage by wildlife around Kibale National Park, Uganda. Conservation Biology 12: 156-168.

Naughton-Treves, L., Rose, R. and Treves, A. 1999. Social dimensions of HEC in Africa: a literature review and case studies from Uganda and Cameroon. Human Elephant Conflict Taskforce, IUCN. Gland, Switzerland.

Nelson, G.C., Harris, V. and Stone, S.W. 2001. Deforestation, land use, and property rights: empirical evidence from Darien, Panama. Land Economics 77: 187-205.

Nepal, S.K. and Weber, K.E. 1994. A buffer zone for biodiversity conservation - viability of the concept in Nepal Royal Chitwan-National-Park. Environmental Conservation 21: 333-341.

Newmark, W.D. 1996. Insularization of Tanzanian parks and the local extinction of large mammals. Conservation Biology 10: 1549-1556.

Newmark, W.D. and Leonard, N.L. 1991. The attitudes of local people toward Kilimanjaro National Park and Forest Reserve. In: W.D. Newmark (ed.), The Conservation of Mount Kilimanjaro. IUCN, Gland, Switzerland.

Newmark, W.D., Leonard, N.L., Sariko, H.I. and Gamassa, D.M. 1993. Conservation attitudes of local people living adjacent to five protected areas in Tanzania. Biological Conservation 63: 177-183.

Newmark, W.D., Manyanza, D.N., Gamassa, D.G.M. and Sariko, H.I. 1994. The conflict between wildlife and local people living adjacent to protected areas in Tanzania - human density as a predictor. Conservation Biology 8: 249-255.

Newmark, W.D. and Hough, J.L. 2000. Conserving Wildlife in Africa: Integrated Conservation and Development Projects and Beyond. Bioscience 50: 585-592.

Noss, A.J. and Cuellar, R.L. 2001. Community attitudes towards wildlife management in the Bolivian Chaco. Oryx 35: 292-300.

Noss, R.F., Quigley, H.B., Hornocker, M.G., Merril, T. and Paquet, P.C. 1996. Conservation biology and carnivore conservation in the Rocky Mountains. Conservation Biology 16: 949-963.

Nyhus, P., Sumianto and Tilson, R. 1999. The tiger-human dimension in southeast Sumatra. In: J. Seidensticker, S. Christie and P. Jackson (eds), Riding the Tiger: Tiger Conservation in Human-Dominated Landscapes, Cambridge University Press, Cambridge, UK.

Nyhus, P.J., Tilson, R. and Sumianto. 2000. Crop-raiding elephants and conservation implications at Way Kambas National Park, Sumatra, Indonesia. Oryx 34: 262-274.

O'Brien, T.G. and Kinnaird, M.F. 2003. Caffeine and conservation. Science 300: 587.

O'Brien, T.G., Wibisono, H.T. and Kinnaird, M.F. 2003. Crouching tigers, hidden prey: status of Sumatran tigers in the Bukit Barisan Selatan National Park, Sumatra, Indonesia. Animal Conservation 6: 131-139.

Ochoa-Gaona, S. 2001. Traditional land-use systems and patterns of forest fragmentation in the highlands of Chiapas, Mexico. Environmental Management 27: 571-586.

Oli, M.K., Taylor, I.R. and Rogers, M.E. 1994. Snow leopard (*Panthera uncia*) predation of livestock - an assessment of local perceptions in the Annapurna Conservation Area, Nepal. Biological Conservation 68: 63-68.

Osgood, D. 1994. Government failure and deforestation in Indonesia. In K. Brown and D.W. Pearce (eds), The Causes of Tropical Deforestation: the Economic and Statistical Analysis of Factors giving rise to the Loss of Tropical Forests. UCL Press, London.

Pacheco, P. 2002. The Implications of Decentralization in Forest Management: Municipalities and Local Forest Users in Lowland Bolivia. Paper presented at the World Resources Institute's Workshop on Decentralization and the Environment, February 18–22, 2002. Bellagio, Italy.

Pagiola, S. and Ruthenberg, I.M. 2002. Selling biodiversity in a coffee cup. In: S. Pagiola, J. Bishop and N. Landell-Mills (eds), Selling forest environmental services. Market-based mechanisms for conservation and development. Earthscan publications, London.

Panayotou, T. 1995. Environmental degradation at different stages of economic development. In: I. Ahmed and J.A. Doeleman (eds), Beyond Rio: The Environmental Crisis and Sustainable Livelihoods in the Third World. Macmillan Press Ltd, London, UK.

Parker, H.D. 1988. The unique qualities of a geographic information system: a commentary. Photogrammetric Engineering and Remote Sensing 54: 1547-1549.

Payne, J. and Francis, C.M. 1985. A field guide to the mammals of Borneo. The Sabah Society, Malaysia.

Pearce, J. and Ferrier, S. 2000. Evaluating the predictive performance of habitat models developed using logistic regression. Ecological Modelling 133: 225-245.

Peres, C.A. and Lake, I.R. 2003. Extent of nontimber resource extraction in tropical forests: Accessibility to game vertebrates by hunters in the Amazon basin. Conservation Biology 17: 521-535.

Peters, H.A. 2001. *Clidemia hirta* invasion at the Pasoh Forest Reserve: an unexpected plant invasion in an undisturbed tropical forest. Biotropica 33: 60-68.

Petit L.J. and Petit D.R. 2003. Evaluating the importance of human-modified lands for neotropical bird conservation. Conservation Biology 17: 687-694.

Pfaff, A.S. 1997. What drives deforestation in the Brazilian Amazon. Evidence from satellite and socioeconomic data. Policy Research Working Paper No. 1772. The World Bank, Washington D.C.

Pichón, F.J. 1997. Colonist land-allocation decisions, land use, and deforestation in the Ecuadorian Amazon frontier. Economic Development and Cultural Change 44: 127-164.

Poffenberger, M. and McGean, B. 1993. Communities and forest management in East Kalimantan: pathway to environmental stability. Southeast Asia Sustainable Forest Management Network, Research Network Report Number 3, August 1993. Center for Southeast Asia Studies International and Area Studies University of California, Berkeley.

Pollock, K.H. 1982. A capture–recapture design robust to unequal probability of capture. Journal of Wildlife Management 46:757–760.

Pollock, K.H., Nichols, J.D., Brownie, C. and Hines, J.E. 1990. Statistical inference for capture–recapture experiments. Wildlife Monographs 107.

Possingham, H.P., Lindenmayer, D.B. and Norton, T. W. 1993. A framework for the improved management of threatened species based on PVA. Pacific Conservation Biology 1: 39–45.

Pressey, R.L. 1994. *Ad hoc* reservations: Forward or backward steps in developing representative reserve systems? Conservation Biology 8: 662-668.

Pressey, R.L. and Taffs, K.H. 2001. Scheduling conservation action in production landscapes: priority areas in western New South Wales defined by irreplaceability and vulnerability to vegetation loss. Biological Conservation 100: 355-376.

Purvis, A., Gittleman, J.L., Cowlishaw, G. and Mace, G.M. 2000. Predicting extinction risk in declining species. Proceedings of the Royal Society of London Series B-Biological Sciences 267: 1947-1952.

Rabinowitz, A. 1993. Estimating the Indochinese tiger *Panthera tigris corbetti* population in Thailand. Biological Conservation 65: 213-217.

Rainforest Alliance. 2000. The conservation coffee campaign organizer's kit. Rainforest Alliance, New York.

Ramono, W.S. and Santiapillai, C. 1994. Conservation of Sumatran Tigers in Indonesia, In: R.L. Tilson, K. Soemarna, W. Ramono, S. Lusli, K. Traylor-Holzer and U.S. Seal. (eds), Sumatran Tiger Population and Habitat Viability Analysis Report. Indonesian Directorate of Forest Protection and Nature Conservation and IUCN/SSC Conservation Breeding Specialist Group, Apple Valley, Minnesota.

Randi E. 1993. Effects of fragmentation and isolation on genetic-variability of the Italian populations of wolf *Canis lupus* and brown bear *Ursus arctos*. Acta. Theriologica 38: 113-120.

Rao, K.S., Maikhuri, R.K., Nautiyal, S. and Saxena, K.G. 2002. Crop damage and livestock depredation by wildlife: a case study from Nanda Devi Biosphere Reserve, India. Journal of Environmental Management 66: 317-327.

Redford, K.H. 1992. The empty forest. BioScience 42: 412-422.

Revilla, E., Palomares, F. and Delibes, M. 2001. Edge-core effects and the effectiveness of traditional reserves in conservation: Eurasian badgers in Doñana National Park. Conservation Biology 15: 148-158.

Riau Post. 1999. 16th November print, cited in Holmes, D. 2002. Indonesia: Where have all the forests gone? Discussion paper. The World Bank, Jakarta.

Roberts, D.L., Cooper, R.J. and Petit, L.J. 2000. Flock characteristics of ant-following birds in premontane moist forest and coffee agroecosystems. Ecological Applications 10: 1414-1425.

Robinson, J.G., Redford, K.H. and Bennett, E.L. 1999. Wildlife harvest in logged tropical forest. Science 284: 595-596.

Robinson, J.G. and Bennett, E.L. 2000. Hunting for Sustainability in Tropical Forests. New York, Columbia University Press.

Rushton, S.P., Ormerod, S.J. and Kerby, G. 2004. New paradigms for modelling species distributions? Journal of Applied Ecology 41: 193-200.

Sader, S.A. and Joyce, A.T. 1988. Deforestation rates and trends in Costa Rica, 1940 to 1983. Biotropica 20: 11-19.

Sader, S.A. 1989. Multispectral and seasonal characteristics of northern hardwood and boreal forest types in Maine, Image Processing 89, University of Nevada-Reno, ASPRS, Bethesda, MD.

Saj, T.L., Sicotte, P. and Paterson, J.D. 2001. The conflict between vervet monkeys and farmers at the forest edge in Entebbe, Uganda. African Journal of Ecology 39: 195-199.

Sánchez-Azofeifa, G.A., Quesada-Mateo, C., Gonzalez-Quesada, P., Dayanandan, S. and Bawa, K.S. 1999. Protected areas and conservation of biodiversity in the tropics. Conservation Biology 13: 407-411.

Sanchez-Azofella, G.A., Harriss, R.C. and Stoke, D.L. 2001. Deforestation in Costa Rica: a quantitative analysis using remote sensing imagery. Biotropica 33: 378-384.

Sánchez-Azofeifa, G.A., Daily, G.C., Pfaff, A.S.P. and Busch, C. 2003. Integrity and isolation of Costa Rica's national parks and biological reserves: examining the dynamics of land-cover change. Biological Conservation 109: 123-135.

Sankar, K. 1994. The ecology of three large sympatric herbivores (chital, sambar and nilgai) with special reference for reserve management in Sariska Tiger Reserve, Rajasthan. Unpublished Ph.D. thesis. University of Rajasthan, Jaipur.

Santiapillai, C. and Widodo, R.S. 1985. On the Status of the Tiger (*Panthera tigris sumatrae* Pocock, 1829) in Sumatra. Tigerpaper 12: 23-29.

Sayer, J.A., Iremonger, S., Ishwaran, N. and Thorsell, J. 2000. The world heritage convention as a mechanism for tropical forest conservation. In: H.D. Thulstrup (ed.), World Natural Heritage and the Local Community. UNESCO World Heritage and CIFOR, Bogor, Indonesia.

Saxena, N.C. 1988. Social forestry in U.P. Hills. International Centre for Integrated Mountain Development, ICIMOD, Kathmandu, Nepal.

Schadt, S., Revilla, E., Wiegand, T., Knauer, F., Kaczensky, P., Breitenmoser, U., Bufka, L., Červený, J., Koubek, P., Huber, T., Staniša, C. and Trepl, L. 2002. Assessing the suitability of central European landscapes for the reintroduction of Eurasian lynx. Journal of Applied Ecology 39: 189-203.

Shaffer, M. 1987. Minimum viable populations: coping with uncertainty. In: M.E. Soulé (ed.), Viable Populations for Conservation. Cambridge University Press, Cambridge, UK.

Seidensticker, J. 1986. Large carnivores and the consequences of habitat insularization: ecology and conservation of tigers. In: S.D. Miller and D.D. Everett (eds), Indonesia and Bangladesh. Cats of the World: Biology, Conservation and Management. National Wildlife Federation, Washington D.C.

Seidensticker, J., Tilson R.L. and Seal, U.S. 1987. Bearing witness: Observations on the extinction of *Panthera tigris balica* and *Panthera tigris sondaica*. In: R.L. Tilson and U.S. Seal (eds), Tigers of the World: The Biology, Biopolitics, Management and Conservation of an Endangered Species. NJ. Noyes Publications, Park Ridge.

Sekhar, N.U. 1998. Crop and livestock depredation caused by wild animals in protected areas: the case of Sariska Tiger Reserve, Rajasthan, India. Environmental Conservation 25: 160-171.

Shively, G.E. 1997. Poverty, technology, and wildlife hunting in Palawan. Environmental Conservation 24: 57-63.

Siebert, S.F. 1989. The dilemma of a dwindling resource: rattan in Kerinci, Sumatra. Principes 33: 79-87.

Siegert, F., Ruecker, G., Hinrichs, A. and Hoffmann, A.A. 2001. Increased damage from fires in logged forests during droughts caused by El Nino. Nature 414: 437-440.

Siex, K.S. and Struhsaker, T.T. 1999. Colobus monkeys and coconuts: a study of perceived human-wildlife conflicts. Journal of Applied Ecology 36: 1009-1020.

Sinclair, A.R.E., Ludwig, D. and Clark, C.W. 2000. Conservation in the Real World. Science 289: 1875.

Sitati, N.W., Walpole, M.J., Smith, R.J. and Leader-Williams, N. 2003. Predicting spatial aspects of human-elephant conflict. Journal of Applied Ecology 40: 667-677.

Smith, J.L.D. 1993. The role of dispersal in structuring the Chitwan tiger population. Behaviour 124: 165-195.

Smith, J.L.D., Wemmer, C. and Mishra, H.R.A. 1987. Tiger geographic information system: the first step in a global conservation strategy. In: R.L. Tilson and U.S. Seal (eds), Tigers of the World: The Biology, Biopolitics, Management and Conservation of an Endangered Species. NJ. Noyes Publications, Park Ridge.

Smith, J.L.D. and McDougal, C. 1991. The contribution of variance in lifetime reproduction to effective population-size in tigers. Conservation Biology 5: 484-490.

Smith, J.L.D., Ahern, S.C. and McDougal, C. 1998. Landscape analysis of tiger distribution and habitat quality in Nepal. Conservation Biology 12: 1338 –1346.

Smith, R.J. and Kasiki, S.M. 2000. A spatial analysis of human-elephant conflict in the Tsavo ecosystem, Kenya. Human Elephant Conflict Taskforce, IUCN. Gland, Switzerland.

Soulé, M.E. 1980. Thresholds for survival: maintaining fitness and evolutionary potential. In: M.E. Soulé and B.A. Wilcox (eds), Conservation Biology: the Science of Scarcity and Diversity. Sinauer Associates, Sunderland, Massachusetts.

Southgate, D. 1992. Policies contributing to agricultural colonisation of Latin America's tropical forests. In: Sharma N. (ed.), Managing the World's Forests: Looking for Balance Between Conservation and Development. Kendall/Hunt Publishing Company, Dubuque.

Southgate, D., Sierra, R. and Brown, L. 1991 The causes of tropical deforestation in Ecuador: a statistical analysis. World Development 19: 1145-1151.

Starfield, AM. 1997. A pragmatic approach to modeling for wildlife management. J. Wildlife Management 61: 261-270.

Stockdale, M.C. and Ambrose, B. 1996. Mapping and NTFP inventory: Participatory assessment methods for forest-dwelling communities in East Kalimantan, Indonesia. In: J. Carter (ed.), Recent approaches to participatory forest resource assessment. Overseas Development Institute, Regent's College, London.

Stolle, F. and Tomich, T.P. 1999. The 1997-1998 fire event. Nature and Resources 35: 22-30.

Straede, S. and Helles, F. 2000. Park-people conflict resolution in Royal Chitwan National Park, Nepal: buying time at high cost? Environmental Conservation 27: 368-381.

Studsrød, J.E. and Wegge, P. 1995. Park-people relationships: the case of damage caused by Park animals around the Royal Bardia National Park, Nepal. Environmental Conservation 22: 133-142.

Sunderlin, W.D. 1998. Between danger and opportunity: Indonesia's forests in an era of economic crisis and political change. CIFOR. (September 11, 1998. http://www.cgiar.org/cifor). [accessed July 2002].

Sunderlin, W.D. 1999. The effects of economic crisis and political change on Indonesia's forest sector, 1997-99. 15 November 1999. http://www.cgiar.org/cifor/research/projects/effect-crisis.html. [accessed June 2001].

Sunderlin, W.D. and Resosudarmo, I.A.P. 1996. Rates and causes of deforestation in Indonesia: Towards a resolution of the ambiguities. Occasional Paper No. 9. CIFOR, Bogor, Indonesia.

Sunquist, M.E. 1981. The social organisation of tigers (*Panthera tigris*) in Royal Chitawan National Park, Nepal. Smithsonian Contributions to Zoology 336: 1-97.

Sutedjo, E.B. and Suryadi, S. 1997. Government and non-government organization cooperation in the development of a community forestry program, case study: Protected forest of Sesaot, West Lombok. Paper Presented at an International Seminar - "Community Forestry at a Crossroads: Reflections and Future Directions in the Development of Community Forestry", 17-19 July 1997, RECOFTC, Bangkok, Thailand.

Suyanto, S., Dennis, R.A., Ruchiat, Y., Kurniawan, I., Stolle, F., Maus, P. and Applegate, G. 2000. The underlying causes and impacts of fires in South-east Asia. Site 2. Menggala, Lampung Province, Indonesia. Project Report. CIFOR, ICRAF and USFS, Bogor, Indonesia.

Suyanto, S. and Otsuka, K. 2001. From deforestation of agroforests in customary land tenure areas of Sumatra. Asian Economic Journal 15: 1-18.

Swets, J.A. 1988. Measuring the accuracy of diagnostic systems. Science 240: 1285-1292.

Tacconi, L. 2003. Fires in Indonesia: cause, costs and policy implications. Occasional Paper No. 38. CIFOR, Bogor, Indonesia.

Talwar, R. 1999. Insurance Policy for Tigers. In: Three Years and Beyond WWF TCP. WWF-India, New Delhi.

Terborgh, J. 1988. The big things that run the world - A sequel to E.O. Wilson. Conservation Biology 2: 402-403.

Thurber, J.M., Peterson, R.O., Drummer, T.D. and Thomasma, S.A. 1994. Gray wolf response to refuge boundaries and roads in Alaska. Wildlife Society Bulletin 22: 61-68.

Tilson, R. and Nyhus, P. 1998. Keeping problem tigers from becoming a problem species. Conservation Biology 12: 261-262.

Tilson, R.L., Soemarna, K., Ramono, W., Lusli, S., Traylor-Holzer, K. and Seal, U.S. 1994. Sumatran Tiger Population and Habitat Viability Analysis Report. Indonesian Directorate of Forest Protection and Nature Conservation and IUCN/SSC Conservation Breeding Specialist Group, Apple Valley, Minnesota.

du Toit, J.T., Walker, B.H. and Campbell, B.M. 2004. Conserving tropical nature: current challenges for ecologists. Trends in Ecology and Evolution 19:12-17.

Tomich, T.P. and van Noordwijk, M. 1995. What drives deforestation in Sumatra? Paper presented at Regional Symposium on "Montane Mainland Southeast Asia in Transition", 13-16 November, Chiang Mai, Thailand.

Trejo, I. and Dirzo, R. 2000. Deforestation of seasonally dry tropical forest: a national and local analysis in Mexico. Biological Conservation 94: 133-142.

Tyre, A.J., Tenhumberg, B., Field, S.A., Niejalke, D., Parris, K. and Possingham, H.P. 2003. Improving precision and reducing bias in biological surveys by estimating false negative error rates in presence-absence data. Ecological Applications 13: 1790-1801.

USDA. 1978. Soil Taxonomy. Agriculture Handbook no. 436. USDA, Soil Conservation Service. Washington D.C.

de Vaus D.A. 1999. Surveys in social research. 4th edition. UCL Press, London.

Vina, A. and Cavelier, J. 1999. Deforestation rates (1938-1988) of tropical lowland forests on the Andean foothills of Columbia. Biotropica 31: 31-36.

Vincent, J.R. and Rozali, M.A. 1997. Environment and development in a resource-rich economy. Malaysia under the New Economic Policy. HIID Institute of Strategic and International Studies, Harvard University Press, Cambridge, Massachusetts.

Vucetich, J.A. and Creel, S. 1999. Ecological interactions, social organization, and extinction risk in African wild dogs. Conservation Biology 13: 1172-1182.

Wakker, E. 1998. Lipsticks from the rainforest: Palm oil, crisis and forest loss in Indonesia: the role of Germany. WWF, A Forest Campaign Project of WWF-Germany in collaboration with WWF-Indonesia. Indonesia.

WALHI. 2000. WALHI of South Sumatra calls for the closure of PT Barisan Tropical Mining / Laverton gold's mine. Press release January 22. WALHI, South Sumatra.

Walijatun, D. and Grant, C.A. 1996. Land Registration Reform in Indonesia. International Conference on Land Tenure and Administration. Orlando, Florida.

Wells, M., Brandon, K. and Hannah, L. 1992. People and parks: linking protected area management with local communities. The World Bank, Washington D.C.

Wells, M., Guggenheim, S., Khan, A., Wardojo, W. and Jepson, P. 1999. Investing in Biodiversity. A Review of Indonesia's Integrated Conservation and Development Projects. The World Bank, Washington, D.C.

Whitmore, T.C. 1984. A new vegetation map of Malesia at scale 1:5 million. Journal of Biogeography 11: 461-471.

Whitmore, T.C. and Sayer, J.A. 1992. Tropical Deforestation and Species Extinction. Chapman and Hall, London.

Whitten, A.J., Damanik, S.J., Anwar, J. and Hisyam, N. 1984. The Ecology of Sumatra. Gadjah Mada University Press, Yogyakarta.

Whitten, T., Holmes, D. and MacKinnon, K. 2001. Conservation biology: a displacement behavior for academia? Conservation Biology 15: 1-3.

Wielgus, R.B. Sarrazin, F. Ferrière, R. and Clobert, J. 2000. Possible negative effects of adult male mortality on brown bear population growth and persistence. Biological Conservation 98: 293-303.

Wikramanayake, E.D., Dinerstein, E., Robinson, G., Karanth, K.U., Rabinowitz, A.R., Olson, D., Matthew, T., Hedao, P., Connor, M., Hemley, G. and Bolze, D. 1998. An ecology-based method of defining priorities for large mammal conservation: the tiger as a case study. Conservation Biology 12: 865-878.

WildAid. 2002. Surviving together, Khao Yai Conservation Project. Khao Yai Conservation Project Annual Report, January 2002. Bangkok, Thailand.

Wilkie, D., Shaw, E., Rotberg, F., Morelli, G. and Auzel, P. 2000. Roads, development, and conservation in the Congo basin. Conservation Biology 14: 1614-1622.

Williams, P.H., Gaston, K.J. and Humphries, C.J. 1997. Mapping biodiversity value world-wide: combining higher-taxon richness from different groups. Proceedings of the Royal Society of London Series B-Biological Sciences 264: 141-148.

Woodroffe, R. and Ginsberg, J.R. 1998. Edge effects and the extinction of populations inside protected areas. Science 280: 2126-2128.

World Bank. 1990. Indonesia: Sustainable Development of Forests, Land, and Water. The World Bank, Washington D.C.

World Bank. 1993. Indonesia Production Forestry: Achieving Sustainability and Competitiveness. Draft Working Paper. World Bank Report 11758-IND. The World Bank, Washington D.C.

World Bank. 1995. The Economics of Long Term Management of Indonesia's Natural Forest. The World Bank, Washington D.C.

World Bank. 1996[a]. Kerinci Seblat Integrated Conservation and Development. GEF project document, East Asia and Pacific Country Department III, Agriculture Operations Division. The World Bank, Washington D.C.

World Bank. 1996[b]. India ecodevelopment project. GEF project document, South Asia Department II, Agriculture and Water Division. The World Bank, Washington D.C.

World Bank. 2000. Indonesia The Challenges of World Bank Involvement in Forests Evaluation Country Case Study Series. The World Bank, Washington D.C.

World Bank. 2003. Working group statement on forestry. The Consultative Group on Indonesia (CGI) mid year meeting, 2 June 2003, Jakarta. "Mid-Year Review Meeting". Unpublished document. The World Bank, Jakarta, Indonesia.

World Bank. 2003. Kerinci Seblat Integrated Conservation and Development Project. Implementation completion report. The World Bank, Jakarta, Indonesia.

de Wulf, R., Supomo, D. and Rauf, K. 1981. Kerinci-Seblat proposed National Park preliminary management plan 1982-1987. Directorate General of Forest Protection and Nature Conservation, Bogor.

Wunder, S. 2001. Poverty alleviation and tropical forests – what scope for synergies? World Development 29: 1817-1833.

WWF. 1989. Project 3769: progress report, July-September 1989. Unpublished report. WWF-Indonesia, Bogor.

WWF. 1999. Village facilitating NGO Kerinci Seblat-IDCP. WWF-Indonesia quarterly report Oct-Dec 1999. WWF-Indonesia Programme, Sungai Penuh, Indonesia.

WWF. 2001. Tiger Conservation Enhancement in Bhutan. Final WWF Bhutan Program report. Bhutan.

WWF. 2003. The Tesso Nilo Conservation Landscape, Sumatra, Indonesia. Unpublished report, 5 June 2003. WWF Indonesia ARETAS Riau Programme, Sumatra, Indonesia.

WWF and IUCN. 1994-1995. Centres of plant diversity. A guide and strategy for their conservation. Vol 2. IUCN publications Unit, Cambridge, UK.

Appendix 1: Household questionnaire survey (with English translations)

1. INFORMASI UMUM RUMAH TANGGA (*HOUSEHOLD, HH, INFORMATION*)

1.1 Tanggal (*Date*):

1.2 Lokasi (*Location*):

1.3 Informasi dasar rumahtangga dan keluarga (*HH and family information*)

1.3.1	1.3.2	1.3.3	1.3.4	1.3.5	1.3.6	1.3.7	
Nomer ang-gota rumah-tangga (*Number of HH members*)	Nama (*Name*) (1 untuk KK, kalau ada orang lain yang ikut wwncra ini, beri * pada nama mereka – *1 for HH head, if there are other HH persons give * for their name*)	Hubungan Keluarga (*Family connection*) (lihat kode [A] – *see code* [A])	Jenis Kelamin (*sex*) 0=prmpn (*female*) 1=laki2 (*male*)	Umur (*Age*) 888 = RTT (*not known*)	Pendidikan tertinggi (*Highest level of education*) (lihat kode [B] - *see code* [B])	Kegiatan selama satu tahun ini sebelum tgl. (*activities over the past year*) (lihat kode [C] - *see code* [C])	
						a. Utama (*main*)	b. Sampingan (*secondary*) (bisa lebih dari pada 1 – *can have more than one*)
1. (KK)							
2.							
3.							
4.							
5.							
6.							

[A] Hubungan keluarga (*Family connection*): 1 = Kepala rumah tangga (*HH head*); 2 = Isteri / suami (*wife / husband*); 3 = Anak kandung (*own child*); 4 = Orang tua (*parents*); 5 = Mertua (*parent in law*); 6 = Menantu (*child in law*); 7 = Cucu (*grandchild*); 8 = Anak angkat (*not own child*); 9 = Lain (*other*)

[B] Pendidikan tertinggi (*Highest level of education*) (* kalau orang masih belajar – *if person still studying*): 1=sekolah dasar (SD – *Primary school*); 2=SMP (*Junior high school*); 3.=SMU (*Senior high school*); 4.=Perguruan tinggi (*Highest teaching*); 5.=Diploma (*Diploma*); 6.=Kursus keahlian (*Specialist course*) [A=tambahan kalau sekolah agama (*note if religious school*)/ S=tambahan kalau sekolah swasta (*note if private school*)]

[C] Kegiatan utama/sampingan (*Main/secondary activities*):1 = Bertani (*Farmer*); 2 = Berdagang (*Trader*); 3 = Mengumpulkan hasil hutan (*forest product collector*); 4 = Tukang (*handicrafts*); 5 = Kegiatan umum rumah-tangga (*HH activities*); 6 = Bekerja di pemerintahan (*Governmental work*); 7 = Wiraswasta (*private business*); 8 = Bersekolah (*Teacher*); 9 = Tidak bekerja (selain sakit atau cacat) (*Does not work – ill or disabled*); 88 = Responden tidak tahu (*respondent did not know*); 99 = tidak berlaku (orang umur di bawah 12 tahun) (does not work – person under 12 years)

1.4 Kekayaan yang diperoleh / dijual [Semua jenis ditanyakan] (*Possessions owned / sold [all answers included]*)

1.4.1	1.4.2	1.4.3	1.4.4	1.4.5
Jenis kekayaan (*Type of possession*)	Punya (*Own*)? (Berapa – *How many*?) 1 = ya, 1 2=ya, 2…. 0 = tidak (*No*)	Kapan diperoleh (*When did this occur*)? 1 = sebelum pertengahan '01 (*before 2001*) 2 = setelah pertengahan '01 (*During 2001*) 88 = TB (tidak punya – *Do not own*)	Pernah punya tetapi sudah dijual (*Once owned but sold*)? 1 = ya dijual (*sold*) 0 = tidak dijual (*Not sold*) 88 = TB (tidak pernah punya – *Never owned*)	Kalau dijual kapan (*If sold, when*)? 1 = sebelum pertengahan '01 (*before 2001*) 2 = setelah pertengahan '01 (*During 2001*) 88 = TB (tidak punya atau tidak dijual – *Never owned or not sold*)
Kendaraan (*Vehicle*)				
Sepeda (*motorbike*)				
Sepada (*pushbike*)				
Pedati/Bendi (*Cart*)				
Lain (*Other*)				
Peralatan pert. (*Farm equipment*)				
Bajak (*Plough*)				
Gergaji kayu (*Saw*)				
Cinsaw (*Chainsaw*)				
Penyemprot (*Spray*)				
Lain (*Other*)				
Ternak (*Livestock*)				
Kerbau (*Buffalo*)				
Sapi (*Cow*)				
Kuda (*Horse*)				
Kambing (*Goat*)				
Ayam (*Chicken*)				
Lain (*Other*)				
Umum (*House*)				
Parabola (*Satellite dish*)				
Televisi (*TV*)				
Alat pertukangan (*Work tools*)				
Lain (*Other*)				

Jika ada lebih dari satu kekayaan yang pernah dimiliki di dalam satu kategori (misalnya responden pernah memiliki dua sepeda motor), tanyakan kekayaan yang terakhir diperoleh. (*If there are more than 1 types of wealth for a single category – for example, a respondent once had two motorbikes – then use the most recent response*)

1.5. Keadaan rumah (*Housing condition*)

1.5.1	1.5.2	1.5.3	1.5.4	1.5.5	1.5.6
Luas bangunan (m^2) – (*Construction area*)	Atap rumah (*Roof*)	Tembok/ Dinding (*wall*)	Lantai (*Floor*)	Listrik desa (PLTS, PLTMH, Generator) (*Electricity supply*)	WC / Kamar Mandi (*Toilet / bathroom*)
1. <30 2. 30-59 3. 60-99 4. 100-120 5. >120	1. Genteng (*Tiles*) 2. Seng (*Iron*) 3. Sirap (*Ironwood*) 4. Rumbia (*Palm leaf*)	1. Permanen (*Permanent*) 2. Semi-perm. (*semi-perm.*) 3. Papan / kayu (*Wooden*) 4. Bilik / bamboo (*Bamboo*) 5. Kulit kayu (*Tree bark*)	1. Ulin () 2. Keramik (*Ceramic*) 3. Semen / ubin (*Cement*) 4. Kayu papan (*Wooden*) 5. Tanah (*Earth*)	1. Ya (*Yes*) 0. Tidak (*No*)	1. Ya (*Yes*) 0. Tidak (*No*)

Kapan ganti: Pakai bulan/tahun (*When was it replaced: use month/year*)

2. SISTEM PERTANIAN (*FARMING SYSTEMS AND PRACTICES*)

2.1 Sejak kapan ladang ini dibuka? (*How many years on this farm?*)

2.2 Di ladang ini ada jenis tanaman utama apa dan untuk apa? (*What are the main crops grown here and for what purpose?)*

2.2.1		2.2.2
Jenis tanaman (*Crop type*)		Untuk apa (*Purpose*)
1.		
2.		
3.		
4.		
5.		
6.		

2.3 Apa faktor-faktor yang membatasi suatu keberhasilan dari usaha pertanian? *(What are the limiting factors in agriculture success?)* Urutkan sesuai dengan faktor-faktor yang paling berpengaruh. *[Baca semua alasan dan lingkari faktor-faktor yang terjadi, lalu urutkan dari yang paling batasi duhulu]*

2.3.1 Tidak cukupnya lahan (*Not enough space*)
2.3.2 Produktivitas yang rendah (*Low productivity*)
2.3.3 Terbatasnya sarana perhubungan (*Limited means to communication*)
2.3.4 Terbatasnya ketersediaan informasi tentang budidaya pertanian
2.3.5 Turunnya harga pasar komoditas (*Crop prices decreasing*)
2.3.6 Bencana alam (*Natural disasters*)
2.3.7 Terbatasnya hubungan ke pasar (*Limited connection to the markets*)
2.3.8 Gangguan binatang (*Disturbance by animals*)
2.3.9 Serangan hama (*Plant disease attack*)
2.3.10 Tanah tidak cocok dengan tanaman (*Soil not suitable for the crops*)
2.3.11 Banjir (*Flooding*)
2.3.12 Tanah erosi (*Soil erosion*)
2.3.13 Lain (*Other*)

2.4 Bagaimana faktor-faktor yang membatasi tersebut di atas dapat diperkecil untuk memperbaiki situasi pertanian? *(How could these limiting factors be corrected, so the situation for farmers improves?)*

2.5 Apakah Bapak/Ibu pernah mencoba untuk menanam tanaman yang tidak biasanya ditanam di daerah ini? (*Have you ever tried planting crop not traditionally planted in this area?)* Ya (*Yes*) Tidak (*No*)

2.5.1 Kalau pernah mencoba isi tabel dibawah ini (*If yes, then fill in the details below*)

2.5.1.1	2.5.1.2	2.5.1.3	2.5.1.4	2.5.1.5

Jenis (*Type*)	Kapan dicoba (*When tried*)	Tingkat keberhasilan (Level of success)	Apakah dilanjutkan (Any difficulties)	Mengapa/mengapa tdk (What difficulties)

2.6 Apakah bersikapmu pernyataan-pernyataan ini (*To the following statements do you think*):

Ya, setuju (*I agree*) / Tidak pasti (*Unsure*) / Tidak setuju (*Disagree*)

2.6.1 Memotong hutannya berakibat tanah soal-soal (*Cutting the forest increases soil erosion*)

2.6.2 Memotong hutannya tidak berakibat banjir di ladang (*Cutting the forest does not increases flooding*)

2.6.3 Menotong hutannya berakibat tananam penyakit (*Cutting the forest increases disease/insect crop attacks*)

3. PERTANIAN DAN KSNP (FARMING AND KSNP)

3.1 Apa anda tahu lokasi batas TNKS untuk daerah ini? (*Do you know the location of the KSNP boundary for this area?*)

3.2 Apa anda mau membuka lebih banyak ladang? (*Do you want to increase the amount of farmland you have?*) Ya (*Yes*) Tidak (*No*)

3.3 Kalau ya, berapa hektar lebih? (*If yes, how many hectres more?*)

3.4 Siapa yang merancang KSNP? (*Who designed KSNP?*)

3.5 Apakah Bapak/Ibu pernah mengatahui personel KSNP yang datang di desa ini? (*Have you ever meet a personnel from KSNP that came to this village?*)
Ya (*Yes*) Tidak (*No*)

3.7 Apakah Bapak/Ibu menghadapi masalah dengan keberadaan KSNP? (*Have you ever had a problem with KSNP?*) Ya (*Yes*) Tidak (*No*)

3.7.1 Kalau ya, jelaskan (*If yes, explain*)

3.8 Apakah ada halangan untuk mencegah petani dalam membuka lahan pertanian yang baru? *(Are there restrictions preventing farmers from opening new farming areas?)*

Kalau ada, jelaskan (*If yes, explain*)

3.9 Apakah bersikapmu pernyataan-pernyataan ini (*To the following statements do you think*):

Ya, setuju (*I agree*) / Tidak pasti (*Unsure*) / Tidak setuju (*Disagree*)

3.9.1 Tidak ada ilegal untuk membuat ladang di KSNP (*It is not illegal to farm in KSNP*)

3.9.2 Ada ilegal untuk membuat ladang di HPH (*It is illegal to farm in HPH*)

3.9.3 Ada illegal mengambil bukan berasakan hasil-hasil hutan non-kayu dari KSNP (*it is illegal to collect NTFP from KSNP*)

3.9.4 Tidak ada illegal mengambil bukan berasakan hasil-hasil hutan non-kayu dari HPH (*it is not illegal to collect NTFP from HPH*)

4. INFORMASI HASIL-HASIL HUTAN NON-KAYU (*NON-TIMBER FOREST PRODUCTS INFORMATION, NTFP*)

4.1 Apakah anda mengumpulkan buatan dari hutannya, kalau begitu yang mana? (*Do you or people on your farm collect forest products, if so which*)

4.1.1	4.1.2	4.1.3	4.1.4	4.1.5	4.1.6
Nama NTFP	Mengumpul NTFP ini (*Do you collect these NTFP*)	Berapa sering mencari NTFP ini[1] (*How often searching for these NTFP*[1])	Berapa jam di dalam hutan mencari (*How many hours spent searching*)	1 trip berapa jumlah diambil rata-rata (*1 trip how much collected*)	Informasi lain (tulis aja) (*Write any other information*)
Damar (*Heartwood resin*)				kg	
Rattan (*Rotan*)				m	
Gaharu (Aquilaria *spp.*)				kg	
Buah buahan (*Fruit*)					
Ikan (*Fish*)					
Jamur (Mushrooms)					
Lain (*Other*)					

[1] Biasanya berapa sering anda pergi di dalam hutannya: A = Daily (setaip hari), B = 2-3 days (hari), C = Weekly (pernah seminggu), D = Fortnight (dua minggu), E = Month (pernah sebulan), F = 3 months (3 bulan), atau G = 6 months+ (6 bulan atau lebih).

5. KECENDERUNGAN POPULASI BINATANG DAN INFORMASI HARIMAU (*WILDLIFE POPULATION TRENDS AND TIGER INFORMATION*)

5.1 Bagimana populasi binatang dari waktu anda baru ke sini membanding dengan waktu sekarang: ada lebih banyak, lebih kecil, sama, atau tidak pasti? (*How has the population of these animals changed from when you first came here: there are more, the same, less, or not sure?*)

5.1.1	5.1.2	5.1.3	5.1.4	5.1.5
Binatang (*Animal*)	Lebih banyak (*more*)	Sama (*same*)	Lebih kecil (*less*)	Tidak pasti (*not sure*)
Napu/Kancil (*Mousedeer*)				
Kijang (*Muntjac*)				
Rusa (*Sambar*)				
Babi (*Wild boar*)				
Babi jengott (*Bearded pig*)				
Landak (*Porcupine*)				
Beruk (*Macaque*)				
Simpai (*Banded langur*)				
Gajah (*Elephant*)				
Harimau (*Tiger*)				

5.2 Bagimana situasi binatang ini jika orang-orang membuat ladang dari hutan: lebih baik, sama, lebih jelek, atau tidak tahu? (*How do you think replacing forest with farmland will affect these animals: better, no difference, worse, not sure?*)

5.2.1	5.2.2	5.2.3	5.2.4	5.2.5
Binatang (*Animal*)	Lebih baik (*better*)	Sama (*no difference*)	Lebih jelek (*worse*)	Tidak pasti (*not sure*)
Napu/Kancil (*Mousedeer*)				
Kijang (*Muntjac*)				
Rusa (*Sambar*)				
Babi (*Wild boar*)				
Babi jengott (*Bearded pig*)				
Landak (*Porcupine*)				
Beruk (*Macaque*)				
Simpai (*Banded langur*)				
Gajah (*Elephant*)				
Harimau (*Tiger*)				

5.3 Apa harimau binatang yang baik? (*Is the tiger a good species*)?
Ya (*Yes*) Tidak (*No*)

5.4 Apa harimau berkepentingan untuk orang-orang di sini? (*Is the tiger important for the people in this area?*)

5.5 Apa konservasi harimau di TNKS penting atau bukan? (*Is it important to conserve tigers in TNKS?*) Ya (*Yes*) Tidak (*No*)

5.6 Apa anda merasa dengan harimau-harimua ada berbahaya untuk orang-orang? (*do you think that tigers are dangerous to humans?*) Ya (*Yes*) Tidak (*No*)

5.7 **Apa anda pernah punya masalah dengan harimau?** (Have you ever had a problem with a tiger?) Ya (*Yes*) Tidak (*No*)

5.8 **Apa anda merasa harimau bisa masih hidup jika ada orang di daerah ini?** *(Do you think that tigers can live with people in this area?)* Ya (*Yes*) Tidak (*No*)

5.9 **Ada ancaman-ancaman untuk harimau di TNKS atau bukan?** *(Is the tiger threatened in TNKS?)* Ya (*Yes*) Tidak (*No*)

5.10 **Apa harimau-harimau sudah ada diproteksi dengan hokum** *(Are tiger protected by law?)* Ya (*Yes*) Tidak (*No*)

Appendix 2: Crop raiding questionnaire survey (with English translations)

1. PEST SPECIES DAN MENJAGA LADANG (*PEST SPECIES AND GUARDING FARMLAND*)

1.1 Apa spesies yang ada pestisida terburuknya, berikutnya, dan berikutnya. Silahkan, kalau ada lain-lain spesies tidak di sana? (*Which is the worst pest species, then the next and then the next. If there is another species please say which?)*

1.1.1	1.1.2	1.1.3	1.1.4
Binatang (*Animal*)	1	2	3
Napu/Kancil (*Mousedeer*)			
Kijang (*Muntjac*)			
Rusa (*Sambar*)			
Babi (*Wild boar*)			
Babi jengott (*Bearded pig*)			
Landak (*Porcupine*)			
Beruk (*Macaque*)			
Simpai (*Banded langur*)			
Gajah (*Elephant*)			
Lain (*Other*)			

1.2 Apa ada anda sedang awasi ini di ladang anda? (*Do you have the following guarding features on your farm?*)

1.2.1	1.2.2	1.2.3
Jenis (*Type*)	Ada (*Present*)	Tidak ada (*Not present*)
Suara pembuat (*Noisemakers*)		
Orang-orangan (*Scarecrows*)		
Snapang (*Gun*)		
Gardu anjing (*Guard dog*)		
TOTAL		