ELEPHANT ENDURANCE IN ACEH

The effects of habitat disturbance and land cover change on the conservation of Sumatran elephants in Aceh, Indonesia.

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ABSTRACT OF DISSERTATION

The Sumatran elephant is seriously threatened by extinction throughout its range. Here, conservation issues threatening the future survival of Asian elephants in Sumatra, and Aceh in particular, are analyzed and evaluated.

The impact of deforestation on the prevalence of elephants living in isolated subpopulations scattered across Sumatra was addressed by analyzing the spatial patterns of deforestation and habitat use of elephants. Deforestation data was obtained from remotely sensed imagery and elephant habitat use was assessed by means of ecological niche modelling. The Sumatra-wide impact of deforestation on elephant population survival was analyzed by comparing the historic distribution of elephants to their current distribution. The observed incidences of population extinctions were then compared to spatial pattern of land cover change and anthropogenic influences. Moreover, the occurrence of crop raiding by elephants was evaluated against the spatial configuration of the forests and forest disturbances. Finally, the effectiveness of different forest conservation strategies was assessed.

Niche modelling revealed that elephants are mainly confined to closed canopy habitats located within landscape depressions and along the forest edge. Surprisingly, elephants were found over a wide range of elevations and were found at locations within rugged terrain. Since deforestation in Aceh was mainly concentrated within the same areas forming the most optimal elephant habitat, elephants are likely to become displaced from their natural ranges. Also, crop raiding incidents appeared to be most frequent in areas which recently had been cleared, but still had undisturbed or secondary forest patches in the direct vicinity. These findings, together with the observation that elephant population survival was significantly reduced in areas which had little forest cover over an extended period of time, suggest that deforestation is the main factor leading to elephant extinctions. To safeguard the survival of elephant populations into the future,

conservation strategies should attempt to integrate elephant habitat requirements into land use plans while simultaneously considering human economic interests. Conserving forest by reducing access appears to be the most effective measure to reduce illegal logging. The application of buffer zones along the forest edge in which limited resource extraction is allowed is therefore more likely to reduce deforestation as compared to the investments needed to actively protect the forest.

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LIST OF ABBREVIATIONS

AIC	Akaike Information Criterion
AFEP	Aceh Forest and Environment Project
AUC	Area Under Curve
BKSDA	Balai Konservasi Sumber Daya Alam (Nature Conservation Agency; Indonesia)
BPKEL	Badan Pengelola Kawasan Ekosistem Leuser (Leuser Ecosystem Management body)
BTNGL	Badan Taman Nasional Gunung Leuser (Gunung Leuser national park management unit)
CRT	Classification Regression Tree
ENFA	Ecological Niche Factor Analysis
EROS	Earth Resources Observation and Science Centre
ESRI	Environmental Systems Research Institute
ETM+	Enhanced Thematic Mapper Plus
FAO	Food and Agriculture organization
FFI	Fauna and Flora International
GAM	Gerakan Aceh Merdeka
GIS	Geographic Information System
GLM	General Linear Model
HEC	Human Elephant Conflict
ICDP	Integrated Conservation Development Project
IPCC	International Panel on Climate Change
IUCN	International Union on the Conservation of Nature
LANDSAT ETM+	Landsat Enhanced Thematic Mapper Plus
LIF	Leuser International Foundation
MANOVA	Multivariate Analysis of Variance
MDF	Multi Donor Fund
OLS	Ordinary Least Squares (e.g. OLS regression)
PAD	Project Appraisal Document
PES	Paying for Environmental Services
PSM	Propensity Score Matching
REDD	Reduced Emissions from Deforestation and Degradation
RGDP	Regional Gross Domestic Product
ROC	Receiver Operation Characteristic
SECP	Sumatran Elephant Conservation Project
SPOT	Satellite Pour l'Observation de la Terre (Earth observation satellite)
UNEP	United Nations Environment Program
WB	World Bank
WWF	World Wide Fund for nature

LIST OF APPENDICES

<u>Appendix I</u>

Rood, E.J.J., Azmi, W. & Linkie, M. (2008) Elephant Crop Raiding in a Disturbed Environment: The Effect of Landscape Clearing on Elephant Distribution and Crop Raiding Patterns in the North of Aceh, Indonesia. *Gajah*, **29**, 17-23.

<u>Appendix II</u>

Rood, E.J.J., Ganie, A. & Nijman, V. (2010) Using presence only modelling to predict Asian elephant habitat use in a tropical forest landscape: implications for conservation. *Diversity and Distributions*, (Online)

Chapter 1

GENERAL INTRODUCTION

1.1 Conservation threats in Asia

Over the last century tropical forest conversion has progressed at an alarming rate. In Indonesia, 17% of the total forest cover and 41% of the lowland forest was cleared between 1990 and 2005 (Hansen *et al.*, 2009b). Since the late 1980s timber extraction and agricultural expansion, especially for oil palm plantation development, have had a major impact on the forest cover of Indonesia (Curran *et al.*, 2004; Corley, 2009; Hansen *et al.*, 2009b; Rudel *et al.*, 2009a). Forest clearance for subsistence garden development has recently been suggested to be responsible for no less than 44% of the total forest conversion occurring in South Asia (The Climateworks Foundation, 2009). In Indonesia, however, drivers of deforestation did show a gradual shift from deforestation resulting from small holder timber extraction and agricultural development to major enterprise driven forest clearance (Rudel *et al.*, 2009b). In Indonesia alone, 20.3 million tons of palm oil are produced on an annual basis, supplying 47% of the worldwide oil palm demand (Koh & Ghazoul, 2010). Despite the general concerns about the environmental and social impact of large scale oil palm exploitation, the expansion of oil palm plantations is expected to increase by 7% per year in developing countries (Carter *et al.*, 2007; Corley, 2009). The industry greatly contributes to the growth of national economies (Koh & Ghazoul, 2010) and contributes approximately 10% to the Gross Domestic Product of Indonesia, making it the second largest commodity after rice (FAO, 2007).

This development has increasingly found the attention of conservation biologists studying the effect of landscape scale habitat alterations on both species richness patterns and species survival. As palm oil plantations have been shown to hold significantly lower numbers of species as compared to undisturbed forests, expansion and intensification of oil palm plantations has become a prominent threat to the conservation of biodiversity and critical ecosystem services (DeFries *et al.*, 2007a; Fitzherbert *et al.*, 2008). Moreover, as reducing deforestation has recently been recognised as a method to alter climate change (IPCC, 2007), potentially reducing the global carbon emission by 130 gigatonnes by 2100, considerable international attention has now being given to the subject (Gullison *et al.*, 2007).

Over recent years, conservationists working in various tropical regions have emphasized the importance of landscape design and protected area networks to counteract the degradation of ecosystems and loss of biodiversity (Brookes, 2002; DeFries, 2007; Gaveau *et al.*, 2007; Koh *et al.*, 2009). Improving conservation strategies for developing countries with complex multi-functional landscapes requires accurate knowledge of conservation needs and values of different land uses. The enforcement of protected areas, however, has often failed to protect tropical forests (Curran *et al.*, 2004) and its biodiversity (Peh *et al.*, 2006). Unclear protected area demarcation, insufficient funds for protection, conflicting benefits and large scale corruption of funds by local authorities are only some of the causes of failing conservation efforts. Hence, an urgent need exists to revise and improve current forest conservation strategies.

1.2 Forest conservation

Many conservationists around the world have increasingly come to recognize the importance of considering economic interests when developing conservation plans. A wide range of sustainable forest management (SFM) approaches have built on the idea to integrate local economic development and poverty alleviation into *Integrated Conservation Development Projects* (ICDPs). The use of such approaches, however, has been shown to be only marginally effective and depend heavily on sound government functioning and the enforcement of conservation legislation based on a firm legal constitution (Goldman *et al.*, 2008; Tallis *et al.*, 2008). Large scale corruption by authorities, responsible for the management of natural resources as well as poor governance, have often imperilled conservation initiatives (Nasi & Frost, 2009). Still, many conservationists, along with scientists and policy makers, now concur that conservation efforts will succeed if local economic interests are not compromised (Butler *et al.*, 2009). Yet, as annual revenues from oil palm in Indonesia exceed the annual budget for nature conservation by more than a 300-fold (Venter *et al.*, 2008), there is little expectation that external funds will be able to divert local economic interests from unsustainable resource extraction practices. Recognizing the fact that human well-being also depends on services provided by nature, which have recently been impaired, allows us to estimate the costs resulting from deforestation and hence provide an economic value to forests (Balmford & Whitten, 2003; van Beukering *et al.*, 2003; Blom *et al.*, 2010).

The increased appreciation of the value of forests, not only as a source of timber, but also as an important factor in the provision of environmental services and to carbon offsets (Balmford & Whitten, 2003; DeFries *et al.*, 2007a; Gibbs *et al.*, 2007; Hansen *et al.*, 2009b; Venter *et al.*, 2009; Blom *et al.*, 2010) has created new incentives for forest protection. The occurrence of natural disasters in many developing tropical countries have made many people poignantly aware of the function of forest for environmental protection and disaster relief and has created a marked increase in the demand for sustainably managed forests (Dennis *et al.*, 2008). Complex compensation schemes financed by the major beneficiaries of forest protection and which pay people who experience a economic loss as a result of forest protection, have been successfully used in several areas (Tallis *et al.*, 2008). Consequently, investing in local livelihoods while simultaneously promoting alternative livelihoods has emerged as a feasible approach to make conservation cost-effective.

On the Indonesian island of Sumatra, oil palm plantation development and the establishment of subsistence gardens have put an enormous pressure on the islands forest estate (Rudel *et al.*, 2009b). The large scale alteration of the ecological integrity of forest ecosystems has led to a decline of species richness and in some cases to the local extinction of forest dependent species in many formerly forested areas (Peh *et al.*, 2006; Koh & Wilcove, 2008). As a result, many of the islands most endangered mammals, including the Sumatran rhinoceros (*Dicerorhinus sumatrensis; IUCN Critically endangered)*, the Sumatran tiger (*Panthera* *tigris sumatae;* IUCN Critically endangered) and the Sumatran elephant (*Elephas maximus sumatranus* Temmick 1847; IUCN Endangered), are now seriously threatened with the prospect of becoming entirely displaced from their natural ranges (Linkie *et al.*, 2008a; Rood *et al.*, 2010). The establishment of *kawasan lindung*[1] to safeguard critical ecosystem processes including the natural ranges of large mammals, has frequently failed to recognise wider ecosystem functioning and has been focussed on small pockets of high ecosystem of biodiversity value. Consequently, the intensification of land use systems surrounding protected areas has often had a considerable impact on the effective size of protected areas. This process, however, can seriously compromise ecosystem processes including local hydrology and source-sink dynamics (DeFries, 2007).

1.3 Sumatran elephant

1.3.1 Taxonomy

The Sumatran elephant, a sub species of the Asian elephant, is confined to the island of Sumatra (Fleischer *et al.*, 2001; Choudhury *et al.*, 2008). This little known subspecies of the Asian Elephant is relatively small and has slightly larger ears than its continental cousin (Fleischer *et al.*, 2001). It also possesses an extra pair of ribs (i.e. 20 as opposed to 19) which clearly distinguished this group of elephants from the mainland populations (Shoshani & Tassy, 2005). Based on the variety of both morphological as well as genetic differences this population is being recognized as a distinct subspecies of Elephas and it is therefore regarded as an Evolutionary Significant Unit (Vidya *et al.*, 2005).

[1] Indonesian for protected areas. This land tenure class indicates a most general protected status and includes a wide range of protected areas. Both wildlife conservation areas as well as areas protected to maintain critical ecosystem services or disaster relief are dented by this term.

1.3.2 Ecology and diet

The most complete work conducted on *Asian* elephant has been that of Sukamar (1989) who provided a comprehensive overview of the ecology, diet and behaviour of elephants in India including references to different regional sub-species including the Sumatran elephant (Sukumar, 1989b). It should be noted, however, that current knowledge on Asian elephant behaviour, dietary requirements and habitat use are mainly based on the study of continental elephants living in dry deciduous forest habitats. Even though these findings are of great value to our understanding of general conservation issues regarding elephants in Asia, care should be taken when extrapolated to elephants living in a tropical evergreen dipterocarp forest habitat be it in Sumatra, Borneo or mainland Asia .

Asian elephants are generalists that utilize a wide range of habitat types such as: tropical evergreen forest, semi-evergreen forest, moist deciduous forest, dry deciduous forested and grassland. Yet, in the face of large scale habitat conversion elephants are now often found to reside in cultivated and secondary forests as well as scrublands ranging from sea level to over 3,000 m asl (Choudhury, 1999). Elephants are mixed feeders, with varying proportions of grass and browse in their diet throughout the year (Sukumar, 1989, 1990; Campos-Arceiz *et al.*, 2008). For the African elephant (*Loxodonta africana*) it has been shown that, even though grasses almost certainly provide the bulk cellulose for energy, the protein requirements of the elephant, especially in dry seasons, can only be met by herbs and browse (Rode *et al.*, 2006).

Asian elephants, given their body mass (1000-4000kg) and physiology, need vast quantities of food and can consume up to 250kg per day to meet their dietary needs (Sukumar, 1990). Grasses and herbs, being rich in carbohydrates (Sukumar, 1990), typically account for more than 50 percent of the elephant's diet, hence grassland forest-mosaics are therefore believed to form optimal elephant habitat. In India, Asian elephants have been shown to feed on more than 112 plant species (Sukumar, 1990). Depending on the availability of protein rich browse species, elephants will spend up to as much as 90% of their foraging time browsing and 10% of the time grazing. If browse is scarce and grasses are more abundant this ratio can be as low as 19:81 (Sukumar, 1989b). Also, during the wet season, when grasses are more abundant, elephants will spend relatively more time grazing (Sukumar, 1989b). Given their high dietary demands, elephants are wide ranging species and home ranges in excess of 600 km^2 have been recorded for females in south India (Sukumar, 1989b). Other studies have reported annual home ranges sizes between 58 - 538 km^2 in North India with bull elephants generally having larger ranges compared to females (Joshi & Singh, 2009).

1.3.3 Population status

Historic data and distribution maps published in the first half of the 20th century suggest that Sumatran elephants were once common across the island and maintained a contiguous distribution encompassing most of the lowland forests (Heurn, 1929; Pieters, 1932; Groeneveldt, 1938). Hence, the demolition of lowland habitat has increasingly fragmented elephant habitat and has left the elephant populations continuing to exist in isolated sub-populations spread across the island. Past assessments of the elephant population on Sumatra have shown that the island still holds a substantial number of wild ranging elephants (Blouch & Haryanto, 1984; Blouch & Simbolon, 1984; Santiapillai & Jackson, 1990). Based on a crude population estimate conducted in 1985, 2500-4800 elephants are now believed to live on the island scattered over 44 distinct subpopulations (Blouch & Haryanto, 1984; Blouch & Simbolon, 1984). These numbers, however, are unlikely to reflect the current status of elephant populations on Sumatra. Numerous subpopulations have been recorded to have gone extinct since the 1980s (Hedges *et al.*, 2005; Uryu *et al.*, 2008) and up to date information on elephant distribution status is lacking for many other populations.

1.3.4 Human-elephant conflict

As human populations continue to encroach into wildlife habitat and ranges, encounters between humans and elephants have become increasingly common across Sumatra (Lemly *et al.*, 2000; Leeney, 2007). Collisions between humans and elephants often lead to conflicting situations with detrimental consequences for both humans as well as elephants. Entering human inhabited areas, elephants have often been found to feed on locally grown crops, causing great damage to local subsistence gardens. Even though crop damage caused by wild boar (*Sus scrofa*) and macaques (*Macaca facicularis*) is generally more frequent when compared to elephants (Linkie *et al.*, 2007), the potential damage caused by elephants has been shown to negatively influence the farmers' attitude concerning elephant conservation (Hill, 1998; Osborn & Parker, 2003; Zhang & Wang, 2003).

In response to the escalating conflict between elephants and resident farmers, the Indonesian government erected *Pusat Latihan Gajah*[2], aiming to provide accommodation for conflict elephants captured from the wild. These animals, mainly rogue elephants bulls but also adult females with their calves, were subsequently captured and transported to the camps with the purpose of being trained for tourism and sustainable logging practices. However, the general lack of knowledge of elephant biology and elephant keeping, as well as the deficiency and corruption of government funds to operate the ETC camps in an adequate manner, has left them worthless. Of all the elephants entering a ETC-camp 50% of the animals die within a year of entering the camp (Stremme, *pers comm.*, 2009). And even though no clear calculation of the magnitude of elephant captures and deaths exists, *ad hoc* data from various sources have confirmed that, in several cases, complete elephant sub-populations have been removed

[2] Elephant sanctuary or Elephant training Centres (ETC) were first established in 1980 to train elephants so that they could be used for tourism activities or sustainable logging practices. Yet, a lack on knowhow and sound financial management, left ETC's to rapidly abandon this original scheme and left camps to resemble elephant depositories. (Blake & Hedges, 2004).

from the wild (Uryu *et al.*, 2008; Rood, pers obs, 2006). Consequently, the remaining elephant population is believed to have faced a continuous decline over the last two decades. To turn around this trend, serious attempts should be made to address the issues which potentially could result to the total eradication of this species on Sumatra. There is a urgent need to develop a Sumatra-wide conservation strategy which allows for sufficiently large areas of elephant habitat to be protected while simultaneously safeguarding local interests. A knowledge-based protected area network covering a multi land use matrix which anticipates potential conflict between economic and conservation interests therefore forms the first and essential step to realize a sustainable conservation strategy for the future.

1.3.5 Elephants in Aceh

The province of Aceh, in the north of Sumatra, is believed to support one of the largest populations of elephants remaining in Sumatra. Accordingly, it is generally believed that this region may offer excellent opportunities for the conservation of wild ranging elephants. Yet, after a history of armed conflict in the province, the renewed peace in Aceh has accelerated forest encroachment which has led to increasing fragmentation of elephant habitat. The development of estate crop plantations, mainly comprised by palm oil and rubber plantations (Direktorat Perkebunan, 2009), has forced elephants to increasingly compete with humans for available space. As a consequence, human-elephant conflict has become widespread in the province. Even though the amount and causes of conflict have not been well reported, it is generally known that elephants can cause a lot of economic damage. Moreover, as elephants are dangerous to humans, they tend to maintain a higher profile concerning crop damage than other wildlife species and consequently are less tolerated by humans (De Boer & Baquete, 1998; Bandara & Tisdell, 2003, 2004), making human-elephant conflict an important social and political issue over recent years.

To ensure the future survival of elephants within the landscape mosaic of forested and agricultural areas, a robust knowledge of elephants habitat use, distribution and response to alterations of their habitat will be invaluable. The availability of up to date information on habitat quality and connectivity will be essential for effective elephant conservation planning and to develop strategies to mitigate conflict. Until recently, the only known generalized distribution maps of elephants in northern Aceh are those of Blouch (1988), Brett (1998) and Canney et al (2002). Although such maps provide some valuable information on the expected range of elephants across Aceh, they are not suitable for detailed spatial planning purposes and conservation management. Consequently, present interests should focus on developing a robust and effective conservation framework which focuses on mapping elephant habitat and distribution patterns throughout the remaining elephant range. Understanding the effects of deforestation and habitat encroachment on elephant persistence and the initiation on human-elephant conflict can provide valuable knowledge to encourage the coexistence of human and elephants in a multi use landscape matrix .

1.4 Research objectives

Numerous arguments can be made as to why it is important to conserve elephants; the following four argument outline the essence of Sumatran elephant conservation:

(1) Sumatran elephants are a widely recognized, evolutionary unique species, with a large though distinct distribution. Elephant occurrence across Aceh and Sumatra signify the importance of the conservation of a wide range of habitats, while its constraint distribution highlights the uniqueness of this species in the world.

(2) Elephants play an important role within their ecosystem, being an important seed disperser and a keystone species shaping their environment (Bowen-Jones & Entwistle, 2002; Venkataraman *et al.*, 2002). Hence, their central role in ecosystem functioning make them a

useful example illustrating how species-environment interactions are vital to preserve ecosystem functioning.

(3) Being a generalist species, elephants function as an umbrella species. Because of their large area requirements, elephant ranges often encompasses a larger number of less well known species which can profit from the protection of elephant habitat.

(4) Elephants are charismatic animals that play an important role in local culture and traditions. The recognition of elephants in Sumatra and Acehnese culture provide an essential foundation to improve awareness on species protection.

Each of these arguments has important implications for the conservation of Asian elephants in Aceh, Indonesia and the continent. The chance of success of any given conservation strategy, however, largely depends on the ability of different stakeholders to develop and endorse conservation plans based on timely and ample data. If such information remains ambiguous or is purely based on subjective judgments rather than factual field data, conservation plans, no matter how detailed and extensive, are unlikely to engage the true problems threatening species survival (Meijaard & Sheil, 2007). To assemble and implement a sound and effective conservation management policy, allowing for a peaceful coexistence between humans and elephant, an accurate and current knowledge of elephant distribution, habitat use and reactions to alterations of their habitat integrity will be of critical importance. This thesis therefore aims to contribute some fundamental knowledge on these issues to promote effective protection and future survival of elephants in Aceh and Sumatra. The following research themes and objectives are addressed in this thesis.

1. Patterns and processes of deforestation in Aceh
    i. Identify current patterns of deforestation and determine past deforestation rates in Aceh, Indonesia
    ii. Identify deforestation threats and the foremost causes leading to deforestation

2. Elephant habitat use
    i. Identify those factors shaping elephant habitat use in a matrix of forested and disturbed habitat
    ii. Estimate elephant habitat suitability and identify core areas for elephant conservation in the northernmost forest block (Ulu Masen) in Aceh, Indonesia

3. Human-elephant conflict
    i. Identify the spatial pattern of crop raiding incidents by elephants surrounding the northernmost forest block (Ulu Masen) in Aceh, Indonesia.
    ii. Investigate the relation between the spatial pattern of deforestation, habitat suitability and the spatial pattern of crop raiding by elephants.

4. Elephant population extinctions in Sumatra
    i. Identify current patterns of deforestation and determine past deforestation rates in Sumatra, Indonesia
    ii. Assess the past and current elephant distribution across Sumatra.
    iii. Identify the driving factors leading to the local extirpation of elephant subpopulations in Sumatra.

5. Forest conservation strategies
    i. Model the expected progression of deforestation in Aceh
    ii. Investigate the effect of different conservation scenarios on the protection of forest over the coming century in Aceh, Indonesia

2.1 Sumatra

Sumatra, the second largest island on the Indonesian archipelago, still holds a relatively large amount of forests, covering 155.466 km^2 or 35% of the total land cover (figure 2.1). Biogeographically, the island is a part of the larger Sundaland which can be roughly delineated by the outline of Borneo, Java and the southern part of the Thai-Malay peninsula. All of these land masses were connected during the last glacial maximum, approximately 20,000 years ago, when sea levels in the region were approximately 120m lower than they are today (Corlett, 2009). Consequently species dispersal during this period has led to large similarities in current faunal and floral diversity between the islands.

2.1.1 Geology and Climate

The topography of Sumatra is demarcated by the Bukit Barisan mountain range stretching 1700 km from north to south along the west side of the island and the lowland alluvial plains in the east. The mountain range, which was uplifted around 70 million years ago (Whitten, 1987) ranges from sea level up to approximately 3800m at local volcanic peaks such as Mount Kerinci in the province of Bengkulu. The northernmost province of Aceh, where the mountains cover more than half of the province surface area, is relatively more rugged compared to the south where large alluvial plains exist. The complex topography and oceanic environment as well as the equatorial position of the island give rise to a high climatic variability. Rainfall patterns vary greatly across different regions and elevations, but a broad west-east rainfall gradient can be recognized. This is characterized by wet conditions along the west coast (~6000mm/year) to relatively dry conditions along the eastern plains (~2500mm/year). Mean monthly temperatures range from 25°C-27°C at sea level to below zero at elevations above 2700m.

Figure 2.1. Map of Sumatra showing the national road network and the remaining forest cover in 2009 (see also chapter 8)

2.1.2 Flora and fauna

The landscape topography, climatic variability and different soil types give rise to a variety of floristic zones including dipterocarp lowland forests, montane forest, freshwater swamp forest, peat swamp forest and mangrove forest (Rennolls & Laumonier, 2000; Laumonier *et al.*, 2010). Sumatra's forests support high levels of biodiversity, comparable to those of other large islands such as Borneo and New Guinea, and significantly higher than those of other islands forming the Indonesian archipelago such as Java and Sulawesi (Whitten, 1987; Meijaard, 2009). The lowland forests of Sumatra support 111 species of dipterocarp trees, including six endemics (Whitten, 1987). Common dipterocarp tree species include white seraya (*Parashorea spp.*) and merantis (*Shorea spp.*). Other abundant tree families in Sumatran forests are: *Burseraceae, Euphorbiaceae* and *Rubiaceae,* which are common at lower elevations, and *Fagaceae, Lauraceae* and *Myrtaceae* that tend to prevail in lower montane and upper montane forests (UNEP, 2004).

The diversity of Sumatran fauna is extensive with 201 mammal species and 580 species of birds recognized to occur across the island (Whitten, 1987). Nine mammal species are endemic to Sumatra and another 14 species are endemic to the Mentawai island group located west of the Sumatran main island. The fauna of the Sumatran mainland can be separated into two regions north and south of the Lake Toba division (Whitten, 1987). Seventeen bird species are only found north of this line while ten others are only found in the south. Similarly, the white handed gibbon *(Hylobates lar)* and the Thomas leaf monkey *(Presbytis thomasi)* are only found North of Toba while the black handed gibbon *(Hylobates agilis)* and both the mitered leaf monkey *(Presbytis melalophus)* as well as banded leaf monkey *(Presbytis femoralis)* are only found in the south. Moreover the Indian Tapir is only found south of this division. The Toba division of species distributions is commonly believed to be caused by the physical barrier of bare volcanic ash that was created by the eruption of the Toba volcano approximately 74.000 years ago (Whitten 1987). An alternative hypothesis to the Toba eruption theory claims that differences in rainfall intensity and evaporation patterns between the north and the south of Toba could have led to the observed separation of mammal distributions as suggested by (Natus, 2005). This hypothesis, however, finds little support in the scientific literature. A third hypothesis, posed by Meijaard et al. (2004) states that the area north of the division has been cut off from the rest of Sumatra during most of the Pleistocene era due to elevated sea levels.. Amongst the large diversity of Sumatran mammals are some of the most endangered large mammals living in the Asian region including the two-horned Sumatran Rhinoceros (*Dicerorhinus sumatrensis)* the Sumatran tiger (*Pantera tigris sumatrae*), the Sumatra Orangutan (*Pongo albeii)* and the Sumatran elephant (*Elephas maximus sumatranus*) which is the focal species of this dissertation.

2.2 Aceh

2.2.1 Geology and climate

The province of *Nanggroe Aceh Darussalam*[3] covers the northernmost tip of the Bukit Barisan mountain range, running along the west side of Sumatra (figure 2.2). The province sustains the largest remaining contiguous forest area on the Island, stretching 33.1 00 km^2 from the north to the south. Variations in the physical environment give rise to local differences in the hydrology and hence productivity of the supporting abiotic environment. In the north of Aceh extensive limestone areas, which are very porous by nature, support only low density forests. On the other hand, areas of volcanic or sedimentary origin often give rise to more productive forest. Peat swamps are found on the alluvial plains forming a small strip along the west coast of Aceh. A number of discrete vegetation classes can be recognized within the province including: lowland broadleaf forest, montane broadleaf forests, pine forest, freshwater swamp forests, mangrove forests and peat swamp forests (Laumonier *et al.*, 2010).

[3] Honourable Nation of Aceh

Figure 2.2 Map of Aceh showing the locations of the two main protected areas in the province and the remaining forest cover in 2009 (see also chapter 4)

Aceh, as the rest of Sumatra, has an extended wet season (9-6 months) and a short dry season (2-4 months) with less than 100mm of rainfall per month. The climate of Aceh is typically drier than the rest of the Sumatran mainland, with a mean annual rainfall of 3000-5000mm/year along the west coast, decreasing towards the east with increasing elevation. Annual rainfall as low as 1000-1500mm/year can be found in the north east which is located in the rain shadow of the outer stretches of the Bukit Barisan range. Little variation in the average daily temperature exists between seasons (25°C-27°C).

2.2.2 History and Religion

The Islamic sultanate of Aceh was founded around the 15th century and is believed to be the founding region of the Islam Indonesia (Reid, 2004). Until the Dutch siege in the nineteenth century Aceh was an autonomous sultanate with relatively close economic and cultural linkages with the Malaysian and South Asian region, in contrast to the southern parts of Sumatra which were dominated by the Javans and Dutch (Kingsbury, 2007). The strategic location of the Sultanate made the area an important and powerful *negeri*[4] ruling local trade, and extensive trade links existed between the Sultanate and Turkey, India, England, America, France and Italy (Schulze, 2003).

In the 17th century under the reign of Sultan Iskandar Mudah Aceh had become a major competitor in the pepper trade in the region and provided about half of the worlds pepper supply (Reid, 2004). Aceh's influence on ports of the Sumatran and Malaysian coasts initially increased as they allied with the Ottomans and the Dutch East India Company against the Portuguese. However Aceh gradually lost power as the British gained control in Malaysia (Penang) and the Dutch expanded their rule on Sumatra.

In 1873 the Dutch declared war on Aceh to try to gain control over the trade in the Malacca strait. The Acehnese bitterly resisted Dutch occupation in a battle for independence

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[4] Nation (bahasa Indonesia)

during which approximately 100,000 Acehnese were killed (Reid, 2004) . Over time, Acehnese resistance against the Dutch , however, became more dominated by religion and motivated out of Islamic faith as dictated by *Ulamas*[5], than for an independent state or *Darul Islam*[6].

In 1942 the Japanese invaded Aceh and the Dutch were forced to withdraw. After the war the former sultanate officially became a Province of Indonesia, but dissatisfaction with the policies of the central government in Jakarta started to grow (Schulze, 2003; Kingsbury, 2007). Only five percent of the provincial financial contribution to the national economy was returned to the province, and resulting hardship and poverty contributed to the feeling of misfortune and exploitation (McCullogh, 2003). In 1959 the province took up arms against the central government in Jakarta with the objective to create the autonomous Islamic state of Nanggroe Aceh Darussalam (McCullogh, 2003). Consequently, the prevailing perception of being withheld from their resources lead to anger and misfortune led to the establishment of the Gerekan Aceh Merdeka[7] (GAM) (Kingsbury, 2007). Numerous insurgencies during the 1970s and late 1980s were followed by periods of harsh military repression by the Indonesian army (Barter, 2008). In 1998 following the fall of the Suharto regime the conflict escalated and GAM eventually took hold of approximately 70% of the Acehnese territory (Schulze, 2003).

In 2002, the national parliament approved a law which gave a special autonomy status to Aceh. Key provisions of this new law included: (1) enforcement of aspects of shari'ah[8], (2) a larger share of natural resource revenues compared to other provinces and (3) direct elections

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[5] Ulamas (referred to as *Teunku* in Acehnese) are scholars engaged in Islamic studies and are perceived as Islamic arbiters by Islam. In Aceh the Ulamas gradually gained power in the armed combat against the Dutch over the traditional Acehnese warlords (Uleebalang).

[6] Islamic state (translated from Arabic)

[7] *Gerakan Aceh Merdeka* or *GAM* stands for the Free Aceh Movement which was established by the local *ulama* Hassan di Tiro in 1976. GAM was the first organized freedom movement demanding independence from the Indonesian Republic.

[8] The word "Shari'ah" literally means "way or "path" and refers to the canonical law of Islam conveyed to mankind by Allah

of governor and district heads (Aspinall, 2005). However, due to corruption in the local governments and the fact that the new law did not recognize the GAM as a independent political party, the peace attempt floundered and the violence escalated again in May 2003 (Schulze, 2003; Aspinall, 2005). It was not until the devastating Tsunami hit the coast of Aceh in December 2004 that an end to the violent conflict was reached. On 15 August the Indonesian Government and the representatives of GAM signed a Memorandum of Understanding aimed at achieving a peaceful, comprehensive and sustainable solution after more than 30 years of continuous military conflict.

2.2.3 Population and economy

With a population of just over four million people (BPS, 2010b), Aceh remains one of Indonesia's economically least affluent provinces (BPS, 2010a, b). Aceh comprises several ethnic groups including the Acehnese, Gayo, Kluet, Karo and a small minority of Christian Bataks living in the south of the province. Even though the area is rich in natural resources including natural oil, natural gas, minerals and natural resources (i.e. timber) and provides up to 20% of Indonesia's gas and oil, 50% of the population lived below the poverty line in 2006 (World Bank, 2008). Oil and gas form 43% of the Regional Gross Domestic Product (RGDP) while agriculture accounts for 30% of the RGDP (Bappeda, 2006). Albeit the fact that Aceh is rich in resources and has great capacity for economic development, exploitation over the last decades has not led to the improvement of livelihoods of the majority of the population (World Bank, 2008). Rural communities have been deprived of traditionally claimed lands which forms the primary source of rural economic development. The main drawback to economic development in the region has been caused by local governments imposing polices which facilitated investors to invest in the area without securing benefits for local communities. In particular large scale corruption by governmental institutions has caused a major limitation to the development of local livelihoods (McCullogh, 2003).

2.2.4 Land use

Of the 5.64 million hectares in Aceh Province, 3.35 million are currently officially considered *Hutan Negara* (State forest). Various forest tenure classes exist and 52% of the province's forest estate is protected by law (table 2.1). The current land tenancy system allows for different degrees of utilization and resource extraction within designated areas including: protection forest, permanent and limited production forest and conservation forest (table 2.1). At 2.6 million ha the Leuser ecosystem comprises the largest protected area in Indonesia and partly covers the provinces of Aceh and North Sumatra (figure 2.1). In 2010 the Ulu Masen ecosystem (750.000ha, figure 2.1) was put forward as a protected forest area (*Kawasan strategis)* to safeguard environmental services such as water cycle regulation, erosion prevention and nutrient cycling.

Landuse spatial plan NAD 2008	Area (km^2)	%
Protected areas		
Swamp	60	0.1
Nature reserve	1270	2.2
Protected forest	13,380	23.3
Limited production forest	8,180	14.3
Germ plasm/Biodiversity areas	10	0.0
Hunting Park	990	1.7
Grand forest park	140	0.3
National park	6,130	10.7
Total protected areas	30,180	52.6
Production areas		
Conversion forest	670	1.2
Permanent production forest	7,800	13.6
Industry	10	0.0
Wet agricultural lands	3,100	5.4
Dry agricultural lands	4,540	7.9
Aqua culture	730	1.3
Plantations	7,500	14.0
Residence area	2,280	4.0
Total Non-protected areas	26,640	47.4
Total Province Aceh	56,830	100.0

Table 2.1 Overview of land use allocation in Aceh as proposed in the Provincial spatial plan (RTRWP) 2008.

Areas designated as non-protected lands are classified into two broad land use categories which distinguish between (1) areas currently designated as production forests, where various extraction activities are permitted and (2) non forested areas. The non-forest area mainly comprises unclassified land, which is designated for community uses such as construction sites, subsistence agriculture and large scale agriculture (e.g. oil palm plantations, rubber and coffee; table 2.2).

In 2001 the special autonomy law of Aceh, presented the provincial government with self-sufficiency over the management of the province's natural resources. After the election of the current Governor, Irwandi Yusuf, in 2006, a province-wide cessation of logging practices was endorsed by the government of Aceh in 2007. One of the main priorities of the new governor of Aceh is to conserve the natural resources of Aceh through an integrated and sustainable land use management system. In order to engage local stakeholders to participate in the use management process, traditional laws and *Mukim*[9] land management authorities are to be re-established in the province. Participatory land use planning and a wider framework for government support through recognition of forest resource rights is expected to facilitate sustainable forest and resource management in Aceh.

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[9] *Mukim* or *mukiman* is a traditional administration system which currently only exists as an unofficial management structure. Mukims are now subdivisions of *kecematans* (sub-districts) and contain several villages which are lead by a *kepala mukim* (mukim leader).

Table 2.2 Overview of plantations in Aceh per district. Area utilized per crop (103 ha) , percentage of total area utilized and total production (tonnage) are given. The last column indicates the percentage of district surface area utilized for plantations. Source: Direktorat Jenderal Perkebunan: "Statistik Perkubunan 2008-2010" (2010)

District		Palm Oil			Cacao			Rubber			Coffee			Coconut			Cloves			Patchouli			Total Area	
	Area (1000 ha)	Area (103 ha)	Area (%)	Prod. (ton)	Area (103 ha)	Area (%)	Prod. (ton)	Area (103 ha)	Area (%)	Prod. (ton)	Area (103 ha)	Area (%)	Prod. (ton)	Area (103 ha)	Area (%)	Prod. (ton)	Area (103 ha)	Area (%)	Prod. (ton)	Area (103 ha)	Area (%)	Prod. (ton)	Area (103 ha)	Area (%)
ACEH BARAT	277.27	15.7	(41%)	29.3				19.3	(51%)	16.7	0.2	(1%)	0.8	2.9	(8%)	1.6							38.1	14%
ACEH BARAT DAYA	187.43	5.7	(38%)	3.4	3.4	(23%)	0.4	0.2	(1%)	0.2	3.4	(23%)	1.4	2.1	(14%)	1.3				0.1	(1%)	0.0	14.9	8%
ACEH BESAR	293.14	1.2	(4%)	0.1	0.8	(3%)	0.2				7.9	(29%)	4.6	14.6	(54%)	6.5	2.5	(9%)	0.15				27.1	9%
ACEH JAYA	390.20	10.7	(30%)	5.8	3.0	(8%)	0.5	6.8	(19%)	4.6	8.1	(23%)	1.8	5.9	(17%)	1.7	0.3	(1%)	0.02	0.6	(2%)	0.0	35.4	9%
ACEH SELATAN	419.66	10.1	(73%)	8.2	0.4	(3%)	0.0	0.7	(5%)	0.4	0.4	(3%)	2.8	0.1	(1%)	0.0	1.2	(9%)	0.02	0.9	(6%)	0.0	13.8	3%
ACEH SINGKIL/SUBUSSALAM	305.37	40.3	(70%)	45.6	0.5	(1%)	0.1	7.0	(12%)	6.7	1.0	(2%)	0.3	8.6	(15%)	3.8	0.5	(1%)	0.08	0.0	(0%)	0.0	57.9	19%
ACEH TAMIANG	214.87	47.4	(59%)	73.3	2.0	(3%)	0.7	29.7	(37%)	26.1	0.3	(0%)	0.1	0.8	(1%)	0.3							80.2	37%
ACEH TENGAH	445.40				0.3	(1%)	0.0				19.8	(98%)	27.8	0.1	(0%)	0.0				0.1	(0%)	0.0	20.2	5%
ACEH TENGGARA	416.96	1.4	(10%)	0.9	8.4	(58%)	6.0	1.9	(13%)	1.6	1.9	(13%)	0.3	0.6	(4%)	0.1				0.3	(2%)	0.0	14.5	3%
ACEH TIMUR	544.82	54.2	(52%)	72.9	11.8	(11%)	6.5	28.4	(27%)	21.0	3.5	(3%)	0.9	6.3	(6%)	5.5							104.3	19%
ACEH UTARA	278.86	29.6	(50%)	30.9	10.0	(17%)	3.8	9.9	(17%)	2.4	3.5	(6%)	0.9	6.3	(11%)	5.5							59.4	21%
BENER MERIAH	190.40	0.1	(0%)	0.0	0.7	(2%)	0.1				39.5	(87%)	12.4	4.1	(9%)	2.5				1.1	(2%)	0.0	45.5	24%
BIREUEN	182.89	4.6	(14%)	5.3	4.6	(13%)	3.8	6.8	(20%)	4.8	3.0	(9%)	1.8	14.1	(42%)	7.2	0.8	(2%)	0.03				33.9	19%
GAYO LUES	554.99				3.4	(13%)	0.4				21.8	(82%)	4.0	0.5	(2%)	0.1				0.8	(3%)	0.0	26.4	5%
NAGAN RAYA	354.27	51.6	(70%)	43.3	4.2	(6%)	0.6	6.4	(9%)	5.1	8.2	(11%)	3.4	3.1	(4%)	1.2							73.4	21%
PIDIE/PIDIE JAYA	412.93	0.1	(0%)	0.0	8.2	(11%)	2.0	0.0	(0%)	0.3	56.6	(77%)	5.0	8.7	(12%)	5.3				0.0	(0%)	0.0	73.6	18%
SIMEULUE	181.49	0.9	(4%)	0.0	1.6	(7%)	0.1	0.6	(2%)	0.3	0.9	(4%)	0.1	6.9	(28%)	1.3	13.7	(56%)	1.62				24.7	14%
KOTA BANDA ACEH	5.72																						0.0	0%
KOTA LANGSA	17.49	0.5	(24%)	0.6	0.2	(9%)	0.5	1.0	(45%)	0.5				0.5	(22%)	0.4							2.1	12%
KOTA LHOKSEUMAWE	6.97	0.2	(13%)	0.1	0.1	(11%)	0.1	0.1	(8%)	0.0	0.1	(9%)	0.1	0.7	(58%)	0.3							1.2	17%
KOTA SABANG	12.43				0.6	(15%)	0.2							1.3	(31%)	0.6	2.2	(54%)	0.16				4.1	33%
Total	5693.6	274.2	(37%)	319.8	64.3	(9%)	25.8	118.7	(16%)	90.7	180.1	(24%)	68.4	88.1	(12%)	45.1	21.3	(3%)	0.3	3.9	(1%)	0.1	750.6	13%

2.2.5 Deforestation

Over the last decades, one of the foremost drivers of deforestation in south east Asia has been the rampant expansion estate crops such as of oil palm (Holmes, 2002; Hansen *et al.*, 2009b). In Aceh, the conversion of forest for the development of oil palm plantations has been responsible for approximately 64% of the total forest loss observed between 1984 and 1997 (Holmes, 2002). The *Rencana Tata Ruang Wilayah Propinsi*[10] Aceh, drafted by the central government of Indonesia in 2008, allocates 28% of the total land surface to the development of plantations (table 2.2), of which more than half includes currently forested areas. Other major threats to the prevalence of Aceh's forests are posed by the paper-, bio-fuel and timber industries, mining concessions excavating minerals and, to a lesser extent, by seasonal burning (Hoffmann, 2009).

The forests of Aceh are rich in tropical hardwood tree species such as meranti, semaram and merbau which fetch high prices on the timber market, making logging a very lucrative business. Prior to the Tsunami 47 timber companies were granted logging concession in Aceh. Yet, after the instalment of the new government and the enforcement of the province-wide logging ban in 2007, most of the former concession were discontinued. Reconstruction activities, however, created a high demand for building materials with an estimated 861.000 m^3 of timber required for construction of new houses and shelters (World Bank, 2008). Moreover, the development of roads by international aid projects across the province, has created access to previously inaccessible forest areas and has led to an increase in illegal logging activities. Different organizations monitoring forest loss in Aceh by means of the analysis of remotely sensed images reported deforestation rates ranging from 4.7% (0.31%/year) (DisHut, 2004) to 13.8% (0.92%/year) (FFI Aceh 2009) between 1990 and 2005. Eye-on-Aceh, an Indonesian environmental NGO, reported that an astonishing 130.000 ha or

[10] *Rencana Tata Ruang Wilayah Propinsi* or *RTRWP* is the provincial spatial land use plan.

3.9% of the total forest cover was lost due to illegal logging during a single year in 2005-(Eye-on-Aceh 2009).

2.2.6 Threats to biodiversity

The continuous exploitation of Aceh's natural resources over the last few decades has led to a near complete depletion of the provinces' lowland forests (FFI Aceh, 2009) . Undeniably, the continuous conversion and clearance of forest will ultimately lead to landscapes dominated by agriculture in which a severely fragmented forest patches only remain in the least accessible areas. Yet, Aceh still supports a number of threatened large mammal populations (i.e. Sumatran elephant, Sumatra rhinoceros Sumatran Orangutan and Sumatran tiger) , which depend on the province's forest to survive. Several of these species are only found in small isolated populations across Sumatra and the forest realm of Aceh is one of the most important refuges (see chapter eight this thesis). Hence, it is generally believed that deforestation forms one of the major threats to the survival of the area's rich biodiversity. Particularly mega-fauna inhabiting most of the remaining forests of Aceh and which require large stretches of suitable habitat are under severe pressure as a result of deforestation (Kinnaird *et al.*, 2003; Gaveau *et al.*, 2009b).

The displacement of many large mammals resulting from deforestation has now seriously compromised the availability of suitable habitat for many species (Sodhi, 2008; Sodhi *et al.*, 2010). Species, which explicitly rely on forest habitat, such as Rhinoceros and Orangutan, are prone to extirpation when their natural habitat is altered. Hence the occurrence of these species in the forests of Aceh is now believed to be seriously compromised (Griffiths & Schaik, 1993; Foose & Strien van, 1997; Wich *et al.*, 2008; Azmi *et al.*, 2009) On the other hand, the reduction of suitable forest habitat patches has left some more resilient mammals to find their historic ranges completely converted to agricultural lands and plantations. In many of such cases, competition between humans and local wildlife for suitable space has led to conflicting encounters between the two (Nyhus *et al.*, 2000; Nyhus & Tilson, 2004; Linkie *et al.*, 2007).

2.2.7 Elephant conservation in Aceh

The occurrence of human-wildlife conflict in Aceh is exemplified by the occurrence of human-elephant conflict (HEC). Being a wide ranging species living in the direct vicinity of many populated areas, elephants frequently enter populated areas where they cause damage to houses and agricultural fields (Nyhus *et al.*, 2000; Nyhus & Tilson, 2004; Linkie *et al.*, 2007). Crop damage caused by elephants is a common form of human–wildlife conflict occurring within the landscape matrix of isolated forest patches interspersed with agricultural communities. The economic losses suffered from human-elephant conflict can be severe and has made communities antagonistic and intolerant towards wildlife (Rood *et al.*, 2008). Human-elephant conflict has often been found to provoke reprisal killing of problem elephants as well as discouraging conservation strategies amongst local communities (Nyhus & Tilson, 2004; Linkie *et al.*, 2007; Rood *et al.*, 2008; Hedges & Gunaryadi, 2010).

Some of the major challenges of elephant conservation are to create awareness of the economic values of conservation and to reinforce cultural associations, while simultaneously safeguard the economic returns of conservation strategies to local communities. To address conservation issues in Aceh, a landscape level conservation project focussing on the conservation of a culturally important symbol like the Sumatra elephant could provide an important approach to protect large areas of habitat. Hence, in 1998 the UK based conservation agency, Fauna and Flora International (FFI) started a project focussing on the *in situ* conservation of the Sumatran elephant in Aceh and North Sumatra. The main focus of the *Sumatran Elephant Conservation Project* (SECP) was to safeguard significant areas of land, designated as "managed elephant range" (MER). This required support from key groups in Acehnese society and a social change and awareness to reassert or develop positive attitudes towards elephants and their conservation (Jepson *et al.*, 2002). In Aceh the major impasse to anticipate social issues regarding elephants conservation is posed by the occurrence of human-elephant conflict. Nevertheless, elephants have strong traditional and cultural roots in the Acehnese culture which creates opportunities to deal with these issues.

The Sumatran elephant is deeply rooted into the Acehnese history and culture as a symbol of military strength and prosperity (Clarence-Smith, 2004). Early reports from 1640 stated that the Acehnese Sultan Iskandar Muda owned over 900 elephants trained for war and his descendant Sultan Iskandar Thani possessed *"...white beasts with four tusks..."*. After the death of Sultan Iskandar Thani 260 elephants covered is high quality fabrics took part in his funeral parade, emphasizing the importance of elephants in early ceremonial events (Clarence-Smith, 2004).

Even after the colonisation epoch by the Dutch (1873-1940) a wide local knowledge and strong anecdotal traditions regarding elephants are still common in Aceh (Bowen-Jones & Entwistle, 2002). This is especially evident from the fact that the Acehnese people have a high tolerance and respect towards elephants even in the presence of crop raiding and occasional human casualties resulting from human-elephant conflict (Bowen-Jones & Entwistle, 2002).

2.2.8 Conservation framework in Aceh

A number of local and international conservation NGOs (FFI, LIF, WWF, Eye on Aceh, SILFA) as well as governmental institutes (BKSDA, Dinas Kehutanan, BTNGL, BPKEL) are actively involved in elephant conservation in Aceh. The most prominent project of the last few years has been the Aceh Forest and Environment Project (2006-2010), implemented by FFI and the Leuser International Foundation (LIF), funded by the Multi Donor Fund (MDF) and supervised by the Wold Bank (WB), which aims to protect the remaining forest of Aceh through introducing sustainable aspects to the province's economic rehabilitation. Even though the AFEP project activities substantially benefit the conservation of elephants in the province, project aims and regulations, formulated in the project appraisal document (AFEP-PAD 2007), and the lack of participation from local counterparts have constrained the implementation species specific conservation activities.

Aiming primarily at poverty alleviation and economic development to reduce pressure on natural resources and promote conservation, the AFEP project does not specifically support species conservation programmes *per se*, but focuses on forest conservation in Aceh. However useful, it fails to recognize the fact that human-elephant conflict has an immediate impact on local livelihoods and hence the conservation ethic of the communities involved. Elephant conservation schemes could directly benefit both species conservation as well as local livelihoods while simultaneously providing a strong incentive for forest protection. Yet, conflicting interests between government institutions and conservation NGO's as well as competition over the moral ingenuity of conservation initiatives between NGO's have hampered conservation efforts to come into effect. Consequently, cooperation between different government departments and conservation NGO's has been marginal and only limited progress has been made to facilitate collaboration between the parties involved.

The call to deal with problem elephants from both local farmers as well as forestry authorities has been increasing over the last decade. Elephant management strategies, however, have only provided temporal and destructive solution by removing or killing problem elephants and have failed to recognize and deal with the underlying causes ultimately leading to human-elephant conflict. Apart from the Sumatran Elephant Conservation Project, which has now been active in Aceh for more than 12 years, no collaborative conservation framework has been set up to guarantee prolonged elephant conservation. Meanwhile, in the absence of tangible alternatives, the Indonesian Directorate for Nature Conservation has continued to capture problem elephants from their wild populations. An independent investigation by the Aceh based NGO SILFA, showed that between 2007-2008, 45 elephants, which was estimated to represent approximately 10% of the total Acehnese elephant population, were captured from the wild (SILFA, unpublished data, 2009). Within the same year nine had died in elephant camps due to inadequate health care or starvation (SILFA, unpublished data, 2009). Hence, as long as conservation NGOs and government parties involved in elephant conservation fail to join efforts and work in a transparent manner by sharing information, capacity and resources, the investments, no matter how large, will only slow down the total eradication of elephants in Aceh as observed in other Indonesian provinces (chapter eight).

3.1 Introduction

At present, a wide range of analytical methods and corresponding statistical techniques are available to scientists who focus on the distribution of wildlife across the landscape and the effects of perturbations of natural habitats on species behaviour and survival (Guisan & Zimmermann, 2000; Phua & Minowa, 2005; Elith *et al.*, 2006; Guisan *et al.*, 2006; DeFries *et al.*, 2007a). Many of these studies aim to reveal processes or factors which influence species' distribution or spatial organization. Inferences of habitat characteristics on the spatial distribution of a species can provide valuable insights to be used by wildlife managers and conservationists as they can be used to prioritize areas for conservation (Guisan & Zimmerman, 2000).

Here, an overview of the methods used to collect data and a description of the statistical techniques used for data analysis is presented. Hence, the aim of this chapter is to describe and clarify the methods used to collect data rather than to present results and conclusion derived from data analysis. Likewise statistical techniques used in this thesis will be outlined and explained in detail. Yet, the application of the different methods to address the research objectives stated in the introduction, and the presentation of the results and conclusions will follow in the subsequent chapters.

3.2 Forest cover Sumatra

Forest cover change across Sumatra between the years 1990 and 2005 was estimated using 26 images acquired by the LANDSAT 7 ETM+ sensor (2005) and 26 images acquired by the Landsat 4-5 TM sensor (1990). Satellite data were obtained from the USGS Earth Resources Observation and Science centre (EROS) at http//glovis.usgs.gov. Each Landsat image consists of seven spectral bands that approximately span an area of 185km x 170km with a 28.5m x 28.5m resolution (Brown *et al.*, 2009). All satellite images were orthorectified when downloaded and radiometrically corrected using the metadata included in the original data files. Due to a technical failure of the Landsat 7 satellite in 2003 (e.g. USGS SLC-off images) and temporal distortion of images resulting from atmospheric hazes and cloud cover some of the images did not provide a comprehensive coverage for their respective area. Fifteen additional images, derived within a 12 month period from the original image, were consequently downloaded to fill these gaps.

3.2.1 Data analysis

All images were separately classified using a Classification Regression Trees (CRT) algorithm (Lawrence & Wright, 2001). The CRT method is based on a decision tree algorithm that recursively classifies data into subsets based on a measure of misclassification or impurity (Lawrence & Wright, 2001). This method offers several advantages over other classification algorithms such as Bayesian classification or K-means classification. Firstly, CRT do not make any assumptions about the underlying distribution of the data to be classified (e.g. non-parametric data) and can deal with nonlinearity. Secondly, the classification results comprise straightforward cut-off points which are easily interpreted and used for raster classification in a GIS (figure 3.1A). Thirdly, CRT is robust with respect to outliers which make it suitable if training data is scant or contains errors (Moisen & Frescino, 2002).

For each Landsat image a minimum of 100 training data points were manually digitised from Quick bird and SPOT 5 derived satellite images. Forest land cover was defined as old primary forest stands which do not show signs of past logging operations, or that have only marginally been affected by selective logging and consequently have maintained a continuous cover. Non-forest areas included: urban areas, gardens, plantations and secondary forest or regrowth. For each training data point the reflectance values of all spectral bands obtained from the LANDSAT imagery, except for the first band (0.45 – 0.52 μm), were extracted and stored. The first band was omitted from the analysis as this part of the light spectrum is

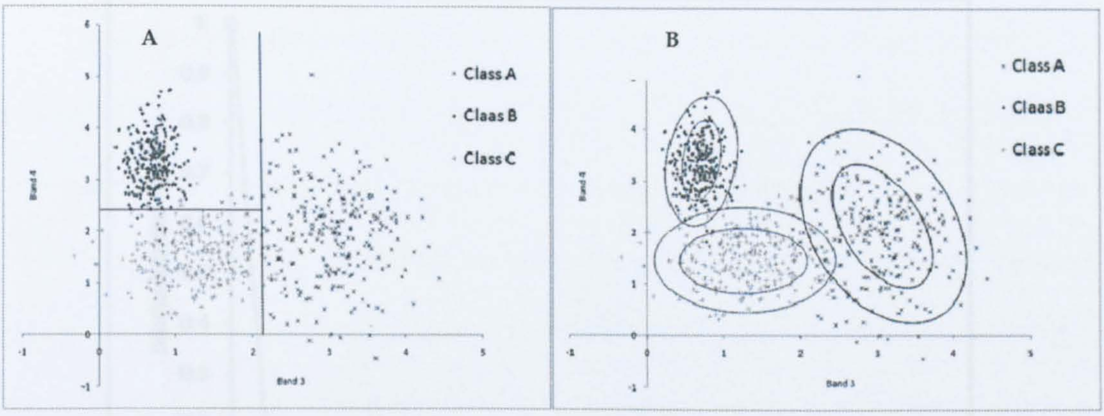

Figure 3.1 Schematic representation of the classification of three land cover classes (A,B.C) based on two reflectance bands (band 2/3). A classification regression tree (CRT) recursively partitions the data into classes based on class purity or the relative misclassification rate (A).The maximum Likelihood approach classifies the data based on the maximum probability of a training data point to belong to a certain class based on the mean value and variance observed within the training data (eg parametric) (B).

sensitive to atmospheric hazes and clouds and thus can potentially bias the results (Phua & Saito, 2003). The training data was exported from a geographical information system (ESRI ArcGIS 9.3, 2008) and analyzed using SpSS (SpSS 16.0.1, 2007) using a classification tree algorithm (Breiman *et al.*, 1984). Cases were split based on a measure of impurity using the Gini coefficient of inequality. First, a tree was grown until no improvement in the class impurity was observed (e.g. no improvement). A ten-fold cross validation was then used to determine the relative misclassification rate of each tree after a single consecutive split. The optimal tree was then found by taking the tree with the least number of splits which had an relative error within one standard error of the minimum error tree (figure 3.2; Breiman *et al.*, 1984). The set of rules predicted by the CART algorithm were then used in a GIS produce a forest cover map.

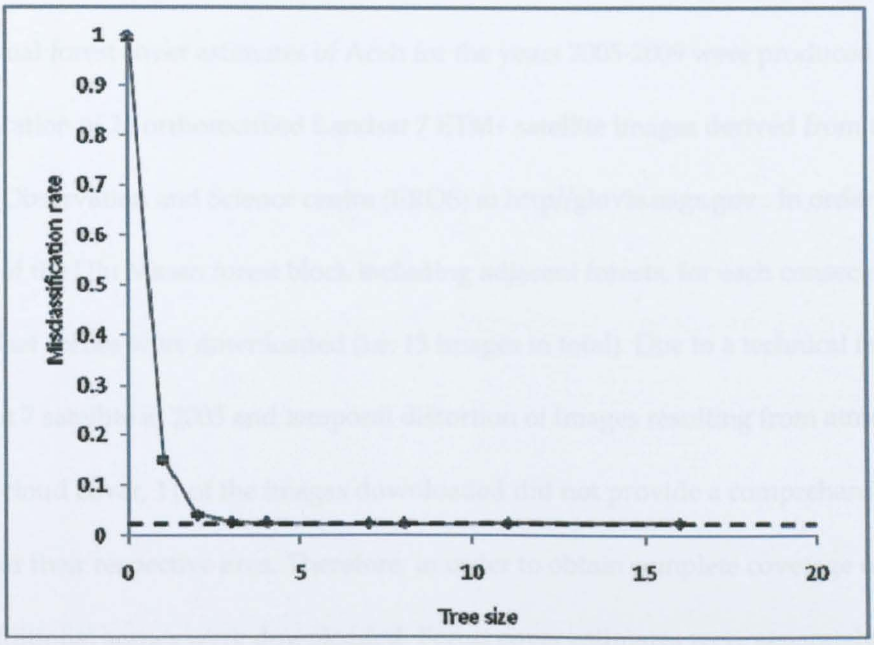

Figure 3.2 Example of a classification tree accuracy plot using four LANDSAT ETM+ spectral bands (b3-6) to classify four land cover classes. The average relative classification errors (misclassification rate), calculated from a ten-fold cross-validation, are plotted against the tree size (number of nodes). The dashed line indicates the minimum relative error of the maximum resolved tree. Error bars represent standard deviations calculated from the tenfold cross validation. In this example the optimum tree is found after four splits as no significant improvement of the classification accuracy is observed.

3.2.2 Validation

The final output of the forest classification was validated using 500 randomly located points which had land cover classes assigned based on the visual interpretation of 2005 0.6 m^2 Quickbird satellite imagery. Points which were located on image pixels that represent water bodies, clouds or cloud shadows, were reassigned to a new, randomly selected location. A 2x2 confusion matrix was constructed and a Kappa-statistic (Moisen & Frescino, 2002) was calculated to assess the accuracy of the classification.

3.3 Forest cover Aceh

Annual forest cover estimates of Aceh for the years 2005-2009 were produced based on the classification of 26 orthorectified Landsat 7 ETM+ satellite images derived from the Earth Resources Observation and Science centre (EROS) at http//glovis.usgs.gov . In order to cover the whole of the Ulu Masen forest block including adjacent forests, for each consecutive year, three Landsat scenes were downloaded (i.e. 15 images in total). Due to a technical failure of the Landsat 7 satellite in 2003 and temporal distortion of images resulting from atmospheric hazes and cloud cover, 11 of the images downloaded did not provide a comprehensive coverage for their respective area. Therefore, in order to obtain complete coverage of the study area, 11 additional scenes were downloaded. Forest cover estimates were separately produced for each image using a maximum likelihood classification algorithm (Mather, 1987; Fraley & Raftery, 2003). The classification algorithm was implemented using R statistical software *mclust* package (Fraley & Raftery, 2006) (http://cran.r- project.org/web/ packages/mclust/index.html) and ESRI ArcGIS 9.2 remote sensing software. A detailed overview of the classification approach used will be further described in chapter four.

3.3.1 Maximum Likelihood cluster analysis

The Maximum likelihood algorithm is a form of supervised classification that uses training data (see 3.3.1) to calculate a land cover class specific Bayesian probability functions (mean, variance/covariance) given a number of input variables or satellite bands (Figure 3.1-B, Mather, 1987). Each pixel in the original image is assigned to the class given the highest probability of class membership. This method generally produces accurate results , but is computationally intensive. Also it can only be applied if the basic assumptions of parametric statistics apply to the data (e.g. normality, homogeneity of variance, independent errors). Therefore a random sample of 500 pixels was taken from each spectral band and tested for normality and equal variances using SpSS 16.0. Hence, all bands of each image were standardized after which all variables met these criteria and were subsequently included in the analysis. Image analysis and classification results are discussed in more detail in chapter four of this thesis.

3.3.2 Training data collection

Between 2006 and 2009 land cover and vegetation structure data were collected across the northern forests of Aceh covering the Ulu Masen ecosystem and adjacent forests. The survey protocol used for vegetation data collection was adapted from a larger collaborative survey originally aimed to monitor large mammals. The original survey design was based on a patch occupancy model (MacKenzie *et al.*, 2002) and developed in collaboration with the Wildlife Conservation Society (Wibisono, 2007). This method is used to estimate the proportion of area occupied by a focal species given a set of discrete habitat patches. Therefore, the total survey area was divided into 17 x 17 km grids (289 km^2), which is assumed to cover a single tiger or elephant range. Within a single grid, a 40km reconnaissance transect was walked by foot (figure 3.3) and vegetation parameters were recorded every kilometre as listed below. Because transects followed the path of least resistance through the landscape, a randomisation facet was introduced by necessitating transects to pass through two points randomly allocated to each grid cell.

Six land cover classes (forest, non-forest, secondary forest, plantation, gardens, grassland) and estimates of canopy cover at three different strata (ground < 1m, understory 1 - 5m , canopy >5m) were recorded at each sample point. Transect data and the position of sample points were recorded using a GARMIN 60-CSx handheld GPS. All field work and data collection was conducted by field teams consisting of FFI-AFEP staff, people from local communities and local district forestry rangers. Training on basic navigation-and data collection techniques were provided by E.Rood as part of FFI-AFEP project activities. Additional training on mammal survey techniques was provided by Hariyo Wibisono from the Wildlife conservation Society.

Figure 3.3 Location of survey grids and related reconnaissance transects across the northern forest of Aceh covering the Ulu Masen ecosystem.

3.4 Deforestation analysis

3.4.1. Logistic regression

A deforestation risk model for Aceh was built by means of logistic regression (Hosmer & Wang, 1978), as described in chapter four. Logistic regression is an extension, or generalization, of the ordinary linear regression model (OLS). The main difference between OLS and logistic modelling is that while in a OLS the dependent variable is a continuous variable which is normally distributed around the mean, in logistic regression, the dependent variable only takes two values, i.e. true (1) or false (0). This poses a problem when using ordinal parametric methods, like OLS, as these assume equal variances of errors (residuals). Hence, a logistic link function is applied in order to limit the expected outcome of the dependent variable to a range of 0-1 (equation 3.1). As such, logistic regression does not use the dependent variable itself to estimate the model parameters. Instead, it determines changes in the log-odds of the dependent (equation 3.2/3.3), which is modelled as a function of the independent variables just as in an OLS.

$$P(Y = 1) = \frac{e^{\pi}}{1 + e^{\pi}}$$ Equation 3.1

$$Log(odds) = \frac{P}{1 - P} = \pi$$ Equation 3.2

$$\pi = C + \beta_1 x_1 + \beta_2 x_2 + \cdots + \beta_k x_k + e$$ Equation 3.3

Y = Dependent variable (0 or 1)

P(Y=1) = probability of observing a positive (true) outcome.

X_k = value for independent variable k

C = constant value

β_j = coefficient for variable k

e = standardized error

When conducting ordinary linear regression, parameters are estimated by minimizing the sum of squared deviations of predicted values from observed values. For logistic regression, however, it is not possible to produce an unbiased approximation of the least squares estimation as the dependent variable is constrained (0-1). A maximum likelihood estimation is therefore used to solve for the parameters that best fit the data (Efron, 1982). Doing so, logistic regression iteratively assesses the functional relationship between the binomial dependent variable and several categorical or continuous independent variables .

3.4.1.1 Model selection

To assess the effect of different parameters on the probability of deforestation, several models were build using different combinations of predictor variables. To assess which model best described the observed pattern of deforestation, they were compared on the basis of the Akaike Information Criterion (AIC; Akaike, 1974)and Akaike weights (wi) (Burnham and Anderson 2002). This information approach to model selection is originates from the concept of entropy. It is used to assess the trade-off between model accuracy and complexity by comparing models based on their maximum likelihood while penalizing for the number of parameters included in the model (Akaike, 1974). Models that are within two AIC units (DAIC) of the top ranked model with the smallest AIC can be considered as plausible candidate models.

3.4.1.2 Goodness of fit

Model goodness of fit was assessed by applying a Hosmer and Lemeshow's goodness of fit test (Lemeshow & Hosmer, 1982). This test divides the dataset into deciles based on ranked predicted probabilities. Next, a chi-square is calculated from observed and expected frequencies. Once the observed probability value exceeds .05, the null hypothesis that there is no difference between observed and model-predicted values cannot be rejected, implying that the model's estimates fit the data at an acceptable level.

3.5 Elephant habitat use

Elephants are surprisingly cryptic animals which are not readily observed in the field. In many cases, failure to detect elephants at a particular site does not implicitly mean that the environment is not suitable. More often, species absence from a site of potentially suitable habitat is a result of stochastic processes, temporal movements or dispersal barriers (Basille *et al.*, 2008). Yet, failure to effectively establish a species absence while it is in fact present could considerably bias results and lead to false conclusion considering species habitat relations (Hirzel *et al.*, 2002; MacKenzie *et al.*, 2002). As a result, elephant absence resulting from local habitat conditions often cannot explicitly be determined as many suitable habitat patches are unoccupied as a result of other factors such as poaching pressure, elephant captures for management or temporal movements. These restrictions to reliably establish elephant absences as a response to the direct environment should be addressed in order to discriminate between candidate areas of importance for the conservation of a particular species.

Recent developments of different analytical techniques using presence data have enabled to make unbiased inferences about species habitat use even when valid absence data is not available (Pearce & Boyce, 2006). These methods use mathematical algorithms to define the ecological niche of a species in the multidimensional environmental space (Guisan & Zimmermann, 2000). As such, models using presence data only can be used to investigate the relation between a set of environmental predictor variables and the occurrence of a focal species (Brotons *et al.*, 2004; Tsoar *et al.*, 2007; Thorn *et al.*, 2009). Constructing habitat maps based on relative suitability allows the identification of core areas of prime habitat. From a conservation point of view this makes sense since we are interested in locating a range of habitat types and areas which can potentially form suitable elephant habitat.

3.5.1 Study design

In order to generate a representative sample of elephant habitat use and ecological niche requirements throughout the northernmost realm of Aceh a systematic stratified sampling scheme was used. The study area comprised the forest of northern Aceh as well as the adjacent secondary forests, production forests, plantations and small holder gardens. Firstly the total area was stratified into :

1) forest area,
2) secondary forest/plantation
3) agricultural area.

Next the area was divided into four different elevation intervals corresponding to :

1) lowland
2) foothills
3) lower montane
4) montane forest.

These vegetation and elevation categories were then combined to produce a stratification map for the study area. Three survey sites were allocated per elevation class. Within each site plots were allocated based on the dominant vegetation type. Preliminary pilot surveys were conducted from April 2006 to January 2007 to validate the different study sites during which five teams (25 people) were trained in elephant surveying. During February and March 2007, data on elephant distribution was collected over 12 different sites (figure 3.4).

3.5.2 Data collection

Within each site five random plots were selected from which transects were started. Subsequently, five parallel transects were walked each separated by 100 meters, resulting in a total of 25 transects per site and 300 transects over the whole study area. Elephant presence was recorded by means of five meter wide line transects that varied between 200 and 400 m in length. Presence was confirmed if fresh elephant dung (i.e. < 1 month old) was encountered and their geographic locations were recorded using GPS.

Figure 3.4 Location of survey plots across the northern forest of Aceh covering the Ulu Masen ecosystem. Land cover classes shown include (1) Forest: old stand forest with a continuous closed canopy cover (75 – 100%); (2) Plantation: converted forest which still comprises a relatively dense canopy cover (25-75%); Gardens: grasslands, small holder gardens and bare land (< 25% cover).

3.5.3 Data analysis

Elephant habitat suitability maps were calculated using Ecological Niche Factor Analysis (Hirzel *et al.*, 2002). Ecological niche factor analysis (ENFA) is a relatively new multivariate approach, similar to PCA, developed to predict habitat suitability when absence data for the species are not available. ENFA compares the distribution of the presence observations in the multidimensional space of the environmental variables to the entire study area (Hirzel *et al.*, 2002). The suitability is based on functions that define (1) the *marginality* of the species: how the species mean differs from the mean

of the entire area (figure 3.5a), and (2) the *specialization* of the species: ratio of the overall variance to the species variance (figure 3.5b). After the first factor (marginality factor) is extracted, multiple orthogonal specialization factors can be calculates from the transformed dataset (Hirzel *et al.*, 2002). The number of factors extracted by the ENFA algorithm can be determined by comparing factor eigenvalues to the McArthurs broken stick distribution (Hirzel *et al.*, 2002) or by selecting factors with an eigenvalue of more than one.

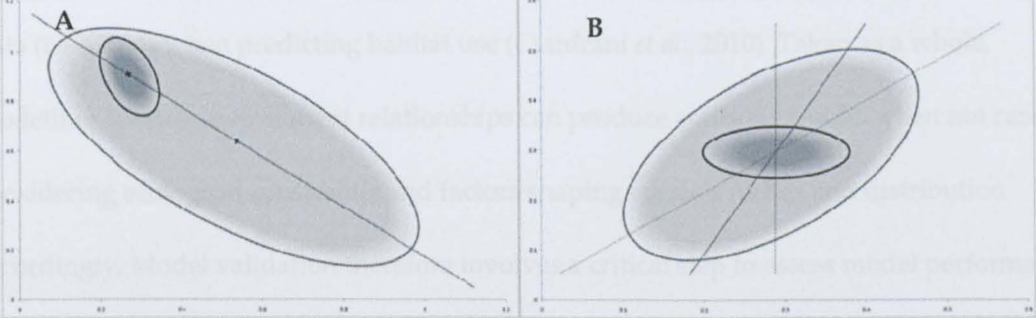

Figure 3.5 Schematic representation of the Ecological Niche Factor Analysis algorithm. First, a marginality factor (μ) is extracted from the original data by maximizing the distance between the species average conditions (dark ellipse) and the average available habitat (light ellipse) (A). Next, a specialization factor (γ) is calculated from the transformed dataset by maximizing the ratio between the species variance and habitat variance (niche width) (B).

A habitat suitability index can be calculated based on several methods. Here we used the geometric mean approach which calculates habitat suitability as the geometric distance between each point in ecogeographical space to all presence records (Hirzel & Arlettaz, 2003). Consequently, the more denser species presence are in ecogeographical space the higher the habitat suitability will be . This method has the advantage that it provides a good trade off between model accuracy and discriminative capacity (Hirzel & Arlettaz, 2003).

Some studies have shown that presence-only based niche algorithms tend to overestimate the habitat suitability (Tsoar *et al.*, 2007; Chefaoui & Lobo, 2008). This is primarily caused by the fact that presence only methods do not use true absences to distinguish unsuitable areas. This can lead to over-predictions when modelling habitat suitability of species naturally occupying a wide range of habitat types or when species habitat selection is not consistent within the study region (Titeux *et al.*, 2007; Basille *et al.*, 2008). Conversely, if habitat use by a certain species is not stationary, for example if wildlife populations are being displaced by human perturbations, the use of absences could result in unreliable predictions and omission of suitable habitat (Hirzel *et al.*, 2001). Given a non-equilibrium situation, ENFA has been shown to outperform methods using presence absence data (i.e. GLM) when predicting habitat use (Cianfrani *et al.*, 2010). Taken as a whole, modelling species environment relationships can produce spurious results when not carefully considering ecological constraints and factors shaping specie's niches and distribution accordingly. Model validation therefore involves a critical step to assess model performance and to determine the reliability of the results.

3.5.4 Model Validation

A wide range of validation techniques are available to estimate model performance and accuracy including the kappa statistic (Cohen, 1986), Boyce cross-validation (Boyce *et al.*, 2002) and the receiver operation characteristic (ROC-AUC, Fielding & Bell, 1997). Here, the predicted habitat suitability model was validated using both a continuous Boyce validation technique (Boyce *et al.*, 2002; Hirzel *et al.*, 2006) as well as the ROC-AUC. The Boyce validation statistic is based on a confusion matrix and is calculated as the ratio between the number of observed elephant presences and the number of presences expected based on a random distribution (Boyce *et al.*, 2002). Good model performance is indicated by a high correlation between the habitat suitability score (HS) and the ratio of observed and expected values (Boyce *et al.*, 2002). ROC-AUC score is calculated as the fraction of true positives or false positives over to estimate model predictive power (Pearce & Boyce, 2006).

3.6 Elephant Crop raiding

The occurrence of human-elephant conflict has long been recognized to be a common problem in Aceh, with the first reports dating back to the start of the 20th century (Heurn, 1929). In Aceh, however, no systematic surveys have been conducted to assess the intensity or scale of the problem. The political instability in the region as well as the large amount of resources needed to address the issue of human-elephant conflict have constrained the efforts to collect timely data (Aceh, 2009; Azmi *et al.*, 2009). However, being a high profile species, reports on the occurrence of human-elephant conflict in Aceh have received a relatively large interest in national newspapers. Consequently these media provide a consistent and long-term overview of the occurrence of human-elephant conflict.

3.6.1 Study design

Since no systematic and consistent data on the occurrence of human-elephant conflict is available, no inferences about trends in time can be made. Also, the nature of human-elephant conflict ranges from mere apprehension towards elephants in areas where elephants are scarce, to crop raiding, encounters and ultimately lethal casualties, in areas where elephants are permanently present. Consequently, human-elephant conflict incidents are believed to occur as a result of area specific conditions elephant habitat and elephant population status (Hoare, 1999; Sitati *et al.*, 2003; Hedges & Gunaryadi, 2010). The pattern of human-elephant conflict events observed across the Ulu Masen ecosystem and adjacent areas (figure 3.6) therefore allows to assess how local patterns of land use and habitat configuration could influence the occurrence of human-elephant conflict. Hence reports on human-elephant conflict events were collated and analysed using a set of landscape predictors shaping elephant habitat use (chapter four) and disturbance to predict where human-elephant conflict is likely to occur (chapter five).

3.6.2 Data collection

Data on human-elephant conflict throughout Aceh were collected by means of three different information sources:

1) Archived reports from the Indonesian conservation agency (BKSDA) in Aceh (1985-1998)
2) Incidental reports published in provincial or national newspapers. (Serambi, Waspada, Antara, 2000-2007)
3) Incidents reported to FFI-district coordinators (2007-2008)

Between 1985 and 1998, 62 records of HEC were reported from the whole of Aceh, all of which originated from interview reports with local communities. From the years 2000 to 2007 another 316 incident records were collected from the Indonesian conservation agency (43) and newspaper archives (273). Since most of the reports published in newspapers were collected ad hoc, when conflict incidents escalated, none of the reports consistently reported about the intensity of HEC intensity. Therefore, these data could not be used to directly relate the intensity of human-elephant conflict to environmental parameters. The available data, however, does provide valuable information on the spatial distribution and local abundance of crop raiding by elephants around the Ulu Masen forest block.

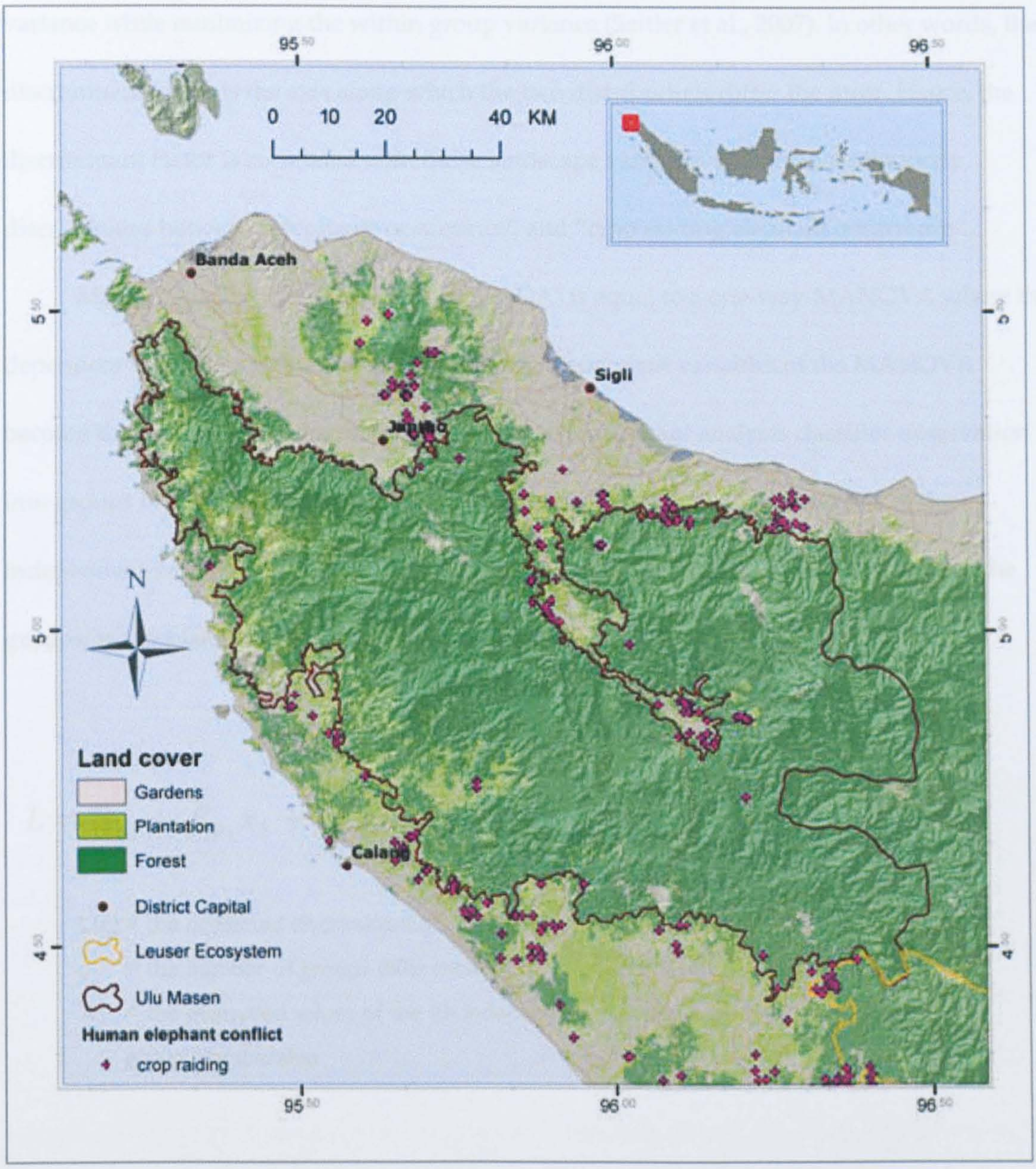

Figure 3.6 Crop Raiding incidents by elephants (N=316) between 200 and 2008.

3.6.3 Data analysis

To investigate which landscape factors are most essential to the occurrence of human-elephant conflict a discriminant analysis was performed comparing elephant crop raiding localities to elephant distribution data, which therefore served as a control group. Discriminant function analysis is used to determine which variables discriminate between two or more naturally occurring groups. It computes a factor that maximizes the between group variance while minimizing the within group variance (Sattler et al., 2007). In other words, the discriminant factor is the axis along which the two distributions differ the most. Hence, the discriminant factor is correlated with those landscape variables which most effectively discriminates between "elephants occurrence" and "crop raiding elephant occurrence".

Mathematically, discriminant analysis (DA) is equal to a one-way-MANOVA where the dependent variable is formed by classes, and the dependent variables of the MANOVA become the predictors for discriminant analysis. Discriminant analysis classifies observation into groups based on discriminant functions, derived from linear combinations of the independent variables (equation 3.4), which yield the largest mean differences between the groups(McLachlan, 2005).

$$L = \beta_{p_0} + \beta_{p_1}x_1 + \beta_{p_2}x_2 + \cdots + \beta_{p_n}x_i \qquad \boxed{\text{Equation 3.4}}$$

L(p) = the predicted discriminant score for group p

p = the number of groups differentiated by the discriminant functions

X = the measured values of the ith independent variables used to predict group membership

Since the occurrence of crop raiding elephants encompasses a nested subset of the total distribution of elephants, a certain amount of overlap is expected to exist in the environmental conditions driving both spatial patterns. Hence, it can be hypothesized that environmental predictors which significantly discriminate between crop raiding and non-crop-raiding elephants are those which are most likely to drive the occurrence of conflict. Here, the same set of predictors which was found to shape elephants' niche were used in the analysis. Additionally, two parameters: 1) proportion of forest logged between 1990 and 2007 and 2) secondary forest in a 10 km surrounding present in 2007, were included to assess their discriminative power on the occurrence of crop raiding by elephants.

3.7 Elephant extinctions

Relatively little is known about the current status of the species on Sumatra. A study conducted by Hedges *et al.* (2005) to estimate elephant population sizes in the southern Sumatran province of Lampung, revealed that only three of the twelve populations, totalling 550-990 elephants living in Lampung province during 1980s (Blouch & Haryanto, 1984; Santiapillai & Jackson, 1990; Hedges *et al.*, 2005), were still extant in 2003. Similar reports have been made for other areas in Sumatra in including Riau province (Uryu *et al.*, 2008) and Benkulu province (Wahdi Azmi, FFI; pers comm.). From this work it has become clear that elephant populations are under continuous threat of displacement resulting from deforestation and populations are likely to decline as habitat conversion continues (Hedges *et al.*, 2006; Choudhury *et al.*, 2008; Hedges & Gunaryadi, 2010).

To provide insight in the processes leading to local population extirpations, the current and past distribution of elephant ranges across Sumatra are compared to the pattern of deforestation and anthropogenic parameters. To identify which factors are most likely to have caused local extinctions of elephants, logistic regression modelling was used. Finally we assess which elephant populations are currently most likely to be prone to extinction and whether current protected areas do provide the necessary means to protect elephant populations in the future.

3.7.1 Study design

In 1984 Blouch *et al* conducted a Sumatra-wide status assessment of elephant populations across the island during the late 80s. This data was subsequently reviewed and published in a IUCN report by Santiapillai and Jackson in 1990 (Santiapillai & Jackson, 1990). The distribution map produced by Santiapillai and Jackson was digitized in a GIS and served as the baseline reference of elephant distribution across Sumatra in 1990. To assess the validity of the map it was compared to several descriptions of the historic elephant distribution available from a variety of resources (Strasters, 1914; Heurn, 1929; Pieters, 1932; Hedges *et al.*, 2005; Rood, 2006; Uryu *et al.*, 2008) and updated where necessary.

The baseline elephant distribution map for 2005 was derived from the Southeast Asian Mammal Databank (Catullo *et al.*, 2008). The elephant distribution data extracted from the SAMD database was compared to the elephant distribution maps published in the Indonesian national Elephant Action plan as well as different Conservation NGO's working on Sumatra (Kinnaird *et al.*, 2003; Hedges *et al.*, 2005; Rood, 2006; Uryu *et al.*, 2008) but showed no significant anomalies and was therefore believed to correctly represent the distribution of elephants across Sumatra in 2005.

3.7.2 Data analysis

Land use and Land Cover Change (LULCC) denotes an important subject in global environmental change and species environment interactions. Hence, empirical methods using generalized linear models have become some of most frequently used models to simulate the effect of land use pattern and its changes on species distributions. The risk of extirpation of elephants occurring in Sumatra was assessed by means of logistic regression as described in the section *logistic regression* above (section 3.4.1).

3.7.3 Spatial autocorrelation

A common challenge to the analysis of spatial distribution patterns is the occurrence spatial autocorrelation. Spatial autocorrelation occurs when spatially adjacent observations are not independent of each other and can lead to an increased risk of a type I error (i.e. falsely reject null hypothesis) (Lichstein *et al.*, 2002; Dormann *et al.*, 2007). Spatial autocorrelation (SA) occurs when nearby points in space have more similar values than would be expected based

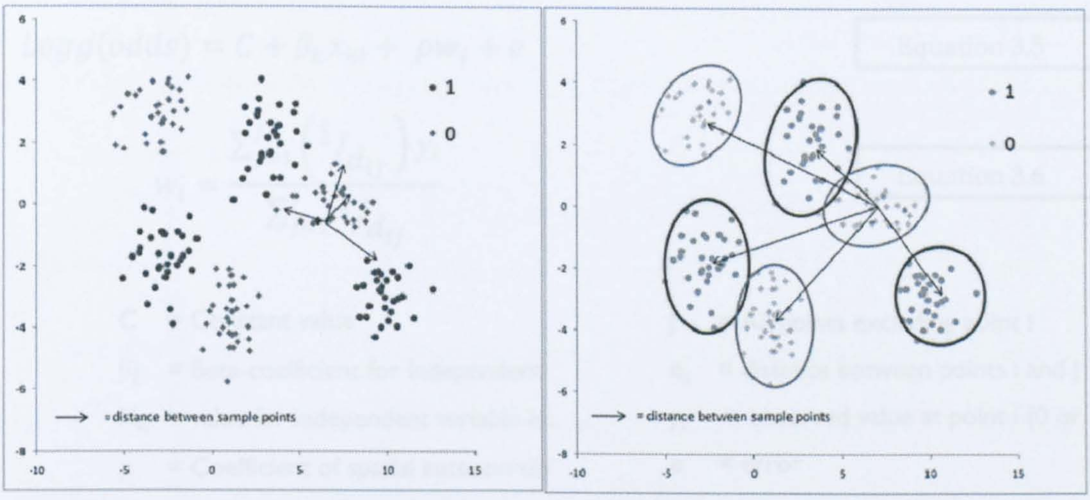

Figure 3.7 Schematic representation of spatial autocorrelation given the distribution of a set of binomial (0,1) data points (A). If sample points are selected at a close distance (eg. less than the scale at which spatial processes occur; ellipse), they are likely to violate the assumption of independence leading to Type I errors (A). If sample points are chosen at an appropriate distance (eg distance at which the spatial process occurs; ellipse), they are expected to be independent and therefore can be used for statistical modelling.

on chance (figure 3.7A). The occurrence of SA is often driven by various causes that can be either exogenous (e.g., autocorrelated environment) or endogenous (e.g., conspecific attraction, dispersal limitations; Lichstein *et al.*, 2002). However, many existing methods like logistic regression, often ignore the fact that spatial autocorrelation occurs within spatial data, which affects the goodness of fit and accuracy land use models (Lichstein *et al.*, 2002; Dormann *et al.*, 2007; McPherson & Jetz, 2007). One way to prevent spatial autocorrelation influencing results is to select data points at sufficiently large distances so that SA does not affect data (figure 3.7B). Alternatively, distance relations are incorporated into the model to explicitly correct for spatial dependence (equation 3.6/3.7). The use of autologistic modelling can be beneficial as the occurrence of SA which can be quantified which could provide additional information about the system studied. (Lichstein *et al.*, 2002).

$$Logg(odds) = C + \beta_k x_{ki} + \boldsymbol{\rho w_i} + e$$

Equation 3.5

$$w_i = \frac{\sum_{j=1}^{J} \left({}^{1}/_{d_{ij}} \right) y_i}{\sum_{j=1}^{J} {}^{1}/_{d_{ij}}}$$

Equation 3.6

C = Constant value

βj = Beta-coefficient for independent

X_{ki} = value for independent variable k

ρ = Coefficient of spatial autocorrela

w_i = auto-covariate for point i.

J = All points excluding point i

d_{ij} = distance between points i and j

y_i = observed value at point i (0 or 1)

e = error

3.8 Forest protection modelling

Recent political and economic developments in Aceh have resulted in an explosive pressure on the remnant forests of Aceh (Rood, 2009; Rood *et al.*, 2009). Research on the investment of conservation resources to prevent deforestation is particularly relevant because strategic protection might not only provide direct benefits to these threatened forests, but also provides critical habitat to a range of threatened species, protects environmental services (DeFries *et al.*, 2007a; Van Beukering *et al.*, 2008), and functions as a buffer (Kinnaird *et al.*, 2003; Nyhus & Tilson, 2004; DeFries *et al.*, 2007a) .

3.8.1 Study design

To investigate the potential effectiveness of conservation management intervention in and around the northern forest of Aceh, Indonesia, patterns of deforestation were modelled and different conservation strategies were tested. Firstly, the drivers of deforestation were determined by means of logistic regression (see chapter 4) and then use this model to estimate deforestation patterns in the absence of active forest protection (chapter 8). Secondly, the impact of a deforestation is under a number of forest protection strategies is assessed. Forest protection efforts that is aimed to (1) protect existing protected areas, (2) protect the most vulnerable patches of forest, (3) prevent encroachment through expansion of newly opened areas and (4) reducing deforestation pressure by applying buffer zones as well a combinations of these strategies are assessed.

3.8.2 Data analyses

The effectiveness of each conservation scenario was assessed by conducting a survival analysis (Breslow, 1975). This approach is particularly useful as it includes the average time to deforestation as the dependent variable enabling the comparison of different in deforestation rates. Survival analysis uses the time for an event to occur, in combination with appropriate covariates, to estimate the hazard- or failure rate (Breslow, 1975; Greenberg *et al.*, 2005). A parametric regression model was fitted to the survival data. This method has the advantage that it considers data in which a number of censored pixels did not experience an event of interest (e.g. deforestation) within the time span of the study (100 years). To investigate the effect of each conservation scenario on the average hazard rate, this was included as a covariate in the analysis. Since we were also interested in the change in the deforestation rate over time, a Weibull distribution (Pinder *et al.*, 1978) was used as it allows the hazard to change as a function of time (equation 3.8). To determine the change in hazard rate (i.e. the deforestation rate) over time a scale parameter (σ) is added to the model. If $\sigma > 1$, the deforestation rates decrease over time (equation 3.8.).

$$\frac{t^{(\frac{1}{\sigma}-1)} e^{-\alpha_i/\sigma}}{\sigma}$$

Equation 3.7

t = Time interval

σ = Scale parameter

α_i = Linear function of predictor values

3.9 Auxiliary spatial data acquisition

3.9.1 Elevation data

An 90x90 meter resolution raster elevation map of Sumatra was obtained from the Shuttle Radar Topography Mission elevation, which was downloaded from the Global Land Cover Facility Earth Science Data Interface (http://glcfapp.umiacs.umd.edu:8080/esdi/index.jsp). This elevation raster layer (Digital Elevation Model, or DEM) was resampled to a 100x100 meter resolution and subsequently used to calculate additional landscape descriptors including: (1) *landscape ruggedness*: standard deviation of all elevation values in a circular surrounding with radius D(m) from the focal cell. (2) *Landscape curvature or convexity*: difference between the elevation value in a focal cell and the average elevation of all cells in a circular surrounding with radius D(m) (3) the steepest slope from a focal cell to any adjacent cell calculated as the percent decline.

3.9.2 Climate data

Nineteen different climate data grids of 1x1 km covering the whole of South East Asia were obtained from the Worldclim world climate database (http://www.worldclim.org/). In order to make each data layer compatible for analysis, each raster layer was sub sampled to a 100m resolution using bilinear interpolation.

3.9.3 Administrative and infrastructure data

The position of settlements and roads was obtained from 1:50,000 maps produced by Indonesian National Coordination Agency for Surveys and Mapping (Bakosurtanal, 1979). Additionally, a digital map of Indonesian cities in 2009 was obtained from the National Geo-Spatial Intelligence agency (http://earth-info.nga.mil/gns/). For compatibility, all the spatial data layers were converted UTM47N projection with a 100x100m resolution raster format.

4.1 Introduction

Even though substantial international funding has been invested to protect rainforests, global deforestation rates show little sign of improvement (Achard, 2007). While the ongoing loss of tropical rainforests represents one of the most serious threats to biodiversity (Sodhi, 2008; Sodhi *et al.*, 2010) recent discussions on tropical deforestation have focussed on its contribution to climate change (Achard *et al.*, 2007; DeFries *et al.*, 2007b; Linkie *et al.*, 2010). The International Panel on Climate Change (IPCC) estimates that destruction of forests contributes around 18% of the greenhouse gas emissions entering Earth's atmosphere (IPCC, 2007). Failure to avoid this deforestation is predicted to greatly accelerate global warming (Fearnside, 2000; Gullison *et al.*, 2007). In response, forest conservation initiatives are considering policy approaches for Reducing Emissions from Deforestation and Degradation' (REDD), which essentially pays governments to reduce deforestation below an estimated background rate (Blom *et al.*, 2010). These schemes require reliable baseline data on their forest stocks, with varying levels of detail. Identifying the location and rates of forest loss is important information for law enforcement agencies responsible for mitigating this threat (Linkie, 2010).

In Aceh indeed, and Indonesia at large, illegal logging and forest clearance poses a serious threat to ecosystem service functioning and therefore human well-being (van Beukering *et al.*, 2003; Van Beukering *et al.*, 2008; Bradshaw *et al.*, 2010). In the aftermath of the devastating tsunami and protracted civil conflict, there was genuine and legitimate concern about the environmental impacts of the reconstruction and development processes. Also, with peace now having been achieved in Aceh, many former farmlands that had previously been abandoned during the conflict period and since turned back to forest, were being reopened for cultivation. Consequently, Aceh faced an unprecedented demand for its natural resources, such as timber, and its space for creating new farmland. With an increase in demand for timber, in part to support the tsunami reconstruction efforts and in part to provide employment, there has been a significant increase in the number of loggers entering the forests. The forests of Aceh are rich in tropical hardwood trees, such as semaram (*Palaquium semaram*), merbau (*Intsia bijuga*) and several species of meranti (*Shoreo spec.*) , which can obtain a high price on international markets and therefore make logging a lucrative business, for those trading, often outside of Aceh. The Government of Aceh's initiative for Reducing Emissions from Deforestation and Degradation (REDD) in Ulu Masen has brought significant international attention to this protection ecosystem. As proper landscape management and well managed forests provide a safeguard of essential environmental services such bioremediation and disaster relief (floods and landslides), the properties provide one of the strongest arguments for forest protection (Van Beukering *et al.*, 2008).

Remotely sensed data offers an inexpensive and reliable option to estimate forest cover and forest cover change over extended periods. Identifying the location and rates of forest loss is important information for law enforcement agencies responsible for mitigating this threat . Because drivers of deforestation are often site and scale specific (Lambin & Geist, 2003) patterns of deforestation should be analyzed at regional scales to predict *in situ* threats and identify the local area at risk of deforestation (Linkie *et al.*, 2010). A wide range of factors driving deforestation, acting on different spatial scales, have been suggested by various authors. The expansion of oil palm estate has frequently been mentioned as the major factor driving deforestation dynamics across the South East Asian region (Achard *et al.*, 2007; Hansen *et al.*, 2009b) as well as in Indonesia (Jepson *et al.*, 2001; Sandker *et al.*, 2007). At a local or sub-national level, a consistent core set of predictors have been found to affect deforestation including: land use and tenure, local administration, soil, elevation, slope, distance to forest edge, distance to roads and distance to nearest settlement (Linkie *et al.*, 2004; Andam *et al.*, 2008; Gaveau *et al.*, 2009c; Gaveau *et al.*, 2009a; Linkie *et al.*, 2010).

To provide a reliable and up-to-date assessment of forest cover change and current threat in and around the forests of Northern Aceh the present and past forest cover in Aceh was estimated for the years 2005-2009 using Landsat satellite imagery. Next, a logistic regression analysis, including anthropogenic as well as topographic parameters was used to identify which factors influence local deforestation processes and to predict local patterns of deforestation.

4.2 Methods

4.2.1 Study area

Deforestation rates and patterns were investigated using data from the proposed protected area of Ulu Masen (see section 2.2.4). The forest area spans six districts (Aceh Barat, Aceh Besar, Aceh Jaya, Bireuen, Pidie and Pidie aya) and adjoins the districts of Aceh Tengah and Nagan Raya (figure 4.1). Forest cover in 2009 and forest cover change were separately estimated for the six Ulu Masen districts together with Nagan Raya and the eastern forests of Aceh Tengah. The forest boundary used for the analysis consists of forest inside the proposed Ulu Masen boundary (Pasya *et al.*, 2007) and adjacent forest that extends outside the border. Peat swamp and coastal mangrove forest, being subjected to other deforestation dynamics, were omitted from analyses conducted within this study.

4.2.2 Data processing

Annual forest cover estimates for the years 2005-2009 were produced based on the classification of 26 orthorectified Landsat 7 ETM+ satellite images using a maximum likelihood classification algorithm (see section 3.3). To ensure accurate vegetation classification and to reduce erroneous results caused by noise and bias, anomalies such as water bodies and clouds were masked from the original images. As water is a strong absorber of near infra red waves (Mather, 1987) band 4 was used to delineate water bodies. Strong atmospheric haze and clouds were extracted from the images using a linear combination of

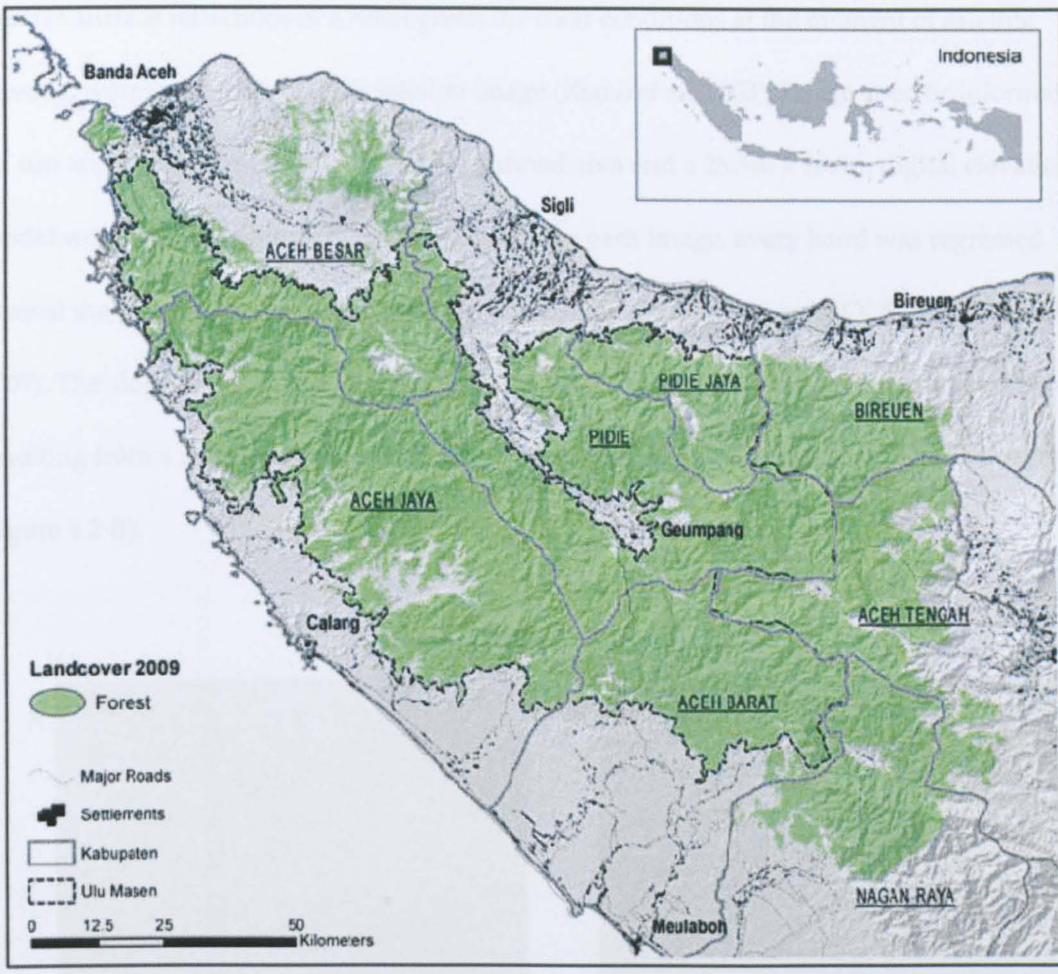

Figure 4.1 Forest cover over the various administrative districts across the Ulu Masen Ecosystem.

the first and seventh spectral band which define reflectance of blue-green light and mid infrared light, respectively.

The Ulu Masen forest block largely covers the northernmost tip of the Bukit Barisan mountain range which generates differences in topographical relief throughout the study area. Consequently, there are various patterns of hill shading present on the images. As a result, a single land cover type can produce different patterns of spectral reflectance, or spectral signatures, depending on site-specific topographical orientation and slope (figure 4.2-A). These variations will lead to erroneous results if not controlled for during image classification. To remove shade effects from the images, different hill shade models, which mirror the relative surface reflections of a relief given the solar conditions at the moment of satellite passing, were produced for each satellite image (Riano *et al.*, 2003). Image specific information on sun angle and azimuth relative to the scanned area and a 28.5m x 28.5m digital elevation model were used to produce the models. Next, for each image, every band was regressed against the expected illumination values using a ordinary least squares (OLS) regression (Zar, 2007). The slope and gain of the regression line were then used to mask illumination effects resulting from a mixture of different aspects and slopes existing within the original image (figure 4.2-B).

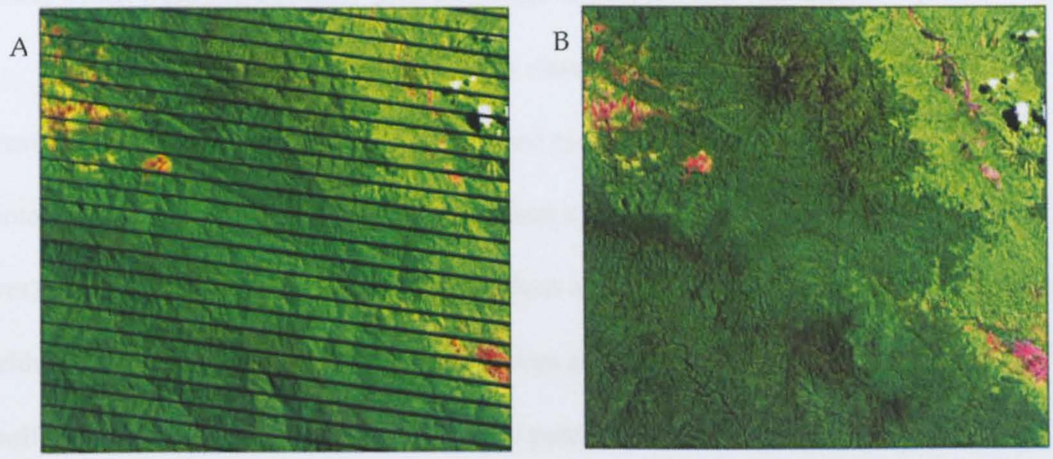

Figure 4.2 (A) Close up of a raw LANDSAT ETM composite image (5-3-4) with stripes (SLC-off).
(B) Same close up after stripe filling and topographic illumination correction.

4.2.2.1 Band selection

To determine which combination of spectral bands most effectively separated variations in reflectance from bare areas, deforested areas and forested areas, a discriminant analysis was performed. Five spectral bands (2/3/4/5/7) and a Normalized Difference Vegetation Index (NDVI; normalized ratio between red and near infrared light using band 3 and 4 respectively) were used to define the three land cover classes described above.

4.2.2.2 Image classification

Since different landscape elements reflect and absorb different parts of the solar light spectrum, they produce explicit spectral signatures that can be used for enhanced image classification (Mather, 1987). Training areas, used to extract spectral signatures for different land cover classes, were collected by FFI field teams between 2006 and 2009 (see section 3.3.1). Vegetation data consisted of 1244 point records which provided information on land cover classes that were encountered during field surveys (e.g. primary/undisturbed forest, secondary/disturbed forest, plantation, garden, grassland or bare soil, as well as relative forest cover classes (0-25%, 25-50%, 50-75% and 75-100%). The data set was split into two equal sized subsets which were subsequently used for image calibration and validation.

Each Landsat scene was then separately classified into three land cover classes: (1) Forest: old stand forest with a continuous closed canopy cover (75 – 100% cover); (2) Plantation: converted forest which still comprises a relatively dense canopy cover (25-75% cover); Gardens: grasslands, small holder gardens and bare land (< 25% cover). A maximum likelihood algorithm was used to classify the data as explained in section 3.3.1. After classification small scale anomalies, defined as patches of 1 ha or less (<12 cells), were removed by merging them with their respective surroundings based on the longest shared border. The resulting forest cover estimates for the years 2006 - 2009 were validated using ground-truthed control points. For the years 2006, 2008 and 2009, a sample of 200 independent ground truthed records and for the year 2007 another 100 ground control points were used to validate the image interpretations of 'forest' and 'non-forest'. Since no validation data was available for the year 2005, the accuracy of the estimate was assumed to be within the same range of those of the subsequent years.

4.2.3 Spatio-temporal analyses

Several different methods to calculate deforestation rates have been proposed over the last decade (Puyravaud, 2003). Many of these methods are based on standardized deforestation ratios facilitating comparisons of deforestation rates globally. However, to enable direct interpretation of these results and to then compare these deforestation rates with those from elsewhere in Sumatra and Borneo, annual deforestation rates were calculated as: (1) Percentage forest loss per year (%/yr), defined as the proportion of forest lost against a baseline forest cover estimate of the previous year; and (2) Forest hectares loss between consecutive years (ha/yr). Forest coverage data layers were overlaid within the GIS to determine the location and rates of deforestation over successive years (i.e. from 2005-2009) for the entire study area and for each focal district.

4.2.4 Deforestation modelling

To investigate deforestation risk, the occurrence of deforestation was analyzed by means of logistic regression (see section 3.4.1) using topographic and anthropogenic parameters as predictors. Previous studies have emphasized the importance of area accessibility and human pressure to predict deforestation patterns (Kinnaird *et al.*, 2003; Gaveau *et al.*, 2009c; Linkie *et al.*, 2010). A GIS dataset containing two topographic parameters (elevation, slope), two anthropogenic parameters (distance to nearest village, distance to nearest roads) and distance to forest edge was produced. Elevation data was obtained from the Shuttle Radar Topography Mission, which was used to produce the slope layer. The forest edge information was taken from the 2005 forest cover classification. The position of settlements was obtained from 1:50,000 maps produced by Indonesian National Coordination Agency for Surveys and Mapping. Section 3.9 provides a detailed description of the different sources addressed to obtain topographic and administrative data for the area. For compatibility, all the spatial data layers were converted UTM47N projection with a 100x100m resolution raster format.

The forest risk model was determined using data from 100 forested points that were cleared between 2005 and 2009 and another 100 points that remained forested during this period. Each set of points was randomly selected using the Hawth's tools ArcGIS extension (http://www.spatialecology.com/htools/tooldesc.php). To reduce the likelihood of spatial autocorrelation (see also section 3.7.3), points were selected with a minimum distance of two km between points. These points were then used in ArcGIS to extract the physical covariates values at each of the 200 points. These spatial variables were then imported into SPSS v.16 statistical software package (SpSS Inc., Chicago, IL) and log-transformed to prevent outliers from having a disproportionate influence on the result of the analysis. A Spearman's rank correlation was conducted to test for collinearity between the four spatial covariates. Non-independence was identified between slope and elevation, as well as between distance to nearest road and distance to nearest village. Hence, a data reduction technique (PCA) was performed to produce uncorrelated variables for both the combined topographic variables as well as the anthropogenic variables. Factors with an eigenvalue of more than one were extracted and used in the subsequent analysis. This resulted in one factor describing the topographic variation present in the Ulu Masen area (eigenvalue: 1.461; 73% variance explained) an one factor describing the anthropogenic variation (eigenvalue: 1.524; 76% variance explained).

Multiple Logistic regression analyses were performed to determine which parameters, individually and in combination, best explained deforestation across the study area. Models were compared based on the Akaike Information Criterion (c.f. Burnham & Anderson, 2002; Burnham & Anderson, 2004). Models that were within two AIC units (ΔAIC) of the top ranked model with the smallest AIC were considered as plausible candidate models and their results discussed. The performance of a final regression model was then evaluated by calculating the area under the curve of receiver operating characteristics (ROC) plots. The presence of spatial autocorrelation in the model was then tested by calculating Moran's I statistic using the SAM vs. 3.0 software package (Rangel *et al.*) Subsequently, a spatially explicit deforestation risk model was constructed, using the parameters estimates for each predictor variable included in the final logistic model.

4.3 Results

4.3.1 Band selection

The results of the Discriminant analysis show that a combination of Bands 2/3/4/5/7 and the additional NDVI layer, effectively separated bare soil from deforested and forested areas. Spectral bands 3 and 4 (0.63-0.69 μm and 0.76-0.90 μm), which are absorbed by chlorophyll and water, respectively, proved to be the most reliable predictors for separating primary forest from other vegetation types, while Bands 5 and 7 were included to distinguish between vegetation and barren soils (table 4.1). Because considerable overlap existed between secondary regrowth or and undisturbed forest, these groups were merged in the final classification.

Function	Eigenvalues			Structure coefficients						Significance		Class centroids		
	Score	(%)	R	b2	b3	b4	b5	b7	NDVI	Wilks	P	Bare	Sec	Forest
1	4.98	(77.0)	0.91	-0.78	-0.83	-0.23	-0.46	-0.50	0.47	0.07	<0.0001	-5.08	-1.69	1.58
2	1.49	(23.0)	0.77	-0.10	0.32	-0.77	-0.41	-0.11	-0.82	0.40	<0.0001	2.32	-1.74	0.39

Tabel 4.1 Results of the multivariate Discriminant analysis using six Landsat derived spectral bands to distinguish three land cover classes: Bare, Secundary forest/Degraded forest (Sec) and Forest. The eigenvalues and the canonical correlation coefficient (R) as well as the structure coefficients between each function and the original variables are given.

The result of the final classification were found to be highly accurate (> 90%) with the kappa statistic being >0.82 statistic for all years (table 4.2). For every year, the proportion of control points correctly predicted as non-forest land cover (specificity) was generally lower as compared to the proportion of points correctly predicted as forest (sensitivity).

Year	Sensitivity	Specificity	Accuracy	Kappa
2009	92%	91%	92%	0.83
2008	98%	88%	94%	0.88
2007	98%	81%	93%	0.83
2006	98%	80%	92%	0.82

Table 4.2. Accuracy of the predicted forest cover maps based on a maximum likelihood estimations of Landsat data. Sensitivity: correctly predicted forest cover (eg true positives). Specificity: correctly predicted non-forest (eg true negatives). Accuracy: total correctly predicted. Kappa: chance corrected proportional agreement.

4.3.2 Spatial patterns of forest cover change

Comparing the 2005 and 2009 forest cover maps showed that a total of 36600 ha of forest had been cleared during that period, equivalent to a mean deforestation rate of 1.11%/yr ±0.513 (±95% C.I.) or 113.1 km^2/yr ±5.33. As a result forest covered 9924.7 km^2 in 2009 (figure 4.3a, table 4.3). Comparing deforestation rates across the study area districts revealed that the mean rates recorded in the non-Ulu Masen districts of Aceh Tengah (1.34%/yr±0.68) and Nagan Raya (1.18%/yr±1.809), but also Pidie Jaya (1.41%/yr±0.754) were higher than the study area average (1.11%/yr±0.513, table 4.3). The lowest deforestation rates were recorded in Aceh Besar (0.78%/yr ±0.437), then Bireuen (1.02%/yr±0.521) and Pidie (1.10%/yr±0.466). The most rapidly cleared forest type was lowland (2.1%/yr), followed by sub-montane (0.6%/yr), hill (0.4%/yr) and then montane (0.3%/yr).,

Table 4.3. Remaining forest cover (ha) and annual forest loss (ha and %) for the Ulu Masen study area and two adjacent districts from 2005-2009.

District	***2005***	***2006***		***2007***		***2008***		***2009***		**Total**	
	Forest Cover	Forest Cover	Deforested	Forest Cover	Deforested	Forest Cover	Deforested	Forest Cover	Deforested	Deforested	Average annual deforestation
	*1000 ha	*1000 ha	*1000 ha (%)	*1000 ha	*1000 ha (%)	*1000 ha	*1000 ha (%)	*1000 ha	*1000 ha (%)	*1000 ha (%)	% year^{-1}(± 95% CI)
Aceh Barat	119.8	119.1	0.7 (0.6)	117.2	1.9 (1.6)	114.8	2.4 (2.1)	114.3	0.5 (0.42)	5.5 (4.6)	1.18 (±0.788)
Aceh Besar	136.6	136.1	0.5 (0.4)	135	1.1 (0.8)	133.1	1.9 (1.4)	132.4	0.7 (0.52)	4.3 (3.1)	0.78 (±0.437)
Aceh Jaya	254.6	252.8	1.8 (0.7)	250.1	2.7 (1.1)	246.8	3.3 (1.3)	243.6	3.2 (1.3)	11 (4.3)	1.1 (±0.277)
Pidie	207.7	207	0.7 (0.4)	204	3 (1.4)	201.1	2.9 (1.4)	198.8	2.4 (1.19)	9 (4.3)	1.1 (±0.466)
Pidie Jaya	58.9	58.6	0.3 (0.5)	57.6	1 (1.7)	56.3	1.3 (2.3)	55.6	0.6 (1.15)	3.3 (5.6)	1.41 (±0.754)
Bireuen	66.6	66.1	0.5 (0.8)	65.7	0.4 (0.6)	64.5	1.2 (1.8)	64.5	0.6 (0.89)	2.1 (3.2)	1.02 (±0.521)
Nagan Raya	80.4	79.39	0.5 (0.6)	79.8	0.1 (0.2)	76.6	3.1 (3.9)	76.6	0 (0.01)	3.8 (4.7)	1.18 (±1.809)
Aceh Tengah	112.6	111.5	1.2 (1)	109.8	1.7 (1.5)	107.3	2.4 (2.2)	106.7	0.6 (0.61)	5.9 (4.7)	1.34 (±0.68)
Total (UM)	777.6	773.6	4 (0.51)	763.9	9.7 (1.25)	752.1	11.8 (1.54)	744.6	7.4 (0.99)	33 (4.2)	1.07 (±0.429)
Total (No-UM)	259.6	256.99	2.2 (0.85)	255.3	2.2 (0.86)	248.4	6.7 (2.62)	247.9	1.2 (0.5)	11.8 (4.5)	1.21 (±0.941)
Grand total	1037.2	1030.59	6.2 (0.6)	1019.2	11.9 (1.15)	1000.5	18.5 (1.82)	992.5	8.6 (0.86)	44.8 (4.3)	1.11 (±0.513)

4.3.3 Drivers of deforestation

Based on the AIC value, a logistic model including distance to forest edge and both the anthropogenic and topographic factors as predictor variables best explained the observed pattern of deforestation (table 4.4). This model received high support ($\Delta AIC = 7.96$ w_i=0.98) and had a good fit (Hosmer-Lemeshow test: χ^2= 12.839, p = 0.117) and high accuracy (ROC = 0.93 ±0.017; table 4.4). Deforestation was closely related to anthropogenic pressure and topographic constraints (table 4.5) . Areas subjected to a high level of anthropogenic pressure, corresponding to forest closer to settlements and roads, and low topographic constraints relating to forest occurring at lower elevations and on flatter land being more likely to be cleared (table 4.5). Deforestation risk was most strongly related to the distance to forest edge, emphasizing the importance of forest access. The final regression model correctly predicted 86.0% of the original observations of deforestation and was not affected by spatial autocorrelation (Moran's I =-0.002, P = 0.271).

The spatially explicit forest risk model (figure 4.3b), which was based on the results of the final regression model (table 4.5), was found to accurately predict deforestation that occurred between 2005 and 2009 with of kappa = 0.72.

Logistic Model	-2log(L)	K	ΔAIC	w_i	HL-test	Sig.	ROC±SE
Dist. Forest edge + Anthropogenic + Topographic	124.5	4	0	0.9816	12.839	0.117	0.933±0.017
Dist. Forest edge + Anthropogenic	134.46	3	7.96	0.0184	15.46	0.051	0.923±0.018
Dist. Forest edge	190.21	2	61.71	0	5.19	0.736	0.896±0.024

Table 4.4 Overview of logistic regression models ranked according to their ΔAIC, Goodness of fit was assessed by means of a Hosmer-Lemeshow test (HL-test). Model accuracy is given by the ROC value ±SE

Best logistic Model	B	S.E.	Wald	df	Sig.	Exp(B)
Distance forest edge	-3.24	0.67	23.76	1	<0.001	0.04
Anthropogenic	-1.67	0.31	29.25	1	<0.001	0.19
Topographic	-0.8	0.27	8.73	1	<0.001	0.45
Constant	-0.65	0.31	4.4	1	0.040	0.52

Table 4.5. Parameter estimates and significance under the best performing logistic regression model describing the relationships between landscape variables and deforestation patterns across the northern forest of Aceh.

4.4 Discussion

4.4.1 Deforestation model performance

The current estimates of forest cover and deforestation for the northern forest of Aceh comprise the first step in realizing a framework for the implementation of REDD in Aceh. The estimated forest cover and hence deforestation rate encompass a guideline to assess the total forest estate available and loss for to be used for REDD purposes. Yet, small-scale forest disturbances (~100 m²) encountered on the ground cannot be distinguished using satellite data (~30x30m resolution). For that reason, these results reflect the total amount of forest cover and clearance rather than forest degradation as a result of selective logging. Lower specificity scores in relation to specialization scores (table 4.1) suggest that ground observations of non-forest land cover have more often been erroneously classified than was the case if forest was observed on the ground. This discrepancy between the land cover classes observed on the ground and the predicted forest cover maps is likely to be a result of the method used. Removing small-scale (e.g. 1ha) anomalies from the predicted forest cover maps increased accuracy of the forest cover maps at the cost of introducing a small bias towards forested land cover. However, one can argue whether small-scale alterations of forest canopy integrity

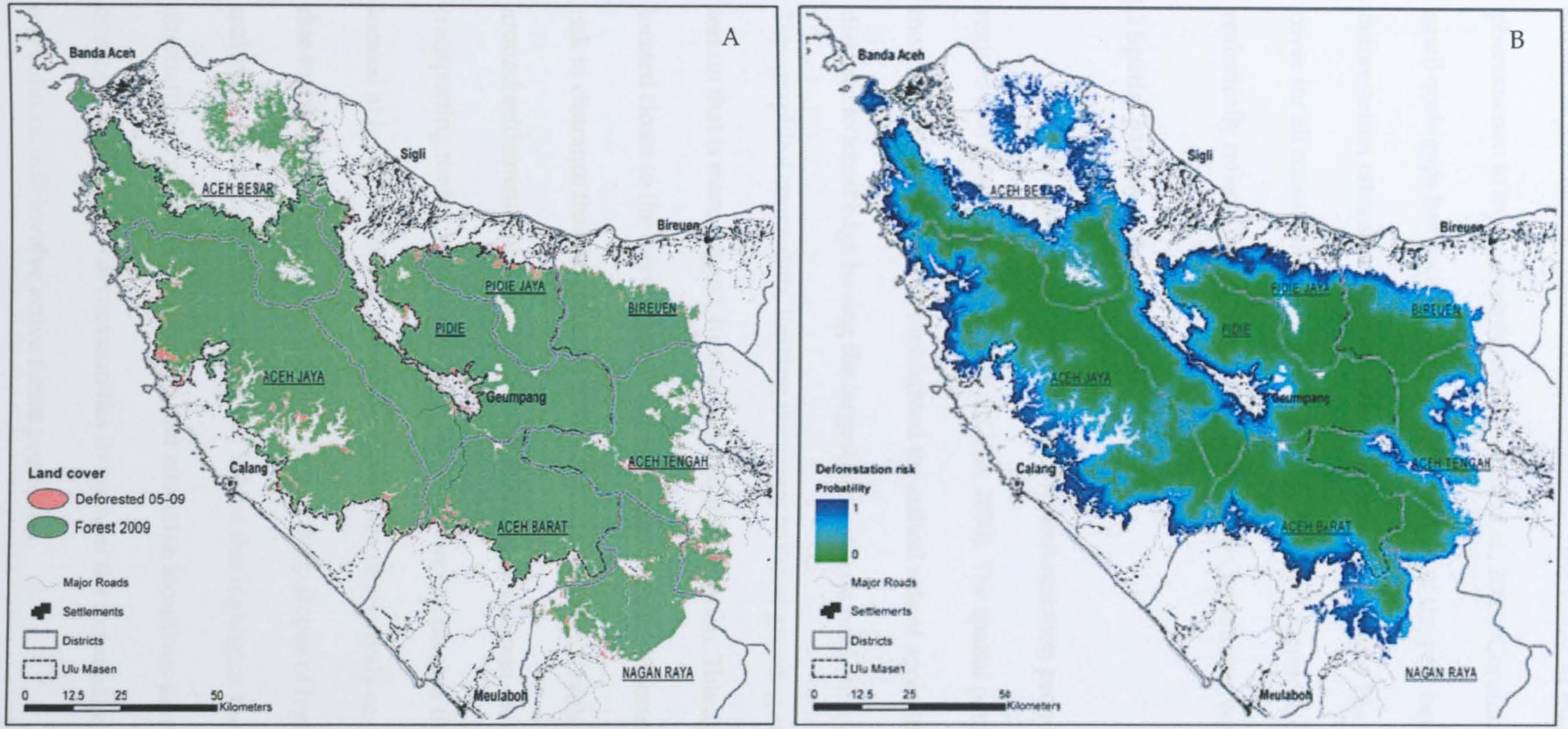

Figure 4.3(A) Deforestation in the northern forest of Aceh between 2005 and 2009. (B) Estimated deforestation risk across the northern forest of Aceh.

should *de facto* be interpreted as true non-forest land cover, as these are a naturally occurring phenomenon in tropical rainforests (Bischoff *et al.*, 2005). Considering that the omission of small-scale gaps has been introduced independent of the year being estimated, the deforestation rates are consistent between consecutive years. The overall predictions of forest cover for all consecutive years proved highly accurate, therefore they are believed to realistically reflect spatial patterns of deforestation within the Northern Forest of Aceh.

4.4.2 Spatial pattern of deforestation

Ultimate and proximate causes driving deforestation processes differ between various regions and spatial scales (Lambin & Geist, 2003). The spatial patterns of deforestation across the forest of northern Aceh highlighted the critical role of accessibility, with the importance of distance to forest edge having the largest influence on predicting deforestation (table 4.2). Topographical constraints, limiting the access to forest growing at higher elevations or in terrain that is more rugged, further reduced deforestation. This also explained why forests located closer to the forest edges and to settlements than hill forest, tended to be at a greater risk to clearance than hill forest (figure 4.3b). Deforestation levels were generally higher around settlements, presumably because travel time and cost are considerably lower when transporting timber across shorter distances. However, most of these settlements are also located at lower elevations adding the advantage of relatively easy access to lowland forests due to the lack of topographic barriers such as steep slopes of high elevation gradients, making it most susceptible to clearance. Whilst this highlights the importance of providing alternative livelihood opportunities and attractive incentives to reduce illegal logging and overexploitation by local communities living near the forest edge (Linkie *et al.*, 2004), part of any solution will involve active forest protection.

4.4.3 Historic deforestation trends

The government financed and unprompted transmigrations from Java to the northern parts of Sumatra in the early 1990s led to massive amounts of forest being converted to small-scale farmland (Linkie *et al.*, 2004). The deforestation pattern spread from the lowland coastal areas, where most transmigrants initially settled, inwards up the mountain slopes and higher forest plateaus. The 1990s featured a considerable economic growth in the South East Asia region resulting in a vast expansion of estate crops and the development of oil palm. In Aceh this has led to a reduction in forest cover of no less than 60.4% of total forest loss between 1984 and 1997 (Holmes, 2002).

Comparing deforestation rates across Indonesian regions revealed a marked variation between and within the islands of Sumatra and Borneo (table 4.6). For example, the central Sumatra region had the highest deforestation rates reaching up to 5.50%/yr. From the case studies found, the Ulu Masen and Leuser regions in Aceh had some of the lowest deforestation rates that were much lower the average rate recorded from the selected case studies (table 4.6). Over the period 2005-2009 a small increase in the annual deforestation rate was observed over the study area. This finding agrees with the observed deforestation rates

Location	Year	Deforestation rate (%/yr)	Source
Central Sumatra region	1990-1997	3.20 - 5.50	Achard et al. (2002)
Bengkulu province, Sumatra	1985-2002	1.41	Linkie et al. (in press)
Southern Sumatra region	1972-2006	0.64 -2.86	Gaveau et al. 2007
Riau province, Sumatra	1982-2007	1.68	Uryu et al. (2008)
Sumatra island	1990-2000	2.56	Gaveau et al. (2009)
Borneo island	2002-2005	1.7	Langer et al (2007)
Ulu Masen-Aceh, Sumatra	2005-2008	1.11	This study
West-central Sumatra region	1995-2001	0.96	Linkie et al. (2008)
Leuser region-Aceh, Sumatra	2006-2009	0.9	AFEP-LIF (2009)

Table 4.6 Overview of comparable deforestation rates recorded in Indonesia over the period 1990-2009

reported by Hansen et al (2008). Yet, the temporal resolution of one-year intervals used in this study did not allow to make inferences about the statistical accuracy of these estimates and hence cannot be used to draw conclusions about deforestation trends over time.

4.4.4 Implications for REDD

The results presented in this chapter provide the first accurate estimates of local deforestation rates in Aceh. These result therefore provide valuable information on the current state of the forest in Aceh which are critical to meet REDD guidelines outlined in the IPCC COP 13 action plan (IPCC, 2007). Even though the accuracy of optical remote sensors to reliably estimate forest carbon stocks has been questioned, the wide availability of application of these data still provide globally consistent and robust estimates of deforestation (Gibbs 2007). Consequently, combining deforestation data as presented in this chapter with biome average biomass inventory data, raw estimates of carbon stocks can be produced. Also, REDD schemes require estimates of background deforestation to determine net losses of carbon stocks, temporal trends in deforestation rates can be used to establish baseline deforestation rates.

As has been shown in this chapter deforestation rates across Ulu Masen (1.1%/year) currently remain amongst the lowest found in Sumatra and Borneo (2.56%/year: Gaveau, 2007, 1.7%/year Langer, 2009). Hence when incentives generated by reduced deforestation (eg REDD) would be determined based on local deforestation rates, Aceh would experience relatively lower benefits from prevented deforestation as compared to other provinces. Yet, since the renewed opportunities for corporate plantation development after the newly established peace agreement deforestation rates in Aceh are expected to increase in the near future (Gaveau 2009). REDD benefits based on predicted future deforestation rates could therefore provide a viable alternative to compensate deforestation agents. Also, with an estimated forest cover of 9920.0 km^2 this part of Aceh remains one of the most forested areas in the Indonesian archipelago. A REDD scheme in which a nation wide deforestation rate would be used to determine carbon profits are therefore be highly beneficial to the region.

5.1 Introduction

Across South-East Asia tropical deforestation of critical wildlife habitats continues at alarming rates (Santiapillai & Jackson, 1990; Achard *et al.*, 2007; Linkie *et al.*, 2008b; Hansen *et al.*, 2009a). Large-bodied mammals, depending on large areas of suitable habitat to meet their dietary demands, are considered to be particularly vulnerable to the effects of habitat transformation (Leimgruber *et al.*, 2003; Shannon *et al.*, 2009). At the same time, however, the replacement of primary forests, which tend to be low in terrestrial forage, by farmlands abundant in nutrients and energy-rich crops or secondary forest with higher levels of forage, may yield benefits to large-bodied herbivores. This apparent dichotomy is well illustrated by the Endangered Sumatran elephant (*Elephas maximus sumatranus*) which has been replaced from its natural habitat by forest conversion and is now considered a farmland pest species throughout its range (Choudhury, 1999; Zhang & Wang, 2003; Rood *et al.*, 2008).

Previous research on Sumatran elephants conducted by Kinnaird et al (2003) found an edge effect with elephants avoiding forest boundaries by up to 3km, indicating that elephant populations depend on undisturbed forested habitat. However, as the forested landscape is increasingly encroached upon by humans and most lowlands are now dominated by agriculture, the availability of suitable habitat has been reduced. The accelerate intrusion of elephant habitat by human settlers has recently resulted in an escalation of human-elephant conflict across the remaining elephants range (chapter 6/7). As elephant habitat on Sumatra gets increasingly fragmented, the remaining elephant groups are forced to reside in smaller isolated patches of forest occurring on the higher mountain slopes (Rood *et al.*, 2008)

At present, elephant research and conservation efforts have focused on estimating elephant densities by assessing populations using a variety of field survey and analytical techniques (Walsh *et al.*, 2001; Hedges *et al.*, 2005). Although such studies have been proven to be useful to monitor elephant population trends, they provide limited information on elephant habitat use and range. Knowledge on habitat selection processes and the consequences of habitat transformation on elephant distribution is essential to develop conservation strategies to improve their long-term survival prospects (Leimgruber *et al.*, 2003; Gaucherel *et al.*, 2010). In this study we investigate which environmental factors, both biotic and abiotic, constrain the current distribution of elephants in northern Sumatra. Secondly, we assess how elephant utilize their niche to find which areas represent core areas.

An emerging statistical technique that can be used for addressing these fundamental conservation and research needs is the Ecological Niche Factor Analysis(see section Ecological Niche Factor Analysis chapter 3; Hirzel *et al.*, 2002). The main advantage of the ENFA approach is that absence data, which is often unreliable due to problems associated with false absences (Hirzel *et al.*, 2002), is not required to conduct the analysis. Despite their size forest elephant are cryptic and highly mobile animals and recorded absences could result from the failure to detect actual presences or through spatial or temporal variation in habitat use. Disagreement between the geographical distribution of suitable sites and actual site occupancy could lead to low predictive accuracy when modelling species-environment relations (Cianfrani et al. 2010). Hence we determined habitat suitability based on presence only data rather than estimating occupancy using detection/non-detection data.

Since the ENFA algorithm calculates habitat suitability using raw presence data collected in the field, it diverges from the concept of the fundamental ecological niche (Hutchinson, 1957) but represents a approximation of the realized niche which can substantially deviate from the fundamental or core niche (Chefaoui & Lobo, 2008). Elephants are known to move between patches of high suitability (Sukumar, 1989), therefore a number of presence records can be accounted for by movements through areas of low suitability. These records do not describe core habitat characteristics necessary but merely represent an adaptation to local conditions. This effect is expected to be confounded if elephants live in a landscape containing highly fragmented habitat. To delineate areas of core elephant habitat models should aim to identify and account for presence records that were located within marginal habitats (c.f. Titeux et al., 2007).

In this chapter elephant habitat use is assessed by identifying those environmental factors, both biotic and abiotic that constrain the current distribution of elephants in northern Sumatra. Secondly, the elephants' ecological niche optimum is determined by analyzing the distribution of elephant presences in ecogeographical space. Finally a spatially explicit habitat model is built to establish core habitat areas and to assess the impact of forest encroachment of the prevalence of elephant habitat in Aceh.

5.2 Methods

5.2.1 Study area

Data were collected within the forests of northern Aceh (95°25'E-96°40'E and 05°30'N-04°08'N).The altitudinal range covered ranges from sea level to 2,697m asl, with ~50% of the forest below 800m asl. The geology of this area is predominantly sandstone and granite, but limestone formations are common along the west coast. The vegetation is dominated by dipterocarp forests interspersed with patches of pine forest, disturbed or secondary forests and *Imperata cylindrica* dominated grasslands. Most of the area has a protected status (i.e. hutan lindung; see section *Land use* chapter 2), but remnants of former commercial logging concessions can be found up to 20 km into the forest. Whilst all commercial logging has been stopped, illegal logging is rampant and patches of previously logged forest are rapidly converted into agriculture (Rood *et al.*, 2009).

Elephant populations within northern Aceh are believed to be fragmented into three distinct subpopulations separated by the Bukit Barisan Mountain Range and areas of human communities (Canney & Jepson, 2002). Even though no current estimates of the population size is available for the late 1980s Santiapillai & Jackson (1990) estimated the population to comprise 200-300 individuals.

5.2.2 Data collection

Initial pilot surveys were conducted from April 2006 to January 2007 by EJJR and AAG across the northern forests as to identify potential study areas, during which five teams were trained in elephant surveying. During February and March 2007, data on elephant distribution was collected over 12 different sites (figure 5.1). Data was collected using a stratified sampling design for which the study area was classified according to four elevation classes (500m intervals) and three land cover types (forest, non-forest, plantation). Within each site five random plots were selected from which transects were started. Subsequently, five parallel transects were walked each separated by 100 meters, resulting in a total of 25 transects per site and 300 transects over the whole study area. Elephant presence was recorded by means of five meter wide line transects that varied between 200 and 400 m in length. Presence was confirmed if fresh elephant dung (i.e. < 1 month old) was encountered and their geographic locations were recorded using GPS.

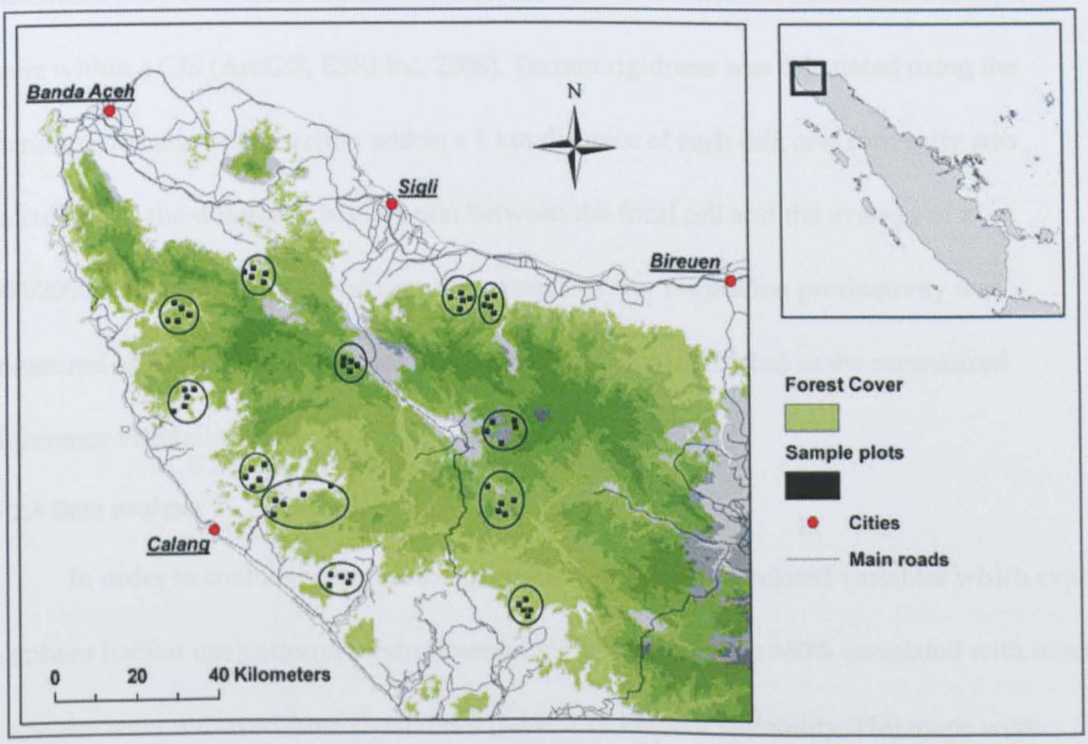

Figure 5.1. Overview of the study area showing 2006 primary forest cover and elevation (in 500 m intervals), with the 12 study sites encircling the plots. The insert shows the island of Sumatra with Aceh in the north

5.2.3 Data preparation

All presence data were transformed to a 90x90m raster format (WGS84, UTM-46n projection). A further 160 raster cells were randomly selected from the study area to describe the available background environment. Selecting background values randomly will lead to the occurrence of pseudo-absences at sites which possible represent suitable habitat (Chefaoui & Lobo, 2008). Consequently, potentially suitable habitat will be contrasted against the species optimum leading to a more conservative habitat suitability prediction.

To predict elephant habitat suitability and habitat distribution across the study area, a total of twelve habitat variables were used (table 5.1) based on their reported relevance to elephant ecology (Sukumar, 1989a, 1990; Pradhan & Wegge, 2007). Forest cover data was derived from three Landsat ETM+ satellite scenes from 2006 using a classification regression tree algorithm (Lawrence & Wright, 2001; Moisen & Frescino, 2002). Cross-validation of the resulting forest cover map proved the prediction to be accurate (94% accurate) with a kappa statistic of 0.87. A 90x90m digital elevation model was to calculate additional descriptors were within a GIS (ArcGIS, ESRI Inc. 2008). Terrain rigidness was calculated using the standard deviation of elevation within a 1 km distance of each cell; and convexity was calculated as the difference in elevation between the focal cell and the average of a 500/2000/5000 meter circular surrounding, respectively. Vegetation productivity was measured as the relative greenness of a pixel which was calculated as the normalized difference vegetation index (NDVI, c.f. Hansen et al., 2009a)

5.2.4 Data analysis

In order to enable the model to discriminate between correlated variables which explain elephant habitat use patterns, environmental variables that were >50% correlated with other variables were removed from the dataset (table 5.1). Habitat suitability (HS) maps were calculated using the geometric mean algorithm (Hirzel & Arlettaz, 2003). The ENFA algorithm was implemented using Biomapper 3.1 software (Hirzel *et al.*, 2002).

Variable	Description	Standardised	Included
Landscape			
Elevation	Elevation above sea level	Yes	Yes
Slope	Steepest slope in degrees	Yes	Yes
Terrain ruggedness 250m	Standard Deviation of elevation in a 250m circular surrounding	Yes	Yes
Terrain ruggedness 500m	Standard Deviation of elevation in a 500m circular surrounding	Yes	No: Correlated to Ruggedness 250
Terrain ruggedness 5000m	Standard Deviation of elevation in a 5km circular surrounding	Yes	No: Correlated to Ruggedness 250
Curvature 500m	Curvature: Relative elevation in relation to a 500m circular surrounding	Yes	Yes
Curvature 2000m	Curvature: Relative elevation in relation to a 2000m circular surrounding	Yes	No: Correlated to Curvature 500m
Curvature 5000m	Curvature: Relative elevation in relation to a 5km circular surrounding	Yes	Yes
Resource			
Forest Cover	Proportion forest cover in a 5km surrounding	No	Yes
Productivity	Normalized Difference Vegetation Index	Yes	Yes
Disturbance			
Road density 1000m	Road length in a 1km circular surrounding	Yes	Yes
Road distance	Euclidian distance to the nearest road	Yes	No: Correlated to Road density

Table 5.1. Description of each habitat variable used in the analysis. Variable standardization and whether a variable was excluded from the final analysis are indicated.

5.2.5 Model Validation

The habitat suitability model was validated using a continuous Boyce validation technique available within Biomapper software (see section Validation, chapter 3; Hirzel *et al.*, 2006; Pearce & Boyce, 2006). The validation statistic was calculated using a ratio between the number of observed presences and the number expected based on a random distribution (Hirzel et al., 2006). Good model performance is indicated by a high correlation between the habitat suitability score (HS) and the ratio of observed and expected values. Additionally, ROC-AUC scores were calculated to estimate model predictive power (Pearce & Boyce, 2006).

5.2.6 Core habitat areas

To identify core areas of elephant habitat a second habitat model was calculated excluding presence records which were found to deviate strongly from the average condition in which elephants were found. The method outlined by Titeux (2007) was used, who defined spatial outliers as those presence records located at the outermost 10% of the marginality axis, focusing however on the 90 % percentile interval of the marginality *and* specialization scores (figure 5.2). As such 88 (out of an initial 112) independent presence records were included to model core areas. Employing Boyce continuous validation (Hirzel *et al.*, 2006) plots, areas of high suitability were defined as those that were used disproportionally more than expected based on random use (i.e. HS>50%) and highly unsuitable areas were defined as those that the model indicated as being avoided (in this case HS<10%).

5.3 Results

5.3.1 Habitat analysis

During the initial training surveys, the presence of elephants was found at each of the 12 study sites, either through direct observations or indirectly by their sign (dung, vocalizations, tracks, etc.) confirming their presence throughout the northern forest. Within the survey plots, elephant presence was established on 35% (1.6± 0.09 95% C.I. transects per plot) of the transects surveyed. Trails were most abundant on flat areas, but narrow trails were present across a large altitudinal range from fresh water swamp forest at sea level to ridges up to 1600 m asl.

The ENFA analysis showed that elephants occupied areas deviate substantially from the average available habitat (Marginality; M=0.49). Elephants were found to be more frequent

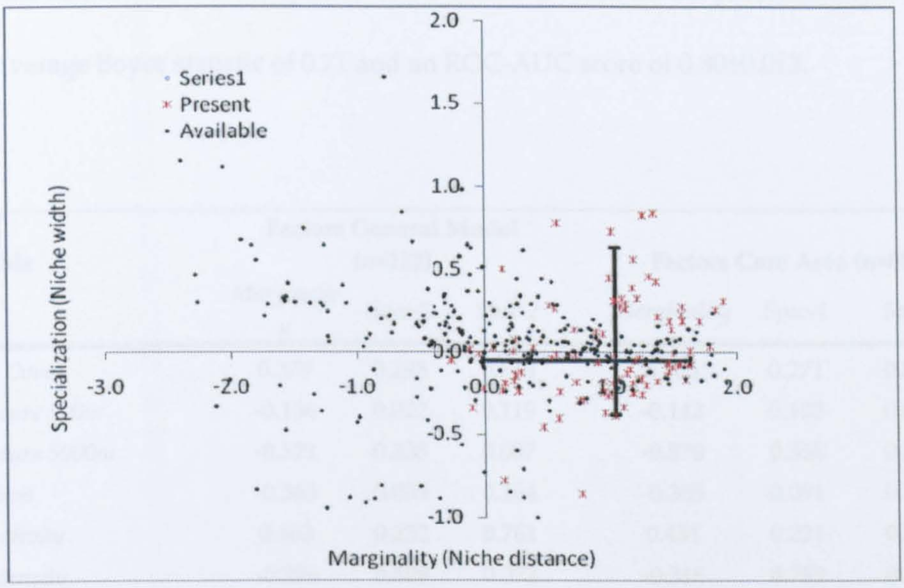

Figure 5.2. Distribution of elephant presence records (crosses, n=112) and a random sample of background records (representing available habitat; small black dots, n=200) along the marginality and specialization scores. Core elephant presence records (n=88) are those presences located within the 90% percentile intervals

in forested areas with relatively high amounts of productivity and in valleys. Moreover, the marginality score was weakly and negatively correlated to slopes (table 5.2), indicating that elephants did not show a strong preference for flat areas. Similarly elephant marginality was negatively related to the road density (table 5. 2), which implies an avoidance of areas with a dense road network.

Within the total range of potentially occupied habitats, elephants appeared to be restricted to a narrow range of specialized habitats (Total Specialization; S = 2.87), suggesting that elephants tend to occupy a relatively small ecological range as compared to habitat conditions available on a landscape scale. The first factor of the ENFA analysis (marginality), which maximizes the distance between the average conditions present and the average conditions at which elephant were found, accounted for 48% of the total variation described by the presence records. This indicates that the environmental factors describing the distance between the elephants' optimal niche and the available habitat are the same factors which describe the actual width of the species' niche width. Overall model predictability was good with an average Boyce statistic of 0.71 and an ROC-AUC score of 0.80±0.013.

Variable	Factors General Model (n=112)			Factors Core Area (n=88)		
	Marginality	*Spec-1*	*Spec-2*	*Marginality*	*Spec-1*	*Spe-2*
Forest Cover	0.379	0.298	**0.538**	0.366	0.271	**0.547**
Curvature 500m	-0.136	0.022	0.119	-0.142	0.103	0.050
Curvature 5000m	**-0.571**	0.338	0.007	**-0.570**	0.358	0.047
Elevation	-0.363	0.035	0.264	-0.365	0.091	0.185
Productivity	0.463	0.232	**0.761**	0.431	0.221	**0.762**
Road density	-0.296	**0.809**	0.072	-0.316	**0.799**	0.049
Slope	-0.177	0.012	0.037	-0.188	0.035	0.038
Terrain ruggedness	-0.218	0.295	0.203	-0.258	0.302	0.277
Variance explained	48%	24%	10%	40%	36%	10%
Total Variance Explained	82%			86%		

Table 5.2. Scores of the Ecological Niche Factor Analysis. Marginality indicates the distance between the average conditions at which elephants were found present and the average ecological conditions present in the study area. High values (>0.500) indicate higher use by elephants than expected on availability. Specialization factors indicate the ecological range present in the study area actually occupied by elephants.

5.3.2 Core areas

Plotting all elephant presence records against the calculated marginality and specialization showed that the distribution of elephant presence records is highly skewed towards the positive values of the marginality factor (figure 5.2), which corresponds to forested habitat types located within landscape depressions. After excluding outlier presence records, a second habitat model was calculated (table 5.2). The core area ENFA analysis factor scores are similar to the general habitat model, but showed a higher marginality score (M=0.501 core-model vs. M=0.493 general model), indicating that core presences deviate more from the average available conditions compared to the total range of ecological conditions used by elephants.

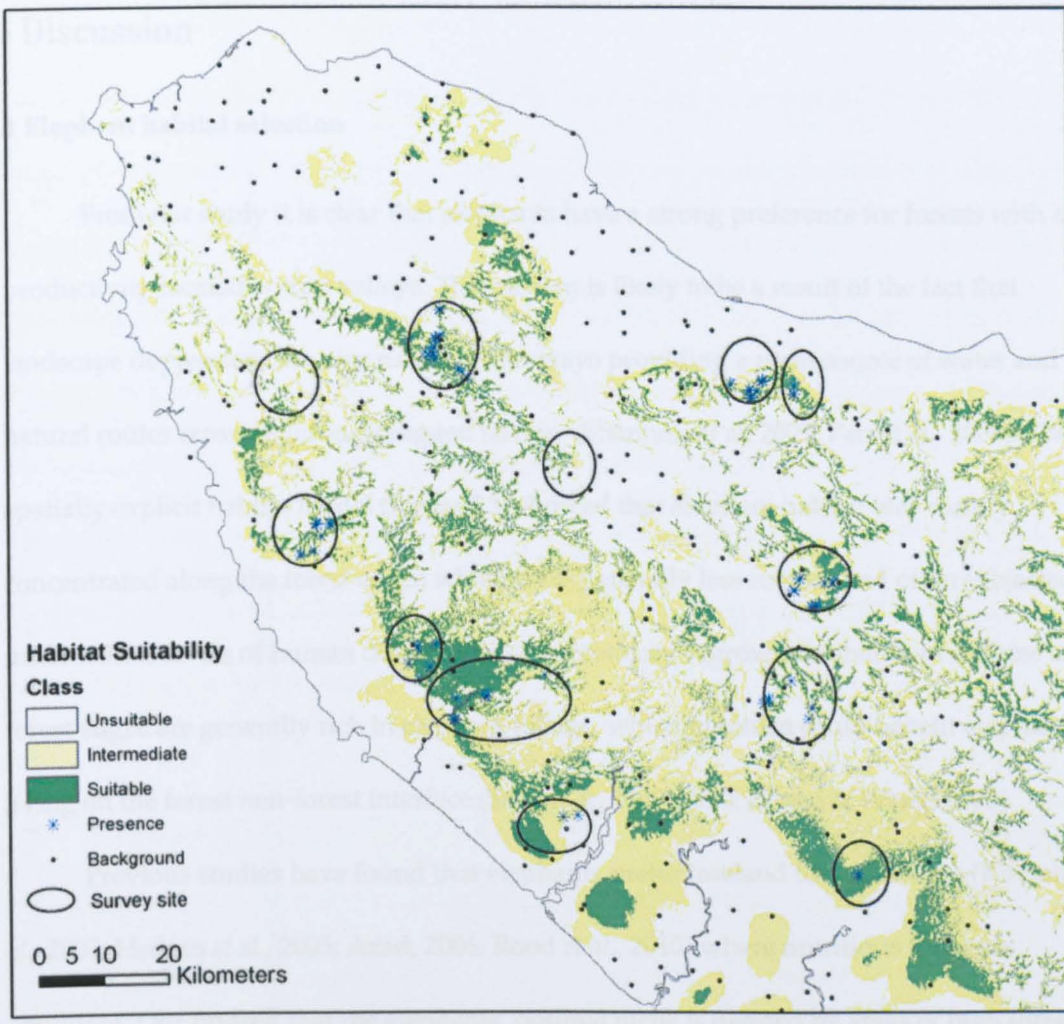

Figure 5.3. Elephant habitat suitability map based on ENFA analysis indicating areas of high and marginal suitability. Recorded elephant presences are indicated (asterisks), as well as the location of survey sites (ellipses).

The marginality factor and two specialization factors were used to produce a final habitat suitability map (figure 5.3). The Boyce continuous validation plot shows that the proposed core habitat model preformed well at distinguishing areas of highly suitable elephant habitat (Boyce = 0.75, AUC = 0.85±0.014). These results can therefore be used to identify areas of critical elephant habitat (figure 5.3).

Based on the spatial explicit model of elephant suitability across the northern forest there is no clear indication that populations are isolated. Even though highly suitable habitat appears to be fragmented, the connecting matrix of marginal habitat can provide possible migration routes to elephants moving between patches.

5.4 Discussion

5.4.1 Elephant habitat selection

From our study it is clear that elephants have a strong preference for forests with a high productivity located within valleys. This pattern is likely to be a result of the fact that landscape depressions are also natural waterways providing a main source of water and natural routes crossing through rugged terrain (Shannon et al, 2009; Pan et al., 2009). The spatially explicit habitat model (figure 5.3) showed that elephant habitat was mainly concentrated along the forest edges which were generally less rugged and often subjected to intermediate levels of human disturbance. As secondary regrowth is abundant in these areas, forest edges are generally rich in elephant foliage, which in return could benefit elephants living on the forest non-forest interface (Sukumar, 1989, 1990; Zhang & Wang, 2003).

Previous studies have found that elephants prefer lowland forest habitats (Kinnaird *et al.*, 2003; Hedges *et al.*, 2005; Azad, 2006; Rood *et al.*, 2010) where nutritious foliage is abundant. Our finding that the elephants' optimal niche is defined by areas of high forest cover as well as of high productivity (NDVI, table 5. 2) support this conclusion (figure 5.3, table 5.2). However, our finding that elephant occurrence is concentrated at forest edges does not agree with the results published by Kinnaird et at (2003) who conclude that elephants avoid forest edges. Yet, incongruent forest edge definitions and edge effects with diverging ecological conditions encountered in the field could have led to these observed differences. Moreover, the study presented here does not relate elephant abundance to habitat characteristics at small scales (cf Kinnaird et al, 2003), but rather reflects elephant habitat use at larger landscape scales .

Steep slopes have been mentioned to constrain elephant movements (Feng et al., 2008; Pan et al., 2009). (Sukumar, 1989). We found elephant to use areas up to 1600 m asl, and, concurrently to our study, fresh signs have been observed at 2200 m asl in the north-central part of our study area (M. Kamsi, pers. comm. Nov 2007). Our analysis found no marked relationship between slopes and the predicted distribution of elephants. Thus, while elephants might prefer flatter, lowland area, this does not imply that elephants are absent from mountainous areas with steep slopes that could limit their movements. The low correlation between slopes and the principal factors describing the elephants' niche suggest that elephants are well capable moving through mountainous areas. Terrain ruggedness, however, seems to constrain elephant niches to some extent, with lower frequencies of elephant occurrence in highly rugged terrain (eg elevation deviated approximately 50m or more between neighbouring cells) and elephant presences occurring over a relatively narrow range of relative ruggedness (table 5.2.)

The avoidance of areas with high road densities relating to high human population pressure implies elephant avoidance of human encroachment. Consequently, elephants are believed to move away from human dominated areas and move into more forested areas available within more mountainous areas. Including other parameters describing anthropogenic influences, such as human population density, in the analysis would enable a more thorough analysis of the effect of human presence on elephant habitat selection. Such data, however, are scantly available and often is outdated or unreliable, making comparisons hard to accomplish. Still, road density (or distance) denotes a well established parameter which has often been used as an indirect measure of human influence throughout conservation literature (Brooks et al., 1999; Linkie et al., 2004; Fuentes-Montemayor et al., 2009; Linkie et al., 2010).

5.4.2 Implications for conservation

The changing landscape across northern Aceh and the use of elephants of this area presents a conservation dilemma. Whilst elephants did indeed reside at forested edges rather than at the primary forest interior, it is unclear how deforestation will affect elephants in the long-term. In Aceh, elephant habitat use is limited by the total area of lowland forest, congruent to the work of Kinnaird *et al* (2003) in southern Sumatra. Further clearance of these areas could therefore lead to further deterioration of available habitat and may ultimately lead to the escalation of human-elephant conflict in the area and a decline of conservation moral amongst local stakeholders (Rood *et al.*, 2008; Uryu *et al.*, 2008). As land use planning for conservation landscapes within and outside established conservation areas is becoming a new standard in large mammal conservation practices (Nyhus & Tilson, 2004; Linkie *et al.*, 2006), the effects of land use configuration, elephant behaviour and human response are amongst the most important issues to account for when setting long-term elephant conservation priorities. This study has provided an initial step to identify and prioritize core areas for elephant conservation. Hence, local authorities have been provided with the foremost tools to incorporate species conservation priorities to be built on when future land use plans for the region are developed.

6.1 Introduction

Over the last decade, elephant conservation across the island of Sumatra has increasingly been coping with the occurrence of conflict between humans and wild ranging elephants, which is commonly known as Human-Elephant Conflict (Nyhus *et al.*, 2000; Rood, 2006; Linkie *et al.*, 2007). Human elephant conflict arises when human-elephant interaction result in negative effects on either human social, economic or cultural life, on elephant conservation or on the environment (Wemmer, 2008). In many parts of Asia, continuous forest conversion for the purpose of agricultural development, wood extraction and the opening of community gardens for subsistence has virtually eliminated all lowland elephant habitat (Leimgruber *et al.*, 2003). Likewise, in Sumatra, deforestation of lowland forests has progressed at an alarming rate (Hansen, 2009). Forested elephant habitat is now mainly to be found at higher elevations which has forced wild ranging elephants to move up the slopes of the Bukit Barisan mountain range where undisturbed habitat is still available(Hedges *et al.*, 2005; Rood *et al.*, 2008).

In many parts of Sumatra and Aceh likewise, the current landscape configuration, in which small patches of degraded forests are interspersed with small scale gardens and plantations, are believed to result in HEC (Nyhus *et al.*, 2000; Kinnaird *et al.*, 2003; Rood, 2006; Linkie *et al.*, 2007). In India, the replacement of elephant habitat with agricultural crops of high nutritious value to elephants, has led to a significant increase in crop raiding incidents by elephants (Sukumar, 1989b; Linkie *et al.*, 2004). In some regions of Aceh, the conversion of forest has led to a complete removal of natural forest occurring within historic elephant ranges and has left remnant elephant populations to dwell in a landscape dominated by agriculture (Rood, 2006).

The absence of distinct elephant ranges which are well separated from human populations, has led to an increased number of encounters between humans and elephants and is believed to be the primary cause of human-elephant conflict (Nyhus, 2008). Especially in mountainous areas where human settlements and gardens are often restricted to small stretches of relatively flat areas in mountain valleys or plateaus to grow crops, the chance of encounters with elephants is likely to increase (Azmi, *pers comm.*, 2006), as these are often used by elephants as natural pathways across the landscape (chapter 5). Consequently, the ongoing competition between human settlers and elephants for suitable living space is believed to make the occurrence of conflict inevitable.

Since elephants are wide ranging species the occurrence of human-elephant conflict is not limited to specific areas or villages, but rather occurs over large areas covered by elephant ranges. Alterations of elephant habitat at a local scale will therefore not necessarily result in an increase of human-elephant conflict. Large scale habitat encroachment, however, can ultimately cause a significant decrease in suitable habitat and the availability of suitable elephant forage equally, which will force elephants to utilize alternative resources. Yet the exploitation of such renewed resources by elephants often comes at the cost of increased contact with humans and hence an increase in conflict with human residents.

In order to mitigate the occurrence of human-elephant conflict, a landscape planning approach in which the both the requirements and interests of both humans and elephant are considered will be necessary (Nyhus, 2004). Prior knowledge on the effect of landscape configuration and land use on the instigation of human-elephant conflict have therefore been advocated to be integrated into wildlife management policies and regional spatial plans (Sitati *et al.*, 2003; Fernando *et al.*, 2005, Nyhus, 2006). Current land use plans in Indonesia, however, seldom account for species specific habitat or range requirements (Wich 2008, Gaveau, 2009). Regional differences in land use, local topography, elephant habitat availability do not allow the extrapolation of results from similar studies conducted in other Asian or African regions to the specific circumstances in found in Aceh. Moreover, the proximate effects of habitat alterations on the instigation of human-elephant conflict largely remain unknown. For that reason, a high necessity exists to identify those factors leading to the incidence of human-elephant conflict within the disturbed landscape matrix in Aceh.

A number of studies conducted across Asia and Africa have tried to identify global trends in the processes and patterns leading to crop raiding by elephants (Sukumar, 1990; Barnes, 1996; Hoare, 1999; Hoare, 2000; Williams *et al.*, 2001; Osborn & Parker, 2003; Sitati *et al.*, 2003; Zhang & Wang, 2003; Fernando *et al.*, 2005; Sitati *et al.*, 2005; Venkataraman *et al.*, 2005; Webber *et al.*, 2007). Many of these studies have mentioned alterations of elephant habitat integrity and habitat destruction as the ultimate causes leading to the occurrence of human-elephant conflict. Even though this view has now been widely accepted by many scientists and policy makers (Hoare, 1999; Williams *et al.*, 2001; Sitati *et al.*, 2003; Sitati *et al.*, 2005), little quantitative research has been undertaken to determine how alterations of natural habitat or forest configuration shape the spatial pattern of crop-raiding.

This study aims to investigate the patterns of crop raiding occurring over the province of Aceh, North Sumatra. Since the occurrence of crop raiding is believed to emerge from habitat degradation and consequently from a decrease in resource availability, the pattern of crop raiding incidents across Aceh is compared to the spatial configuration of forest stands and forest clearing patterns across the northern forests of Aceh (see also chapter 4). As habitat fragmentation is expected to cause an increase in the frequency of encounters between humans and elephants, the spatial pattern of crop raiding is hypothesized to be concentrated in areas where elephant habitat has been highly fragmented over the last two decades. Moreover, recent deforestation (i.e. between 1990-2005) has totally converted many of the historic elephant ranges (Heurn, 1929; Uryu *et al.*, 2008). Consequently, elephant groups now range within a landscape matrix which is highly dominated by humans. Crop raiding is therefore expected to be a result of the contemporary displacement of elephants from their historic ranges and will therefore be more frequent in areas that have been subjected to forest clearing in the during the last 10-20 years. However, if elephants are able to endure continuous habitat alteration by moving into alternative forested habitats, the occurrence of HEC will not exclusively occur in recently cleared areas but is more likely to be correlated to both recent deforestation as well as the total amount of forest cover available to elephants. Finally, as topographical factors can seriously limit elephant movements and will constrain elephants to re-colonize patches of suitable habitat, the effect of topography on the occurrence of HEC is also investigated.

6.2 Methods

6.2.1 Study area

Data were collected within the forests of northern Aceh, ranging from 95°25'E-96°40'E and 05°30'N-04°08'N (see figure 3.6 section 3.6.2). Most of the area has a protected status, but traces of prior logging concessions, which had been abandoned due to the armed conflict, can be found up to 20 km into the forest. Resultantly, between 1980 and 2000, 20% of the total forest cover was cleared, mainly for the timber trade (Rood *et al.*, 2009). Current logging activities are illegal but nevertheless rampant throughout the area.

6.2.2 Crop Raiding Data

Data on the occurrence of human-elephant conflict were collected by means of newspaper archives and reports made available by the Indonesian nature conservation agency. For the purposes of this study, only the crop raiding records compiled between 2000 and 2007. To prevent potential spatial biases of the data due to inaccurate reporting of the exact locality of an conflict event, only those records that specifically stated the location of an event down to the level of a settlement were used in the analysis. Consequently a total of 316 spatially explicit crop raiding events, were used in the analysis (see section 3.6.2). Even though the data was not derived form first hand field observations and hence does not guarantee the absence of spatial outliers, the landscape-scale analysis of these crop-raiding data outlined below are believed to be robust to small deviations of spatial localities.

6.2.3 Landscape descriptors

The spatial pattern of crop raiding was analyzed by means of six landscape descriptors. Four topographical descriptors were used to assess the relative importance of topography on the occurrence of elephants including: (1) elevation (2) landscape ruggedness (3) landscape curvature and (4) slope (see section 3.9). However, since the last two descriptors appeared to be highly correlated to the landscape ruggedness they were discarded from the analysis. Four land cover and land use descriptors used in the analysis included: (1) proportion of forest cover in 2007 within a five km radius of the focal cell, (2) proportion of secondary forest cover in 2007 within a five km radius of the focal cell (3) proportion of forest logged between 1990 and 2007 in a five km diameter from a focal cell (4) standardized Euclidian distance to roads. To enable comparisons between individual landscape descriptors, all landscape maps were standardized and converted to a 100m x 100m resolution before the subsequent analysis.

6.2.4 Data analysis

Ecological niche factor analysis was used to calculate the relative contributions of each of the six landscape descriptors to the occurrence of crop raiding by elephants (see also section 3.5.3 , Hirzel & Arlettaz, 2003). This allowed to predict where human-elephant conflict is likely to occur given the landscape configuration and topography of the area. The resulting model was validated by means of the AUC-ROC and a Monte Carlo randomization trial. A randomization estimation was obtained from 999 permutations of 316 crop raiding locations randomly distributed throughout the study area. At each permutation, Ecological Niche Factor Analysis was performed and the eigenvalue of the first factor extracted. The eigenvalue of the first ecological niche factor extracted using the crop raiding dataset was then compared to the observed average (± 95% confidence interval) of the simulated niche factor eigenvalue

(Calenge, 2007). The data was analysed using the R statistical software *adehabitat* software package (Calenge, 2007).

In order to assess how the observed pattern of crop raiding was affected by alterations of the available elephant habitat, the observed pattern of crop raiding incidences was compared to the distribution of elephants throughout the study area. Elephant distribution data collected across the study area (see section 3.5.2) was used to delineate the environmental conditions favoured by elephants, but where human-elephant conflict was absent (viz. control group). Next, this sample was compared to the environmental conditions at which crop raiding incidents had occurred (viz. treatment group). Additionally, a random sample of 500 background points was used to describe the average conditions present throughout the study area.

The same set of landscape descriptors previously used to model the occurrence of human-elephant conflict, was used to delineate the environment characteristics for each group. A stepwise discriminant analysis (Legendre, 1998) was then performed to investigate the relative influence of each of the descriptors to discriminate between the two groups using SpSS 16.0 software package.

Like the ENFA, discriminant analysis works in the space defined by the descriptors but it uses the distributions of both datasets to calculate an index that maximizes the interspecific variance while minimizing the intra-specific variance. Therefore, the discriminant factor is a linear combination of several predictor variables along which the two groups differ the most, i.e. it is correlated with the variables on which they are most differently distributed (see section 3.6.3). To assess whether the predictors were able to discriminate between localities of crop raiding and elephant occurrence, both datasets were plotted against their relative discriminant scores and a one-tailed t-test was applied to test for significant differences between population means.

6.3 Results

6.3.1 Crop raiding patterns

The ENFA analysis of the elephant crop raiding data showed that the six landscape predictors used in this analysis accounted for 94% of the variation present in the dataset (table 6.1). The overall model performance was good with an average ROC-AUC of 0.80 (± 0.024 SD). The results of the randomization test showed that the observed pattern of HEC was significantly different than expected based on a random distribution (σ= 3.77 p-value < 0.001, 999 permutations). The marginality score (distance from the average ecological conditions) showed that crop raiding occurs in distinct areas which deviate highly from the average conditions present throughout the study area (M= 0.98) and is most frequent in areas which have low forest cover, have recently been logged and are near to roads (table 6.1, figure 6.1). Of all crop raiding events, 27 % occurred within recently logged areas and 96% occurred in areas where logging had taken place within a vicinity of maximally five km.

Descriptor	Marg (59%)	Spec-1 (16%)	Spec-2 (11%)	Spec-3 (8%)
Forest cover	-0.473	-0.705	-0.221	-0.323
Elevation	-0.340	0.423	-0.723	0.195
Proportion deforested	0.444	-0.029	0.005	0.098
Distance to roads	-0.422	0.553	0.397	-0.330
Ruggedness	-0.354	-0.129	0.519	0.855
Poportion secondary forest	0.400	0.027	-0.004	0.085
Marginality(M) = 3.7				
Tolerance (1/specialization) = 0.455		Total Variance explained : 94%		

Table 6.1. Scores of the Ecological Niche Factor Analysis. Marginality indicates the distance between the average conditions at which crop raiding incidents occurred and the average ecological conditions present in the study area. High descriptor coefficients indicate a higher correlation with the occurrence of crop raiding than expected based on availability. Specialization factors indicate the ratio between the range of conditions present in the study area and the range of conditions where crop raiding was observed.

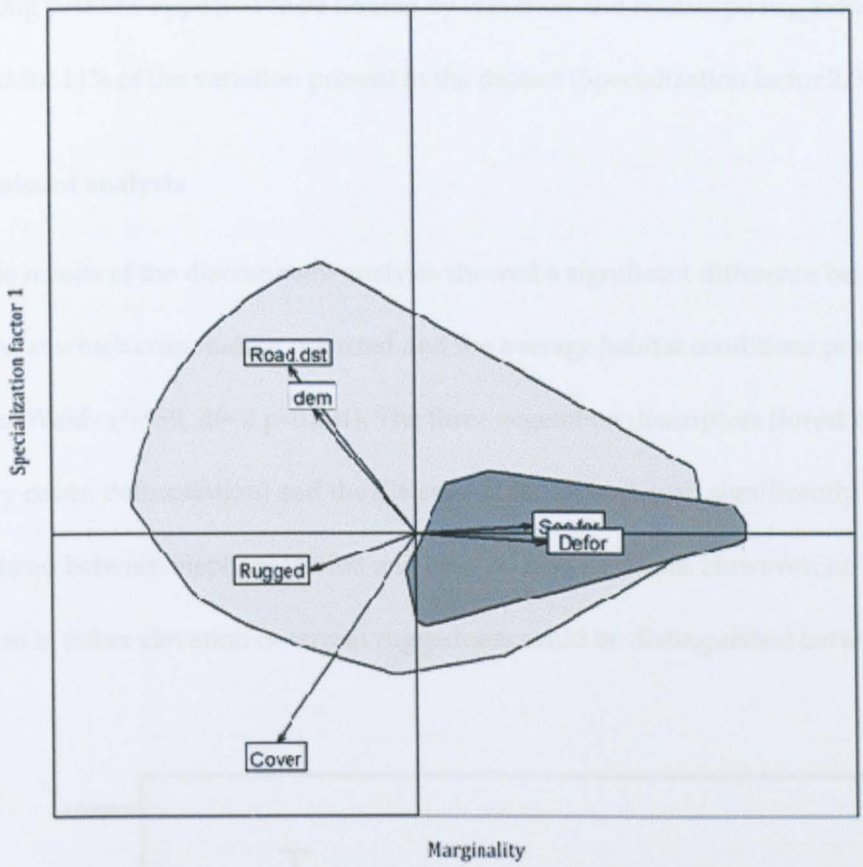

Figure 6.1 Biplot of the ENFA, formed by the marginality factor (X-axis) and the first specialization factor (Y-axis). The factor graph shows the occurrence of crop raiding incidents (dark polygon) across the study area (light polygon). The arrows are the projections of the environmental variables: (1) Road_dst: standardized Euclidian distance to the nearest road; (2) Cover: proportion of forest cover in 2007 within a five km radius of the focal cell, (3) Sec_for: proportion of secondary forest cover in 2007 within a five km radius of the focal cell (4) Defor: proportion of forest logged between 1990 and 2007 in a five km diameter from a focal cell (5) Rugged: Standard deviation of elevation within a five km radius of a focal cell. (6) Dem: standardized elevation asl.

Even more, 25% of the crop raiding incidents took place in areas which had no forest cover left within a five km radius of the crop raiding location.

The results of the ENFA analysis showed that the range of ecological conditions where crop raiding was observed was most restricted by the amount of forest cover within a five km distance and the distance to the nearest road. Hence, crop raiding was restricted to areas depicted by a low forest cover which were in the direct vicinity of roads (Specialization factor 1, table 6.1). Only after accounting for the spatial configuration of forest and deforestation, crop raiding patterns appeared to be limited by elevation and landscape ruggedness which accounted for 11% of the variation present in the dataset (Specialization factor 2/3, table 6.1).

6.3.2 Discriminant analysis

The results of the discriminant analysis showed a significant difference between the conditions at which crop raiding occurred and the average habitat conditions preferred by elephants (Wald- χ^2=969, df= 4 p<0.001). The three vegetation descriptors (forest cover, secondary cover, deforestation) and the distance to the nearest road significantly differentiated between elephant habitat and crop raiding locations. However, no significant differences in either elevation or terrain ruggedness could be distinguished between the two groups.

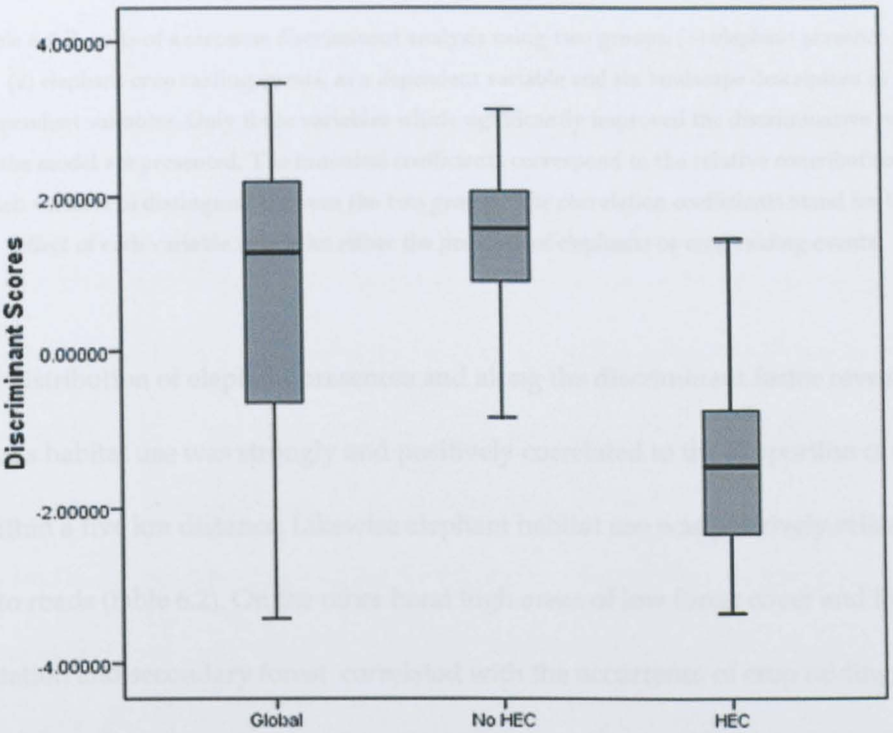

Figure 6.2. Boxplots of discriminant scores for (1) Whole study area (global) (2) Elephant presences without HEC and (3) Human-elephant conflict events (HEC). Boxes represent the 1st and 3rd interquartile ranges around the median. The discriminant function significantly differentiates between elephant habitat characteristics (No HEC) and landscape characteristics at sites subjected to human-elephant conflict (No HEC) (two-tailed t-test: t = 42.36, p<0.0001, df=866).

Even though the discriminant analysis did not completely differentiate between the occurrence of crop raiding incidents and the presence of elephants, the group means were found to be significantly different (Wilk's $\Lambda=0.325$, Two tailed t-test: $t=42.36$, $p<0.0001$, figure 6.2). This indicates that the occurrence of crop raiding and the distribution of elephants throughout the study area can be clearly separated based on the landscape descriptors used for this analysis.

Predictor	Canonical coef.	F	p	Correlation coef.
Secondary Forest	0.211	1432.037	0.000	-0.368
Forest Cover	1.064	813.212	0.000	0.893
Deforestation	0.263	581.438	0.000	-0.345
Road distance	0.356	447.095	0.000	-0.613

Table 6.2 Results of a stepwise discriminant analysis using two groups: (1) elephant presence and (2) elephant crop raiding events, as a dependent variable and six landscape descriptors as independent variables. Only those variables which significantly improved the discriminative power of the model are presented. The canonical coefficients correspond to the relative contribution of each variable to distinguish between the two groups. The correlation coefficients stand for the effect of each variable to predict either the presence of elephants or crop raiding events.

The distribution of elephant presences and along the discriminant factor revealed that the elephants habitat use was strongly and positively correlated to the proportion of forest present within a five km distance. Likewise elephant habitat use was positively related to distances to roads (table 6.2). On the other hand high areas of low forest cover and high levels of deforestation and secondary forest correlated with the occurrence of crop raiding incidents. The most apparent difference between elephant habitat and crop raiding sites therefore appears to be the amount of forest cover present in a five km circular surrounding. Moreover, crop raiding was significantly more frequent in areas which had been subjected to high levels of forest clearance between 1990-2005, but still had sufficient canopy cover due to secondary regrowth (table 6.2).

6.4 Discussion

The results of the analysis presented in this study show that deforestation and forest conversion do not always result in a total eradication of elephants from their natural ranges. In many cases, the spatial matrix of secondary forest interspersed with agricultural areas adjacent to primary forest stands provide sufficient habitat for the elephants to prevail (Nyhus *et al.*, 2000; Nyhus & Tilson, 2004). As forested areas are partially opened for agricultural purposes, elephants reside and utilize the subsequent regrowth as a resource of protein rich foliage (Sukumar, 1990). Previous research in India has shown that in an landscape with limited shelter, the remaining forested patches are intensively used and are likely to provide an essential place to shelter for the elephants during the day (Sukumar, 1990).

As the conversion of lowland habitat continues, elephants do not inevitably respond by moving to alternative still forested areas, but rather reside in smaller patches of less suitable habitat (Sukumar, 1989a; Nyhus & Tilson, 2004). Consequently, the currently observed distribution of elephants might not purely mirror the elephants' preference of the available resources, but is likely to reflect their historic ranges and movements. In those cases where elephant habitat is totally converted and the remaining groups permanently reside within a matrix of secondary forest and areas designated for agriculture or estate crop plantations, crop raiding behaviour by elephants was shown to become more likely.

The results of the analysis presented in this study support the idea that the incidence of crop raiding by elephants is concentrated in areas which recently have become deforested, have a low forest cover remaining and are in the direct vicinity of human populations (e.g. close to roads). A decrease in forest cover, however, does not unambiguously lead to an increase of crop raiding. Yet, elephants which inhabit degraded forests or areas with high levels of secondary forest regrowth, are highly likely to raid crops. Likewise, as the remaining forest patches are being cleared for agricultural expansion, the frequency of crop raiding by elephants is likely to increase . Discriminant analysis of our data showed that the elephant distribution patterns and crop raiding incident patterns clearly can be clearly distinguished based on the availability of forested habitats and opened forest. Also, crop raiding is more likely to occur in areas which still hold stands of secondary forest. The lower discriminative influence of the proportion of secondary forest to distinguish between the occurrence of elephants and crop raiding events emphasizes the fact that these habitats, to some extent, encompass natural elephant habitat.

6.4.1 Implications for conservation

`The finding that elephant habitat use is restricted by the availability of forests of high productivity, which was found to be concentrated along the forest edges (chapter 5), supports the idea that further forest encroachment and deterioration of critical elephant habitat will ultimately lead to a rise in human-elephant conflict. Resultantly, escalating conflict will decrease human tolerance towards elephants which again could lead to the killing and capturing of so called "problem elephants as has been observed in other areas in Sumatra (Hedges *et al.*, 2005; Uryu *et al.*, 2008) and Africa (De Boer & Baquete, 1998; Blake *et al.*, 2007).

As land use planning for conservation landscapes within and outside accomplished conservation areas is becoming a new standard in large mammal conservation practices, the effects of land use configuration, elephant behaviour and human response are the most important issues to account for when dealing with elephant conservation (O'Connell-Rodwell *et al.*, 2000; Leimgruber *et al.*, 2003; Venkataraman *et al.*, 2005). Since the majority of natural elephant ranges across Asia are situated outside the existing protected area network (Leimgruber *et al.*, 2003), appropriate conservation management and efficient land use will be of critical importance to minimize conflict and to guarantee the prevalence of local elephant populations. Land use zoning and forest rehabilitation should therefore be used to segregate areas of human interest and elephant habitat.

Chapter 7

ELEPHANT EXTINCTIONS IN SUMATRA

The effects of deforestation and habitat encroachment on elephant subpopulation survival.

"Zowel in Regeeringskringen als bij het groote publiek maakt men zich een verkeerde voorstelling van het aantal dezer dieren (olifanten, author note), dat zeer overschat wordt: in werkelijkheid is het slechts een poover overblijfsel van de groote kudden, die vroeger in Sumatra rondzwierven en waaronder een veelal noodelooze en ergerlijke slachting is aangericht(....)Het is dringend nodig dat de lacunes in beschermingsmaatregelen spoedig aangevuld worden"

W. Groeneveldt -1938-

"Both government as well as the general public have spurious views on the number of these animals (ie elephants, author's note) which has been highly overestimated: in reality it is only a poor relict of the large herds which once ranged across Sumatra and which have been unnecessary and aggravatingly slaughtered(....) There is a stringent need to fill this gap in the existing conservation policies."

7.1 Introduction

Ever since the start of the 20th century conservationists have recognised the deteriorating effect of elephant displacement resulting from competition between humans and wild ranging elephants for suitable living space (Pieters, 1932; Groeneveldt, 1938; Santiapillai & Jackson, 1990; Barnes, 1996; Hoare, 2000; Leimgruber *et al.*, 2003; Blake & Hedges, 2004; van Aarde *et al.*, 2006). Decreasing habitat availability is currently still believed to be the major driving force behind the continuously declining Asian population. Thus far, approximately 25,000-50,000 animals have been estimated to be living the wild (Santiapillai & Jackson, 1990; Leimgruber *et al.*, 2003). To counteract population decline, various studies have been conducted to identify and turn round factors driving the observed decline of elephants throughout its range (Choudhury, 1999; Johnsingh & Williams, 1999; Leimgruber *et al.*, 2003; Hedges *et al.*, 2005; Rood *et al.*, 2008). From this work it appears that the ongoing conversion and degradation of natural wildlife habitats, poaching and the killing of elephants responsible for conflicting encounters with humans over available resources pose the most serious threat to the future survival of elephant populations in the wild (Leimgruber *et al.*, 2003; Blake & Hedges, 2004; van Aarde *et al.*, 2006; Wemmer & Chirsten, 2008; Barua, 2010).

An Asia wide assessment of elephant ranges conducted by Leimgruber (2003) showed that 59 distinct populations of elephant can now be recognized across Asia. Yet, only six populations were found to be located within unfragmented habitats and no less than 37 are located within areas of high habitat fragmentation (Leimgruber *et al.*, 2003). In Sumatra, Indonesia, past assessments of the elephant populations have shown that the island still holds a substantial number of wild ranging elephants. On the basis of expert opinion, Blouch and colleagues (Blouch & Haryanto, 1984; Blouch & Simbolon, 1984) estimated approximately 2800-4800 elephants to occur within 44 distinct subpopulations scattered across he island (figure 7.1). Both Hedges et al. (2005) and Uryu et al (2008) provided updates for the provinces

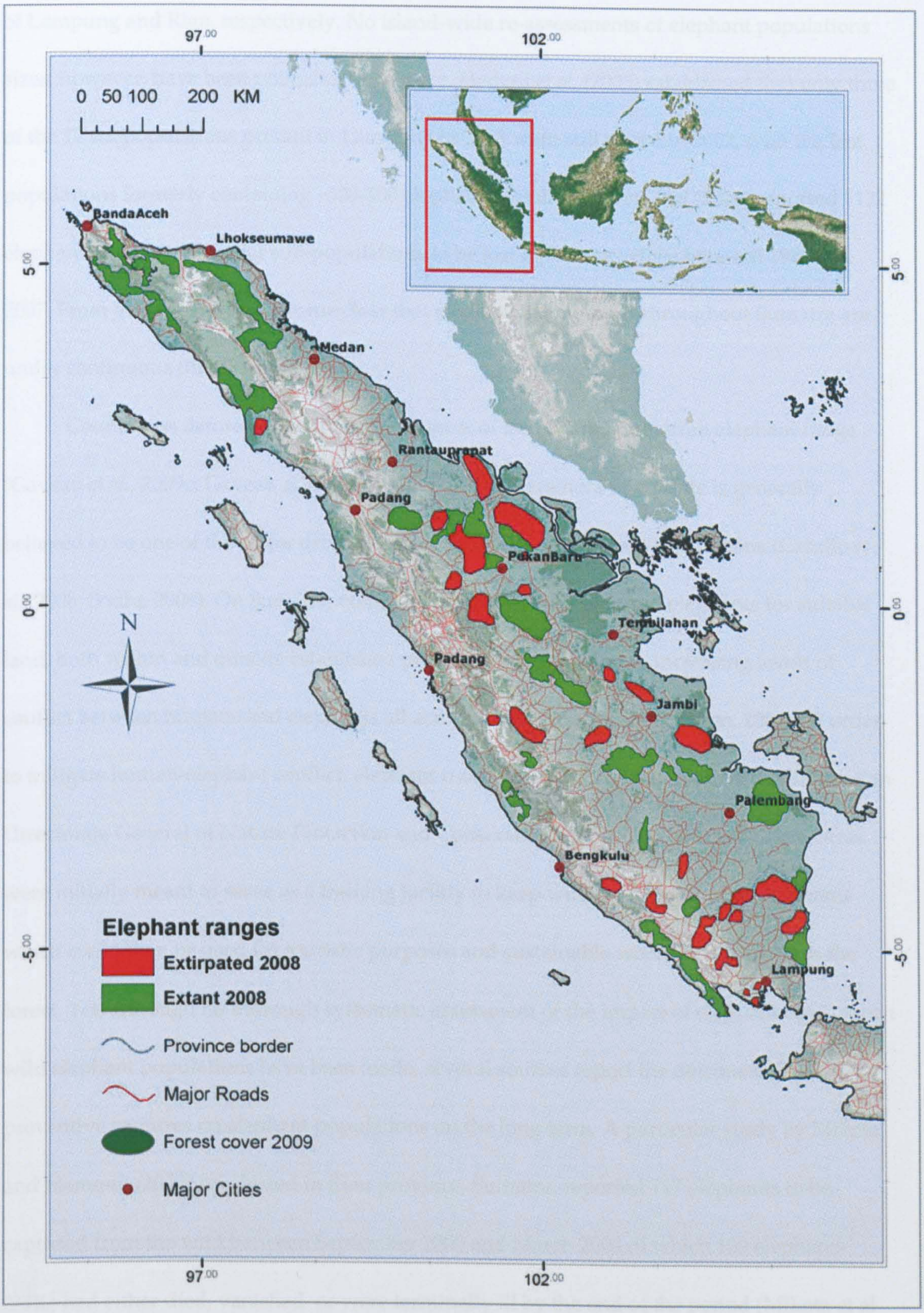

Figure 7.1 Elephant ranges recognized in 1990 which had gone extinct by 2008 (red) and the populations still extant by 2008 (green) .

of Lampung and Riau, respectively. No island-wide re-assessments of elephant populations sizes, however, have been published ever since. Hedges *et al.* (2005) established that only three of the 12 subpopulations present in Lampung in 1985 were still extant in 2002, with the lost populations formerly containing ~300-500 elephants. Similarly Uryu *et al* (2008) reported 1132 elephants from six different sub-populations to be lost in Riau province between 1985 and 2007. From this work it has become clear that elephant populations throughout Sumatra are under continuous threat of extinction.

Continuous deforestation throughout most of the existing Sumatran elephant range (Gaveau *et al.*, 2009c; Gaveau *et al.*, 2009a) and the displacement of wildlife is generally believed to be one of the major driving factors behind local elephant extirpations (Catullo *et al.*, 2008; Sodhi, 2008). On Sumatra, competition between humans and elephants for suitable land, both within and outside established protected areas, has led to increasing levels of conflict between humans and elephants all across the island (Nyhus & Tilson, 2004). In order to mitigate human-elephant conflict, elephant training centres were erected by the Indonesian Directorate General of Nature Protection and Conservation (PHPA). These training centres were initially meant to serve as a training facility to keep wild captured conflict elephants which could then be used for touristic purposes and sustainable wood extraction from the forest. Yet, although no thorough systematic assessment of the impact of elephant captures on wild elephant populations have been made, several sources report the detrimental effect of preventive captures on elephant populations on the long term. A particular study by Mikota and Hammot (2009) conducted in Riau province, Sumatra, reported 117 elephants to be captured from the wild between September 2000 and March 2003 of which 109 elephants (95%) had either died, vanished or were terminally ill by the end of the period (Mikota et al in: Wemmer & Chirsten eds., 2008). Moreover, the total number of elephants declined from 1700, representing 35% of the Sumatran population (Santiapillai & Jackson, 1990), in 1990, to a mere 350 elephants by the year 2006 (Wemmer & Chirsten, 2008). This leads us to the belief that current conflict mitigation practices have led to a considerable decline of elephants in Riau as well as other parts of Sumatra and are likely to have led to the extirpation of isolated elephant populations.

7.1.1 Deforestation in Sumatra

Indonesia has recently been found to have the second highest deforestation rate in the world after Brazil (Achmaliadi *et al.*, 2002; Achard *et al.*, 2007; Hansen *et al.*, 2009b). Deforestation rates were shown to be particularly high in Kalimantan and Sumatra where authorities have allowed vast amount of forest to be cleared to enable agricultural expansion. Over the last two decades, Sumatra has experienced considerable reduction of its forest cover with island wide deforestation rates ranging from 0.59% year^{-1} (Achard *et al.*, 2002) to 2.56% year^{-1} (Gaveau *et al.*, 2009c) between 1990-2000. Moreover, regional deforestation rates have far exceeded the average island wide deforestation rate and annual deforestation rates high as 11% have been reported during a single year in Riau Province between 2005 and 2006 (Uryu *et al.*, 2008).

Lowland forests, where approximately 95% of all deforestation between 1990-2000 occurred (Hansen *et al.*, 2009b), are prone to deforestation as land conversion and agricultural expansion for oil palm estate development (Uryu *et al.*, 2008; Hansen *et al.*, 2009b) advances. During the economic recession in 2000, and the resulting collapse of the Asian palm-oil market, deforestation rates in Sumatra declined significantly (Hansen *et al.*, 2009b). However the observed decrease in the clearing of lowland forest could simply reflect the lower conversion of intact lowland forest to oil palm due to limited availability, demonstrating the appalling state of Sumatra's lowland forest at the end of the 20th century.

This chapter aims to collate the available data on elephant populations in Sumatra to assess their current status and to identify the primary threats to their survival. In order to identify which populations are believed to still exist by the year 2005 and which have gone extinct during the period 1990 to 2005 the current occurrence of elephant populations on Sumatra will be compared to the historic elephant range locations. Moreover, the spatial pattern of observed elephant extirpations over the last two and a half decades will be compared to the pattern of deforestation and anthropogenic parameters to provide insight in the processes leading to local population extinctions. Finally a prediction of extinction risks will be made to assess which elephant populations are currently most likely to be prone to extinction and whether current protected areas do provide the necessary means to protect elephant populations in the future.

7.2 Methods

7.2.1 Forest cover and forest cover change

Forest cover change from 1990 - 2005 was estimated using LANDSAT 7 ETM + orthorectified satellite images that span 185km x 170km with a 28.5m x 28.5m resolution. Images were classified using a Classification Regression Tree (CRT) algorithm which recursively partitions the dataset in homogeneous subsets based on a set of rules (see section 3.2). The set of rules predicted by the CRT algorithm were then used in a GIS to map forest cover across Sumatra in 1990 and 2005. To remove small scale anomalies and to increase the accuracy of the final prediction, the estimated forest cover maps were resampled to a 250 x 250m resolution and areas smaller than 0.1 km^2 were merged into the neighbouring land cover class. The spatial pattern of deforestation was then determined by overlaying the two consecutive forest cover layers using ArgGIS 9.3

7.2.2 Elephant population distribution

Data on elephant range distributions was obtained from published reports and peer reviewed papers as described in chapter 3. As the established elephant ranges present in 1990 proved not to be 100% congruent with the extant elephant ranges identified in 2005, it was not possible to distinguish which part of the original elephant ranges had disappeared due to local extinctions or which were abandoned due to spatial shifts of their original ranges. To distinguish which populations found absent in 2005 were in fact absent due to local extinction and which were absent due to shifts in or contraction of their original ranges, only those distinct ranges recognized in 1990 that did not show a complete or partial overlap with any population range existing in 2005, were recorded as extirpated over the extent of the study period. To prevent biases due to the overrepresentation of areas where several extinctions had taken place within individual fragments of previously connected elephant ranges, the total area of past and current elephant distributions were sub-sampled at a 20x20km grid, corresponding to twice the minimum elephant range found in 1990. Subsequently a random sample of 100 grid cells was taken and for each grid cell elephant extinction (1) or survival (0) was recorded based on a >50% overlap with each respective elephant range.

7.2.3 Elephant population status

An estimate of the impact of recent population extirpations on the total number of elephants living in Sumatra was obtained by contrasting elephant population sizes and densities reported in the past to the currently reported range sizes and population numbers. To do so, the minimum and maximum population sizes reported by Blouch in 1984 were used to calculate the average population sizes and densities for each province (Blouch & Haryanto, 1984; Blouch & Simbolon, 1984). Next, estimates of elephant population sizes present in 2005 were determined from the a number of published reports (Hedges *et al.*, 2005; Rood, 2006; Uryu *et al.*, 2008). In case no population data was available for a specific elephant range, maximum elephant densities observed in 1990 were used to calculate the maximum number of elephants that are expected to be able to survive within the remaining habitat patch.

7.2.4 Landscape variables

Numerous studies stress the effect of human disturbance and habitat degradation on species perseverance as a cause of local species extinctions (Brooks *et al.*, 1999; Cardillo *et al.*, 2006; Meijaard & Sheil, 2008; Sodhi, 2008; Gaveau *et al.*, 2009b; Sodhi *et al.*, 2010). To investigate the relative influence of human disturbance on elephant population survival, two parameters were used in the analysis. Area accessibility was measured as the distance to the nearest road while accounting for topographical relief and slopes and was based on a digital road map derived from the Indonesian spatial planning agency Bakosutanal. Likewise, a digital map of Indonesian cities and villages was obtained from the National Geo-Spatial Intelligence agency (see section 3.9.3) from which the distance to the nearest settlement was calculated.

To assess the effect of habitat degradation on elephant extinction within a gird cell, two covariates were used in the analysis. First, the proportion of forest cover present in 1990 and 2005, occurring in a five km circular surrounding of a focal cell was calculated. Secondly, the proportion of deforestation in a five km radius of a focal cell was calculated. Finally, to assess the possible detrimental effect of human-elephant conflict on elephant perseverance within a grid cell, the occurrence of conflicting events between humans and elephants during the period 1985 within each distinct sub-population was recorded based on report available from published literature (Santiapillai & Jackson, 1990; Nyhus *et al.*, 2000; Nyhus & Tilson, 2004; Rood, 2006; Rood *et al.*, 2008; Uryu *et al.*, 2008). All calculations were completed using the ESRI ArcGIS 9.3 software package.

7.2.5 Logistic model

To compare the effect of environmental and anthropogenic factors on elephant population extinction a logistic regression model was build using the landscape predictor variables described above. Since the existence of spatial autocorrelation in the data could lead to a violation of the statistical assumption independence between samples (Lichstein *et al.*, 2002; Dormann *et al.*, 2007), a second set of autologistic models were built. Autologistic models explicitly account for spatial dependency between sample points by including an auto-covariate term to the model which was calculated as the weighted average of inverse distance between a focal point and every neighbouring point with a 200 km distance (see section 3.7.3 Dorfmann, 2007). Several candidate models were constructed using different combinations of predictor variables. The best candidate model was selected based on the Akaike Information Criterion (AIC) and Akaike weights (Burnham & Anderson, 2002)

7.2.6 Protected areas

The ability of protection areas to prevent the eradication of elephant ranges was evaluated by comparing the observed extinction rate within each of eight different protection tenure classes to the extinction expected if protection was absent. Yet, dissimilarities in tenure-dependent characteristics such as relative forest cover an deforestation rates between the treatment group (e.g. protected areas) and a control group (non-protected), can potentially introduce a significant bias to the results. In order to obtain an unbiased sample of elephant survival and extinction across both sample groups and to correct for those characteristics which are likely to influence elephant survival, Propensity Score Matching (PSM; Rosenbaum & Rubin, 1985; Austin, 2009) was applied to create an independent dataset as follows. First, those confounding variables, shown to significantly predict elephant extinction (see results section this chapter), were used to predict the probability of a given sample point to be located inside a protected area. This probability was then used to match observations made within a protected area (N=99), to an observation outside a protected area based on a > 95% similarity. This resulted in a pruned dataset (N=198) in which both treatment group and the control groups have statistically identical properties for those variables found to significantly affect elephant extinctions. The outcome of the matching operation was evaluated by comparing the differences in the confounding variables between the protected and the unprotected groups before and after matching by means of paired *t-tests* (Rosenbaum & Rubin, 1985; Austin, 2009).

7.3 Results

7.3.1 Forest cover and deforestation

Cross validation of the 2005 forest cover map showed a 94% agreement between the observed and the expected land cover classes (kappa = 0.87 N=500, see also section 3.2.2) and is therefore believed to accurately represent deforestation patterns across Sumatra . Between 1990 and 2005, 41% of the natural forest occurring on Sumatra disappeared as a result of logging operations and forest conversion, equalling an annual deforestation rate of 2.70%/year. By the year 2005, 30% of the Sumatran mainland surface was still covered by primary forest. These results agree with the Sumatra-wide deforestation rates of 2.60%/year, reported by Gaveau (2009c) and 2.76 %/year reported for both Sumatra and Borneo by Hansen (2009b). The provinces of Sumatera Selatan (4.73%/year), Riau (3.34%/year) and Jambi (3.24%/year) had deforestation rates exceeding the Sumatra-wide average. Moreover, deforestation rates in lowland areas below 500 m asl were threefold higher (3.68%/year) as compared to hilly areas with an elevation ranging between 500-1000 m asl (1.14%/year). Finally, deforestation rates observed outside protected areas exceeded the deforestation rate inside protected areas by a factor three (i.e. 3.43%/year and 1.05%/year respectively).

7.3.2 Elephant population status

Over the whole of Sumatra, twenty-three of the initial 44 distinct elephant populations recognized in 1990, were lost by the year 2005. Of the remaining 21 populations, one population was found to be fragmented into two smaller sub- populations (Uryu *et al.*, 2008) and one new population has been identified (Catullo *et al.*, 2008). Consequently, a total of 23 discrete elephant populations were believed to be still extant by the year 2005 (figure 7.1).

Accordingly, 32.500 km² (45%) of the 71.000 km² elephant range disappeared between 1990 and 2005.

In 1990, 25 (56%) of the 44 distinct elephant sub-populations recognized in 1990, were wholly or partially covered by one of the seventy-seven protected areas established in Sumatra to stop illegal deforestation and to protect wildlife. Yet, by the year 2005 eight of these populations had become extinct leaving 17 (65%) of the remaining 26 subpopulations occurring in areas protected by law. This drastic decrease in elephant populations and the associated loss of elephant habitat across Sumatra unmistakably had a devastating effect on the number of elephants surviving to the year 2005. Based on the observed loss of elephant populations and the formerly reported numbers of elephants allegedly enduring in small populations scattered across the island, a shocking population decrease of 1000 -1900 elephants is believed to leave the current population to a critical population size of only 2047 Sumatran elephants to remain in the wild (table 7.1).

	Population size			Populations		
	1990-min	**1990-max**	**2005**	**1990**	**2005**	**Extinct**
Aceh	600	850	550	4	5	1
Benkulu	100	150	140	1	1	0
Jambi	200	500	20	6	2	4
Lampung	500	1200	718	12	2	9
Riau	1200	1700	275	12	7	4
Sumatera Selatan	200	620	344	9	6	4
Grand Total	2800	5020	2047	44	23	23

Table 7.1 Overview of elephant population sizes and numbers reported to exist across Sumatra in 1990 and 2005. Minimum and maximum population sizes reported in 1990 were adapted from Blouch (1984a,b) and Satiapillai (1990). Elephant population numbers in 2005 were collated from published reports and peer reviewed articles (Hedges *et al.*, 2005; Rood, 2006; Gunaryadi, 2007; Maddox *et al.*, 2007; Uryu *et al.*, 2008).

7.3.3 Predictors of Elephant extinctions

Under the best performing logistic model (table 7.2), elephant populations were more likely to go extinct with a decreasing distance to settlements (β = -0.67, $p<0.016$) and a decreasing proportion of forests remnants remaining within the elephants ranges by the year 2005 (β = -1.22, $p<0.001$; table 7.2). Surprisingly, elephant populations were also found to be more likely to go extinct with decreasing deforestation (β = -0.82, $p=0.003$). Yet, populations which had gone extinct between 1990-2005 often had little or no forest cover in 1990 and generally experienced low deforestation rates due to low accessibility of the remaining forest.

A Morans I test for spatial clustering of the data revealed that a significant amount of spatial autocorrelation existed in the model residuals (Morans 'I = 3.25; p = 0.0011). Consequently, spatial dependency between sample points, possibly biasing the results thereby fallaciously accepting relations between the predictor variables and elephant extinctions as being true. Including an autocovariate term to the model significantly improved the prediction accuracy from 79% correct predictions and an ROC value of 0.83 under the general

Autologistic Model	-2LL	K	ΔAIC	w_i	HL-test	Sig.	ROC±SE
For 1990 + City dist + Autocov + Constant	76.66	4	0.00	0.84	7.591	0.474	0.903±0.0320
City dist + Autocov + Constant	82.14	3	3.47	0.15	8.172	0.417	0.8852±0.035
Autocov + Constant	89.24	2	8.57	0.01	10.184	0.178	0.868±0.0370
Logistic Model							
Defor + Forest 2005 + City dist + Constant	102.83	5	0.00	0.90	2.293	0.971	0.830±0.041
Defor + Forest 2005 + Constant	109.25	4	4.42	0.10	7.247	0.510	0.792±0.045
Defor + Constant	137.10	3	30.26	0.00	22.759	0.001	0.547±0.058

Table 7.2 Outputs of multiple logistic regression models, describing the probability of observing elephant extinctions within 20x20 km2 patches of elephant ranges as recognized in 1990. The first three models include an auto covariate term to account for spatial dependencies between observations. Models are raked according to their ΔAIC value with the best performing model showing having a zero ΔAIC. Model fit is assessed by means of a Hosmer-Lemeshow Goodness-of-Fit test (HL-Test). Model prediction accuracy is indicated by means of a receiver operating statistic (ROC).

logistic model to 83% correct predictions and 0.90 (ROC) under the autologistic model.

In contrast to the logistic model, including an auto covariate term to the model reduced the effect of forest cover in 2005 and relative amount deforestation which did not significantly affect the probability of elephant population extirpation in the autologistic model (table 7.2). Elephant extinctions, however, were still negatively influenced by the distance to the nearest city (β = -0.86, p=0.018; table 7.3) indicating a higher extinction risk near populated areas. Moreover, the probability of elephant extinction was negatively correlated to the proportion of forest cover in 1990 (β=-0.70, p=0.024), indicating that elephants were more prone to extinction when living in areas with a relatively low forest cover in 1990. The autocovariate term proved be highly significant (β = 6.82, p< 0.001; table 7.3), demonstrating a strong spatial dependence of elephant extirpations occurring within a 200 km radius (125600 km^2) of another elephant extirpation. Yet, empirical evidence has shown that elephants are able to prevail in areas as small as 240 km^2 (Hedges *et al.*, 2005).

Best Autologistic Model	β	S.E.	Wald	df	Sig.	Exp(B)
Forest 1990	-0.70	0.31	5.06	1	0.024	4.98E-01
City Dist	-0.86	0.36	5.63	1	0.018	4.24E-01
Autocov	6.82	1.80	14.31	1	0.000	9.20E+02
Constant	-3.81	1.08	12.45	1	0.000	2.20E-02
Best logistic Model						
Deforestation	-0.82	0.27	9.06	1	0.003	4.40E-01
Forest 2005	-1.22	0.29	17.76	1	0.000	2.97E-01
City Dist	-0.67	0.28	5.83	1	0.016	5.11E-01
Constant	-0.13	0.25	0.27	1	0.607	8.81E-01

Table 7.3 Parameter estimates of the best performing autologistic and logistic regression model. Variable coefficients (β) as well as a Wald test of parameter significance are shown. Predictor variable abbreviations are as follows: (1) Forest 1990: relative forest cover in 1990 within a 5km circular radius (2) Forest 2005 : relative forest cover in 2005 within a 5km circular radius (3) City dist: Relief corrected distance from the nearest settlement (4) Deforestation: Forest 1990: relative forest cover lost between 1990-2005 within a 5km circular radius (5) Autocov: Inverse distance to extirpations over a 200km distance.

The spatially autocorrelated pattern of elephant extinctions is therefore not very likely to reflect the disappearance of wide ranging populations (e.g. endogenous), but results from an all-encompassing influence (exogenous), not accounted for in the current model.

Overall model performance was good with an overall accuracy of 83 %, a goodness of fit chi-square 7.591 (HL-test $p=0.474$) and a receiver operating characteristic of 87% (table 7.2).

7.3.4 Protected Areas

Since the predictors of area tenure also significantly predict the probability of elephant extinction, the unmatched data could not be used to investigate the effect of area protection on elephant survival as it will be subjected to considerable selection bias (figure 7.2). Four confounding variables: forest cover in 1990, deforestation rate, distance to roads and an auto covariate term, were used to predict whether a given observation belonged to a protected area. The logistic model correctly predicted 89.9% of the observations indicating good performance and had a good fit (HL-test: $\chi_2 = 13.46$ $p=0.09$; $R^2 = 0.74$) and a receiver operating characteristic (ROC) of 95%. The results show that for every observation it was possible to effectively predict whether it was located within a protected area by means of higher forest cover in 1990 ($\beta=0.428$ $p=0.003$) and the autocovariate ($\beta=6.68$ $p<0.001$). Although protected areas were located at greater distances from roads and had lower deforestation rates as compared to areas which are not protected, these differences were not significant (distance to roads: $\beta=0.128$ $p=0.607$; deforestation rate: $\beta=-0.345$ $p=0.139$).

Propensity score matching (PSM) effectively eliminated differences in the confounding variables between the protected or treatment group and the not protected control group (figure 7.2). The discrepancy in relative deforestation rates which was found to significantly differ between areas within and outside protected areas (pre-PSM: $T=2.25$ $p= 0.025$, $df=1$) was absent in the matched dataset (post-PSM: $T=0.053$, $p= 0.958$, $df=1$). Likewise, the difference in forest cover in 1990 was effectively removed by the PSM (pre-PSM: $T=-2.38$ $p= 0.018$; post-

PSM: T=-0.107, p =0.915). Consequently any remaining difference in elephant extinction frequencies between protected and non-protected areas could be ascribed to the effect of area protection status.

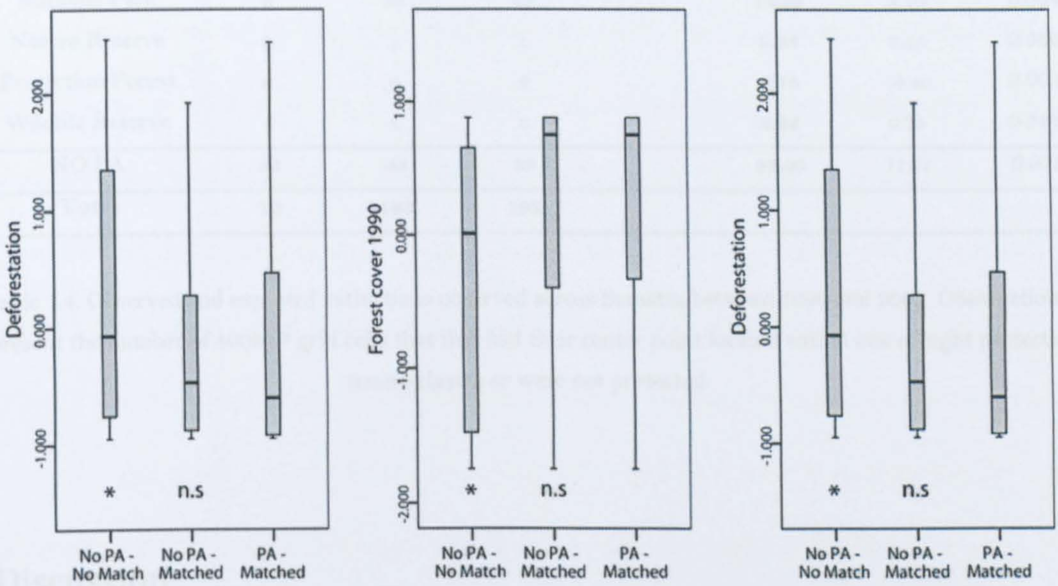

Figure 7.2. Boxplots of three parameters, significantly contributing to the prediction of elephant extinctions, before and after propensity score matching. *No-Pa, No-Match*: observations that were not located within a protected area and were excluded by the matching algorithm. No-PA Matched: observations that were not located within a protected area and were included by the matching algorithm. PA-Matched: observations that were located within a protected area and were included by the matching algorithm. Significant differences between group means and the mean value found in the PA-matched group are indicated by asterisks ($p<0.05$), n.s. indicates no significant difference between group means.

Based on the matched dataset, 198 observations of elephant occurrence recognized in 1990, sixty-one (31%) had disappeared by 2005. The number of observed extinctions in protected areas was significantly lower than the Sumatra-wide average ($\chi_2 = 11.31$, $p < 0.001$; table 7.4). However, the extinction rate observed in areas assigned as *"Protection forest"* was considerably higher as compared to the island wide extinction rate ($\chi_2 = 16.46$, $p = 0.001$; table 7.4) indicating that a land tenure of *"Protection Forest"* increased the probability of elephant extinction.

	Observed			Expected		
Area Status	**Extinct**	Survived	Total	**Extinct**	Chi-sq	P
Conservation Area	**0**	29	29	**10.25**	15.86	0.018
Game Reserve	**0**	4	4	**1.41**	2.19	0.379
Grand Forest Park	**0**	1	1	**0.35**	0.55	0.660
Hunting Park	**0**	5	5	**1.77**	2.73	0.326
National Park	**6**	36	42	**14.85**	8.16	0.004
Nature Reserve	**0**	1	1	**0.35**	0.55	0.660
Protection Forest	**9**	0	9	**3.18**	16.46	0.001
Wildlife Reserve	**4**	4	8	**2.83**	0.75	0.386
NO PA	***51***	*48*	*99*	**35.00**	*11.31*	*0.001*
Total	**70**	**128**	**198**			

Table 7.4. Observed and expected extinctions observed across Sumatra between 1990 and 2005. Observations represent the number of 400km² grid cells that that had their center point located within one of eight protection tenure classes or were not protected

7.4 Discussion

Over the last two decades elephant populations have considerably declined as a result of human induced habitat destruction. Elephant populations in Africa have been reasonably well monitored and approximately 472.000 elephants are believed to live on the continent (Stephenson & Ntiamoa-Baidu, 2010). Even though past assessments of elephant population dynamics in Africa frequently reported population declines due to poaching and habitat conversion (Prins *et al.*, 1994; Okello *et al.*, 2008), other studies have shown a stabilization on population number or even population growth within protected areas (Moss, 2001; Gough & Kerley, 2006; Foley & Faust, 2010). In Asia, a small number of studies have lately emphasized the detrimental effect of commercial development on habitat destruction and elephant population survival (Choudhury, 1999; Venkataraman *et al.*, 2002; Kinnaird *et al.*, 2003; Leimgruber *et al.*, 2003; Hedges *et al.*, 2005). This study has shown a decline of 45% in total elephant range size between 1990 and 2005. The reduction of elephant ranges around Sumatra led to the extinction of twenty three distinct elephant populations or 52% of the populations known to exist in 1990. Moreover, these results have shown that local extinctions are closely related to the amount of forest available to elephants and the average distance to roads indicating a strong anthropogenic influence.

Under the best performing logistic model, elephant extinctions were negatively related to the proportion of forest cover present 2005. Conversely, elephant extinction probabilities were positively related to the proportion of forest loss. This result poses an apparent controversy, as elephants appear to prefer forested habitat but have increased chances of survival with increasing deforestation. Yet, deforestation was more abundant in patches which are relatively accessible and still hold sufficient resources for exploitation. These patches, however have also been shown to form the main habitat for elephants (e.g. results this chapter and chapter five), Hence, the negative relation between elephant extinction probability and the proportion of area deforested is believed to reflect the fact that deforestation is more common in areas which still support elephants and not indicative for elephants preferring deforestation. It can therefore be hypothesized that elephant extinctions are principally driven by a reduction of the total amount of suitable habitat available and not by deforestation *per se*. This finding is supported by the autologistic model which showed elephant extinctions to be more likely in areas where forest cover had been limited since 1990. This suggests that the elephant extinctions occur as a reaction to habitat availability and show a delayed response to alterations of their habitat.

Even though these result provide some interesting insights on the occurrence of elephant extinction in Sumatra, they do not clarify how habitat destruction affects elephant extinctions. Even though it appears unlikely that forest destruction *per se* strongly affects elephants, deforestation has often been shown to be the foremost cause of alterations of ecosystem integrity (Meijaard *et al.*, 2005; DeFries *et al.*, 2007a). Compromising ecosystem properties such as resource availability, alterations of natural movements, fragmentation of populations and increased isolation could eventually lead to a decreased fecundity or increased mortality within elephant populations which can ultimately lead to local extinctions.

Our results have failed to demonstrate a negative effect of human wildlife conflict on the which has been referred to by other studies (Nyhus *et al.*, 2000; Zhang & Wang, 2003; Nyhus & Tilson, 2004; Fernando *et al.*, 2005; Rood, 2006; Rood *et al.*, 2008). Yet, since all of the 26 populations reported to exist in 2005 were subjected to different levels of human-elephant conflict, the current data did not contain information necessary to make inferences about the effect of human-elephant conflict on elephant extinctions. In order to assess the long term effect of human elephant interactions on elephant population dynamics, detailed information on the factors potentially causing human-elephant conflict should be incorporated in the analysis.

The analysis presented here did show that elephant extinction risk increased with decreasing distance to major cities. This anthropogenic effect is believed to reflect additional human pressure on elephant ranges through increased access to the elephant ranges. Several authors have previously noted that increased contact between elephants and human resident will eventually lead to an increase in human-elephant conflict and thereby could lead to elephant captures or even killing by local farmers (Nyhus *et al.*, 2000; Zhang & Wang, 2003; Nyhus & Tilson, 2004; Fernando *et al.*, 2005; Rood *et al.*, 2008). An increased accessibility to elephant ranges in combination with the observed decrease in suitable elephant habitat is therefore believed to have led to an increase in conflicting encounters between humans and elephants with an increase extinction risk as a result.

7.4.1 Spatial processes leading to extinctions

Even though spatial correlation was accounted for by using a sampling scheme in which observations were spaced at least 20km apart, model predictions were shown to be highly influenced by spatial patterns up to a 200 km radius of a sampling point (figure 7.3).

Empirical data has shown that elephant populations have been able to survive in ranges as small as 240 km² (Hedges *et al.*, 2005). Hence, it is highly implausible that the spatial dependency between the observed pattern of elephant extirpations is a mere result of elephant range dynamics. More likely, socio-economic or political processes which act on larger spatial scales but which were not included in this study have led to circumstances of increased pressure on the resilient elephant populations. Ambiguities in the land tenure system and corruption by province governments (Smith *et al.*, 2003) and illegal oil palm development (Fitzherbert *et al.*, 2008) are common and have led to the displacement of elephants from their historical ranges (Hedges *et al.*, 2006; Rood *et al.*, 2008) over whole provinces. Yet, as long as detailed information on land use policies and agro-economic processes within districts or even provinces are not available, no robust inferences on the true effect of economic developments on elephant conservation can be made.

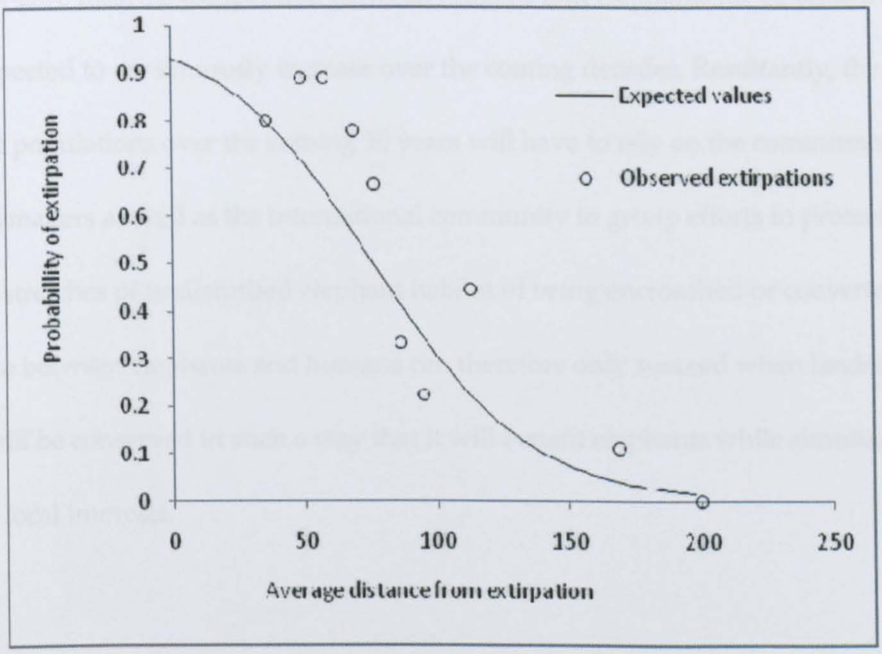

Figure 7.3 Partial effect of spatial autocorrelation on elephant extinction probability. The average distance between a sample point and all observed extinctions in a 200 km surrounding is plotted against the probability of observing extinction. Expected values are derived from the logistic model ($\beta = 6.82$, $p < 0.001$) keeping the other parameters constant at their respective median values. Observed extinction probabilities were calculated within 10 ranked distance bins (N=10).

7.4.2 Protected areas

The effectiveness of establishing protected areas to conserve biodiversity and their habitat has long been debated (Leimgruber *et al.*, 2003; Gaveau *et al.*, 2007; Catullo *et al.*, 2008; Gaveau *et al.*, 2009a; Linkie *et al.*, 2010; Sodhi *et al.*, 2010). Here we have shown that only a small part of both the historic ranges as well as extant elephant are covered by a protected area. After controlling for different levels of forest cover and access to elephant ranges, still, extinctions were found to be less common within areas with a protected status as compared to areas without protection. This suggests that protected area establishment does to some extent provide additional protection to elephants. Nevertheless, even within protected areas 19% of the elephant ranges were lost due to local extinctions stressing the fact that even if protection is enforced, elephant populations are under continuous threat.

As human population growth in Indonesia is amongst the largest in the world (2.6 mlj/year; UN-ESA, 2008) the competition between humans and elephants for suitable living space is expected to continuously increase over the coming decades. Resultantly, the survival of elephant populations over the coming 50 years will have to rely on the commitment of both local policymakers as well as the international community to group efforts to protect the last remaining stretches of undisturbed elephant habitat of being encroached or converted. Coexistence between elephants and humans can therefore only succeed when landscape integrity will be conserved in such a way that it will benefit elephants while simultaneously protecting local interests.

8.1 Introduction

Integrated Conservation and Development projects (ICDPs) are among the most widely applied paradigms in forest conservation in the tropics over the last 20 years (Adams *et al.*, 2004). These programs aim to limit deforestation though the identification of local threats to natural resources and participatory land u se planning (Wollenberg *et al.*, 2009). Payments for environmental services (PES) initiatives and Reduced Emissions from Deforestation and Degradation (REDD) schemes have also become increasingly popular approaches to forest conservation (Batterbury & Fernando, 2006) . These approaches, however, do not necessarily benefit traditional conservation strategies focussing on local area designations and land tenure (Gaveau *et al.*, 2009b; Blom *et al.*) and are unlikely to be effective if sub-national or national interventions are not considered (Blom *et al.*, 2010).

The performance of avoided deforestation schemes currently remains largely unknown as no projects have generated carbon revenue (Linkie *et al.*, 2010). At a national level, protected area networks and land use zoning have been shown to avoid significantly more tropical deforestation than unprotected areas (Andam *et al.*, 2008; Ewers & Rodrigues, 2008; Gaveau *et al.*, 2009c). Within these and other areas, law enforcement is likely to be the principal management strategy that drives most of the avoided forest loss (Gaveau *et al.*, 2009c). In the case of REDD, enforcement will be of significant importance to forest conservation projects to safeguard revenues from the international carbon markets (Venter *et al.*, 2009; Blom *et al.*, 2010). Hence, for this strategy to be effective, local or regional threats should be identified and dealt with at a national or sub-national level.

In the province of Aceh, Indonesia, the devastating tsunami in 2004 and recently established peace agreement in Aceh have led to an increased pressure on the area's natural resources. Many former farmlands that had previously been abandoned due to the armed conflict and since turned back to forest, are being reopened for cultivation. Moreover, Aceh faces an unprecedented demand for its natural resources, such as timber, and space for creating new farmlands. The Aceh Forest and Environment Project was started in 2006 to empower and support government and civil society partners to safeguard the forest and their vital ecosystem services in the area. In support of Aceh-wide environmental goals, the Government of Aceh has founded several initiatives that highlight the political will and commitment to protect Aceh's forests. These efforts are even more noteworthy because they are occurring at a time when many other Indonesian provinces, such as Riau, are rapidly converting their forest estates to oil palm (Uryu et al. 2008).

Anthropogenic factors are generally considered to be the driving forces behind tropical deforestation. Especially the expansion of agricultural frontiers, such as oil palm (Wilcove & Koh), and unsustainable logging practices, which are typically related to accessibility, such as forest proximity to roads and elevation are important factors explaining deforestation patterns in the tropics (Linkie *et al.*, 2004; Gaveau *et al.*, 2009c; Linkie *et al.*, 2010). Lowland forests, which support a high diversity of economically profitable hardwood tree species and are valuable for global carbon markets because of their high storage capacity, are highly threatened (Jepson *et al.*, 2001). At the same time the topographic location of these forests on lowland flats, directly adjoining human inhabited areas make them highly accessible. Hence, preventing deforestation in lowland areas is particularly relevant because strategic protection of the most accessible areas might not only provide direct benefits to these threatened forests, but also act as a barrier to preventing further forest loss (Andam *et al.*, 2008; Linkie *et al.*, 2010). Over the last decade increased attention has been given to the question how law enforcement strategies prevent deforestation in different areas of the world (Leader-Williams *et al.*, 1990; Pasya *et al.*, 2007; Andam *et al.*, 2008; Dennis *et al.*, 2008; Butler *et al.*, 2009; Gaveau *et al.*, 2009a; Linkie *et al.*, 2010) though little attention have been directed to the use of spatial modelling to evaluate conservation strategies.

Here, the effectiveness of conservation management intervention in and around the northern forest of Aceh, Indonesia is evaluated. The drivers of deforestation identified in chapter four of this thesis will be used to model deforestation patterns in the absence of active forest protection. Next, the impact of law enforcement effort that is allocated to protecting the: (1) existing protected areas, (2) the most vulnerable patches of forest, (3) prevent encroachment and (4) reducing deforestation pressure by applying buffer zones will be evaluated.

8.2 Methods

8.2.1 Study area

This study focuses on the northernmost forests of the province of Nanggroe Aceh Darussalam, Indonesia (section 2.2). The area spans 9,727 km² of forest, stretching along the Bukit Barisan mountain range situated between 4''20'3 N - 5''30'0 N and 95''20'0 E - 96''30'0 E (see section 2.2 for more details).

8.2.2 Deforestation modelling

To investigate deforestation risk, the occurrence of deforestation was analysed by means of logistic regression using topographic and anthropogenic parameters as predictors as described in chapter 4 (figure 8.1A). These results showed between 2005 and 2009, an average deforestation rate of 1.1%/yr was recorded in the Ulu Masen forest Block. The most rapidly cleared forest type was lowland (2.1%/yr), followed by sub-montane (0.6%/yr), hill (0.4%/yr) and then montane (0.3%/yr). Deforestation was strongly related to forest accessibility, with forest closer to settlements, to forest edge, at lower elevations and on flatter land being more likely to be cleared for farmland. Previous studies have emphasized the importance of area accessibility and human pressure to predict deforestation patterns (Kinnaird *et al.*, 2003; Gaveau *et al.*, 2009c; Linkie *et al.*, 2010) which therefore form the basis of the predictive model presented here.

Future deforestation patterns were predicted by means of an iterative model in which forest pixels were removed from the deforestation risk model during every consecutive iteration. Therefore, single pixels in the deforestation risk model were randomly selected and allowed to be removed according to their relative deforestation probability. The total amount of forest area predicted to be cleared, was determined according to the median deforestation rate observed between 2005-2009 (0.88% $year^{-1}$). This process was repeated for 20 consecutive iterations in which a single loop represented a period of five years of deforestation (i.e. baseline deforestation rate of 4.4%/year) representing a total period of 100 years of deforestation. Next, this forest loss was then used to update the forest cover and consequently the distance to forest edge covariate which, along with the other spatial covariates, formed a revised spatial dataset. The revised distance to edge layer, which moved further into the interior of the study area, had the effect of increasing the accessibility (and therefore risk value) of forest pixels close to the new edge boundary. Third, an updated deforestation model for the next year was constructed by applying the results of the logistic regression to the updated spatial dataset to then produce a forest risk model for the following year. This iterative process was repeated for each consecutive interval.

For all years modelled, a deforestation threshold was included within the modelling procedure. This threshold represents the net cost of deforestation and was based on the lowest predicted deforestation probability that was found to be cleared between 2005 and 2009 (i.e. p=0.75). This meant that forest pixels with a risk value equal to or lower than the threshold could not be cleared within the modelling procedure thereby reflecting a realistic situation on the ground. As deforestation rates would reduce over time, forest less suitable for clearance, e.g. at higher elevations, would not be cleared at the same rate as the more susceptible forest patches.

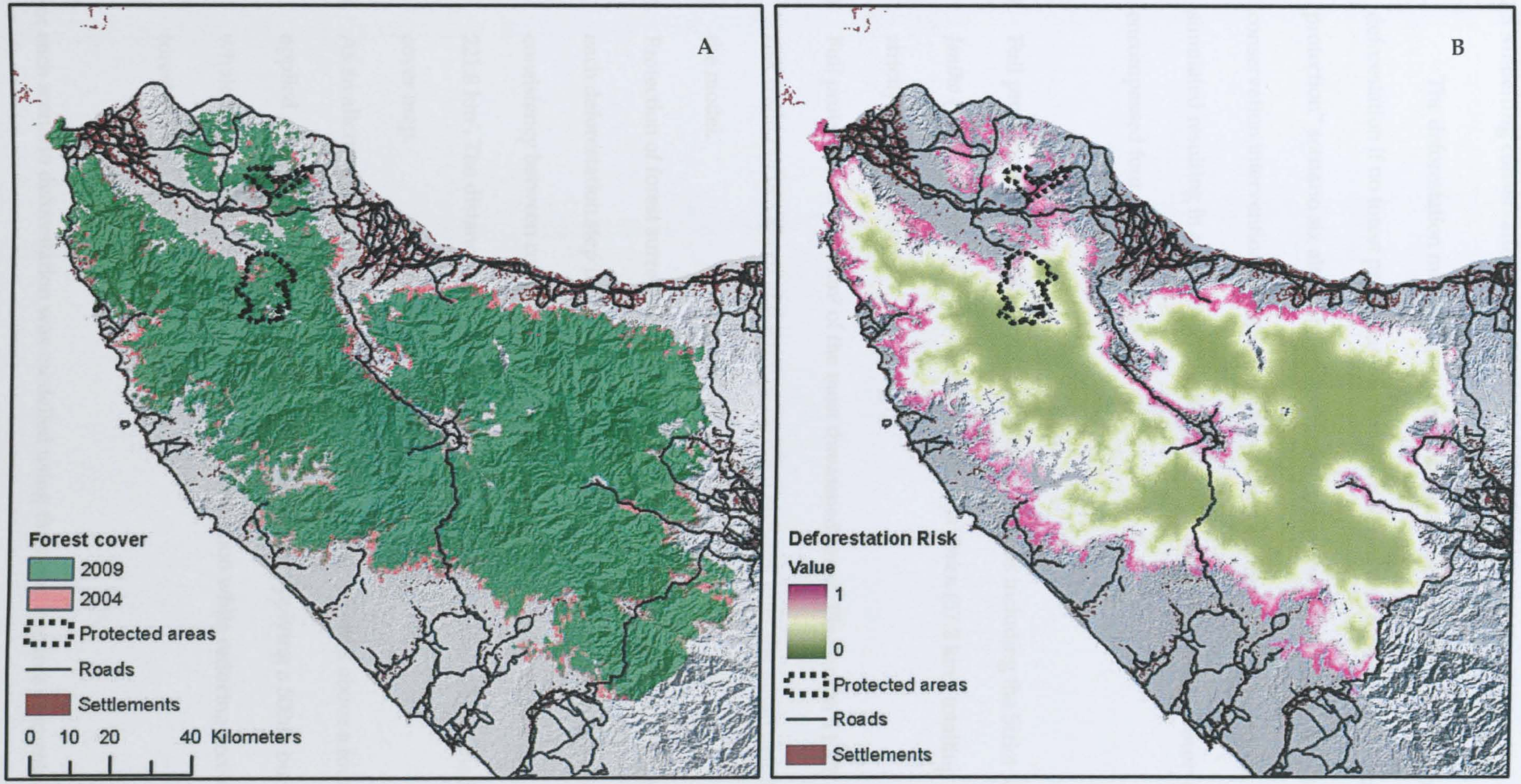

Fig. 8.1 Forest cover (A) and predicted deforestation risk in 2005 (B) in the northern forest of Aceh

8.2.3 Modelling conservation intervention scenarios

The deforestation modelling process was performed to determine the impact of deforestation if no forest protection measures are enforced in the area. In addition to this "No protection" scenario six alternative conservation scenarios representing four different conservation intervention strategies as well as combinations of these strategies (table 8.1) were simulated resulting in a total of nine different deforestation scenarios. Conservation scenarios encompassed four different strategies:

1. Full protection of currently established protected areas including the Strict Nature Reserve *Jantho* (164.6 km^2) and Grand Forest Park *Tjut Nya'Dhien* (57.2 km^2) totalling 221.8km^2 of strictly protected forest.
2. Full protection of 221.8 km^2 of the most threatened forest. Pixels with a high deforestation risk could not be cleared and were reclassified as forest after the initial deforestation step in the model.
3. Protection of forest surrounding newly established forest gaps diminishing access. After each deforestation step in the model, forest gaps < 10 km^2 were identified and filled. For consistency between conservation scenarios, the total area filled was set to a maximum of 221.8 km^2. The distance to forest edge was then calculated based on this modified forest cover map.
4. As an alternative to the active law enforcement strategies stated above a fourth scenario applied a passive protection strategy was modelled by applying a 500m buffer around the whole forest edge allowing limited resource extraction while reducing access to the adjacent forest areas.

For each scenario deforestation was modelled using the same iterative deforestation model.

The effectiveness of each conservation scenario was assessed by conducting a survival analysis using the average time to deforestation as the dependent variable. Survival analysis uses the time for an event to occur, in combination with appropriate covariates, to estimate the hazard- or failure rate (see section 3.8.2). A dataset was constructed by taking a random sample of 1000 pixels from the 2009 forest area. Next, for each pixel the number of model iterations until deforestation took place was recorded. A parametric regression model was fit to the survival data. This method has the advantage that it considers *right censored* data which occurs when a number of censored pixels did not experience an event of interest (*i.e.* deforestation) within the time span of the study (*i.e.* 100 years).

To investigate the effect of each conservation scenario on the average hazard rate, each scenario was included as a nominal covariate in the analysis. Since we were also interested in the change in the deforestation rate over time, a Weibull distribution was used as it allows the hazard to change as a function of time (Pinder *et al.*, 1978). To determine the change in hazard rate (i.e. the deforestation rate) over time a scale parameter (σ) is added to the model. If $\sigma > 1$, the deforestation rate decreases over time and vice versa (section 3.8.2). A dataset was constructed by taking a random sample of 1000 pixels from the 2009 forest area. Next, for each pixel the number of model iterations until deforestation took place was recorded. The final hazard rate functions for those scenarios that significantly reduced the average deforestation rate were calculated and plotted.

An assessment of the relative cost-effectiveness of each strategy was made by comparing the net gain in prevented forest loss relative to ranked costs associated to each strategy based on three nominal classes (low-intermediate-high). Since no factual field data on the costs of patrolling extended

Scenario	Protection measures				Cost	
	Protected area	Threatened Area	Gaps	Buffer	Area (km2)	Operational cost
No Protection	-	-	-	-	-	-
Protection PA	x	-	-	-	222	low
Protection A	-	x	-	-	222	Low
Protection B	-	-	x	-	222	Low
Protection C	-	-	-	x	1269*	Intermediate
Protection D	-	x	x	-	444	Low
Protection E	-	-	x	x	1391	Intermediate
Protection F	-	x	-	x	1391	Intermediate
Protection G	-	x	x	x	1713	High

* Area buffered in 2010

Table 8.1 Protection strategies modelled under different scenarios (A-G). The relative costs associated with the area actively managed under each strategy are also indicated.

areas are available, we assumed these costs to be unequivocally related to the actual area patrolled. Equally the net benefits of forest preservation were assumed to be directly related to the overall reduction in forest loss.

Even though this approach does not incorporate complex socio-economic or temporal discounted valuation of forest and other ecosystem services , it does allow to compare different enforcement strategies based on a single benefit (i.e. prevented forest loss). providing basic insights in the implications of forest management systems. Since the direct costs associated with different scenarios are not known, it follows naturally that the costs associated to land management are directly related to the area protected. The scenario in which forest edges were buffered by establishing zones of limited resource extraction were hypothesized to have low operational costs as no direct patrols will be necessary to implement this scenario.

8.3 Results

8.3.1 Conservation intervention strategies

The model built by the survival analysis using the different protection scenarios to predict the average time until deforestation proved to accurately fit the data (Likelihood ratio test: $\chi^2 = 345.76$, df =8, $p < 0.0001$). The estimated deforestation rate of 4.38%/year (Constant = 4.47, $p < 0.001$ table 8.2) was also very close to the deforestation rate of 4.4%/year used in the initial deforestation model. Moreover, the observed deforestation rate decreased over time ($\sigma = 1.43$), indicating that including a deforestation threshold effectively reduced deforestation rates over time.

The *No Prot* scenario, which modelled forest loss patterns in the absence of active protection, highlighted the critical risk posed the forest. Under the no protection scenario the total area that remained forested by 2110 was the smallest when compared to any other scenario considered (figure. 8.2). If full protection of the currently established protected areas would be accomplished (*Protection PA*), this would only decrease the total forest loss by 0.6%. Similarly, focusing protection on the most threatened forest patches (*Scenario A*) or the areas directly surrounding forest gaps *(Scenario B)* reduced forest loss with merely 1.4% and 2.1% respectively by the year 2110, leaving respectively 70.8% and 71.5% of the forest to remain. Under both conservation intervention scenarios the forest remaining consists of a single forest block of inaccessible forest, with patches of highly threatened lowland forest that were under strict protection scattered around the edges (figure 8.4). Yet, the majority of the other lowland forest had disappeared by 2110.

After limiting forest access by increasing the distance to the forest edge through the realization of a 500m buffer zone alongside the forest edge (*Scenario* C), a reduction of forest loss up to 7.6% was observed, thereby having the largest influence on limiting forest loss.

Combining different intervention strategies resulted in larger reductions of forest loss (Scenario's D/E/F; figure 8.2/table 8.2) over the no protection scenario. Applying a buffer zone around the forest, while at the same time providing full protection to the most threatened patches (*Scenario F*), resulted in a total reduction of forest loss of 7.9% leaving 77.3 of the forest cover remaining in 2110. Yet this scenario (i.e. *Scenario F*) did not significantly reduce deforestation as compared to applying a buffer only (deforestation: 7.6% forest cover remaining 77.0%; *Scenario* C).

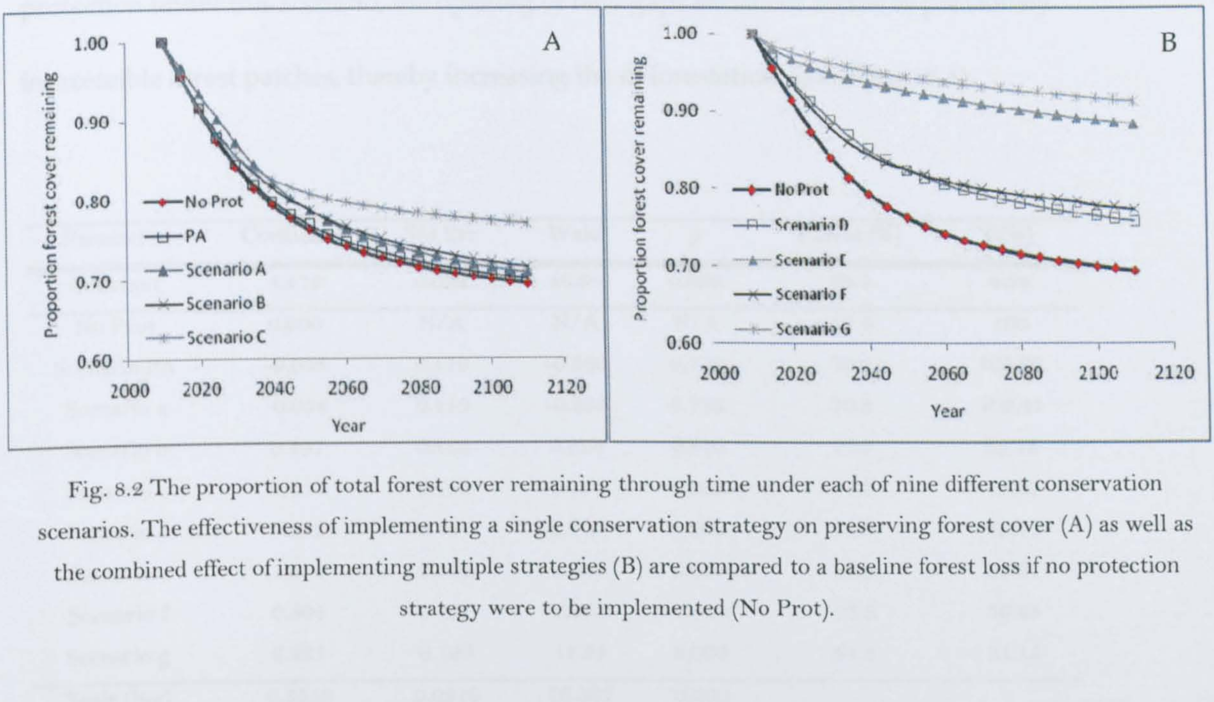

Fig. 8.2 The proportion of total forest cover remaining through time under each of nine different conservation scenarios. The effectiveness of implementing a single conservation strategy on preserving forest cover (A) as well as the combined effect of implementing multiple strategies (B) are compared to a baseline forest loss if no protection strategy were to be implemented (No Prot).

When both the most threatened patches as well as the forest surrounding gaps received protection (Scenario D), forest loss was reduced (6.4%) as compared to the effect of both interventions separately (0.6% and 1.4%, *Scenario A/B respectively*). Protecting forest surrounding gaps, which were most prone to deforestation, while simultaneously reducing forest access by enforcing a buffer around the outer forest edge resulted in 88% of the forest to remain by the year 2110 (*Scenario E;* figure 8.2). Yet, unsurprisingly, the greatest forest protection gains were derived from an conservation scenario that focussed on a combination of three intervention methods. This strategy secured the most accessible forest blocks thereby providing wider benefits to the interior forests by diminishing access and leaving 91.3% of the forest untouched by 2110 (table.8.2).

The different intervention strategies also show a clear trend in their respective annual deforestation rates over time (figure 8.3). Although protection of the most threatened forest patches (Scenario A) led to a apparent decrease in the initial deforestation rate, the annual deforestation rate only decreases slowly and over time even exceeds the deforestation rates observed under the other scenarios (figure 8.3a). As highly threatened forest receives protection under this scenario, the opening of new gaps enhances access to previously inaccessible forest patches, thereby increasing the deforestation risk (figure 8.4).

Parameter	Coefficient	Std.Err	Wald	p	Forest (%)	C(%)
Constant	4.472	0.092	48.64	0.000	69.4	4.38
No Prot	0.000	N/A	N/A	N/A	69.4	100
Scenario PA	-0.043	0.119	-0.360	0.719	70.0	103.06
Scenario a	-0.034	0.119	-0.285	0.776	70.8	102.41
Scenario b	0.101	0.122	0.824	0.410	71.5	93.18
Scenario c	0.407	0.130	3.134	0.002	77.0	75.20
Scenario d	0.352	0.127	2.769	0.006	75.8	78.17
Scenario e	1.474	0.163	9.046	0.000	88.3	35.67
Scenario f	0.504	0.131	3.854	0.000	77.3	70.28
Scenario g	2.221	0.196	11.34	0.000	91.3	21.15
Scale (log)	0.3559	0.0216	16.507	0.000		

Table 8.2 Overview of the Survival analysis using time to deforestation as a dependent variable and protection scenario as an independent predictor. The total amount of forest remaining after 20 iteration is given (Forest%) as well as the reduction of deforestation relative to the non-protection scenario (C%)

Avoiding the expansion of gaps (Scenario B) did not show an instant reduction of the annual deforestation rate, but does, however, reduce deforestation rates more strongly over time (figure 8.3a). Resultantly when conservation incentives exclusively aim to prevent the expansion of forest gaps forest clearance progresses only from the forest edges inward.

The opposite effect is found if the distance to the forest edge is increased by establishing a buffer (*Scenario* C). Since the establishment of buffer areas reduced the deforestation risk around the forest edges, it did not prevent new gaps to be opened and to expand into neighbouring forest areas. Under this scenario deforestations rates do not drop instantly, but do show a more profound decrease over time.

Conservation intervention strategies solely aiming to protect forest based on existing threat did not significantly decrease the average forest survival (or hazard) rate (table 8.2). Yet employing a buffer around the forest edge significantly decreased forest loss (34.8% reduction, $\alpha = 0.41$; $p = 0.002$). The protection of forest surrounding gaps, preventing expansion into other forest areas led to a 6.8% decrease in forest loss, however this difference was not significant ($\alpha = 0.10$; $p = 0.410$). Comparing deforestation rates and forest loss across the conservation intervention scenarios applied in this study revealed that reducing forest access by means of a buffer, while simultaneously protecting forests surrounding gaps to preventing expansion, had the most noticeable difference in reducing the deforestation rates (75% reduction, $\alpha = 1.47$; $p = 0.0001$; table 8.2).

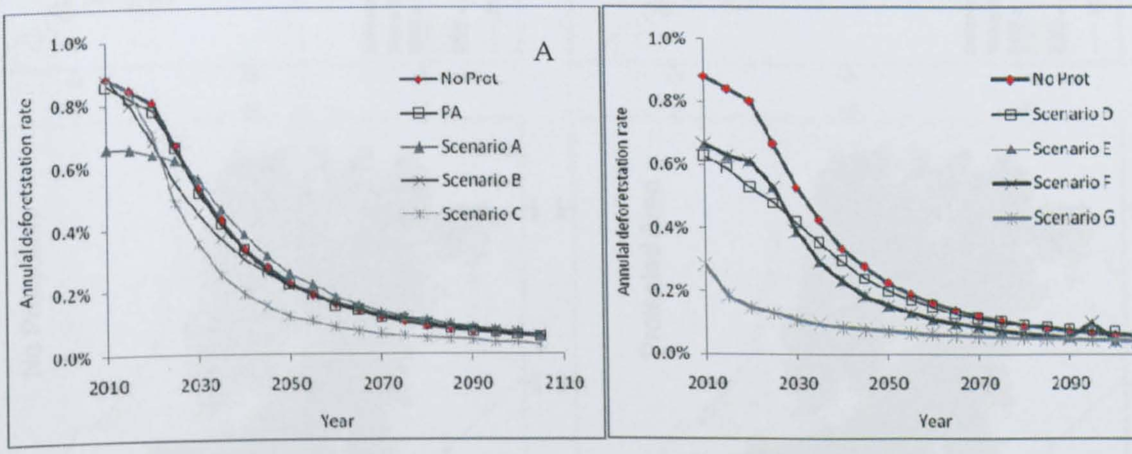

Fig. 8.3 Deforestation rates over time under different law enforcement each of nine different conservation scenarios. The effect of single conservation strategy on reducing deforestation rates (A) as well as the combined effect of implementing multiple strategies (B) are compared to a baseline deforestation rate if no protection strategy were to be implemented (No Prot).

Figure 8.4 part 1

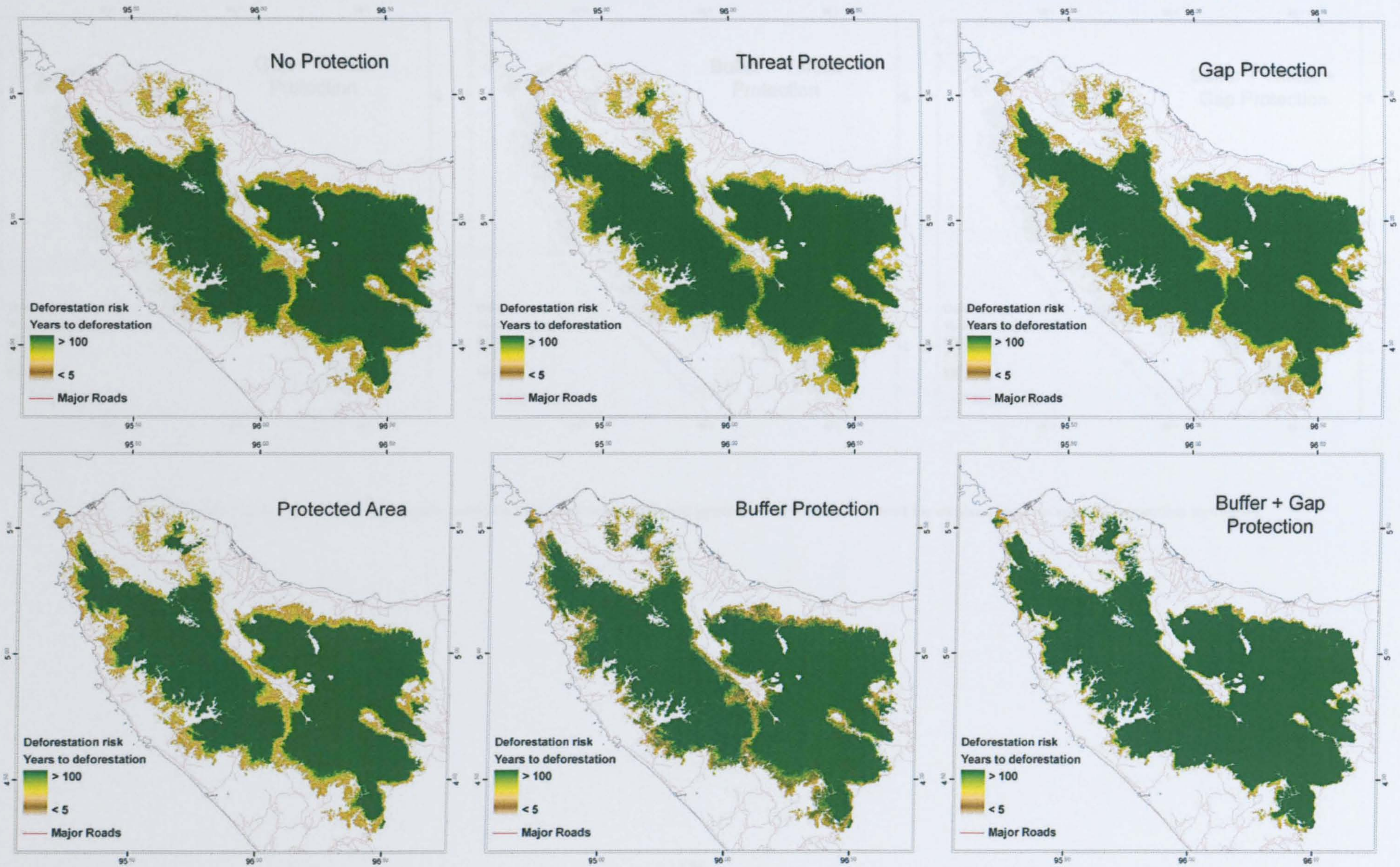

Figure 8.4 Part 2

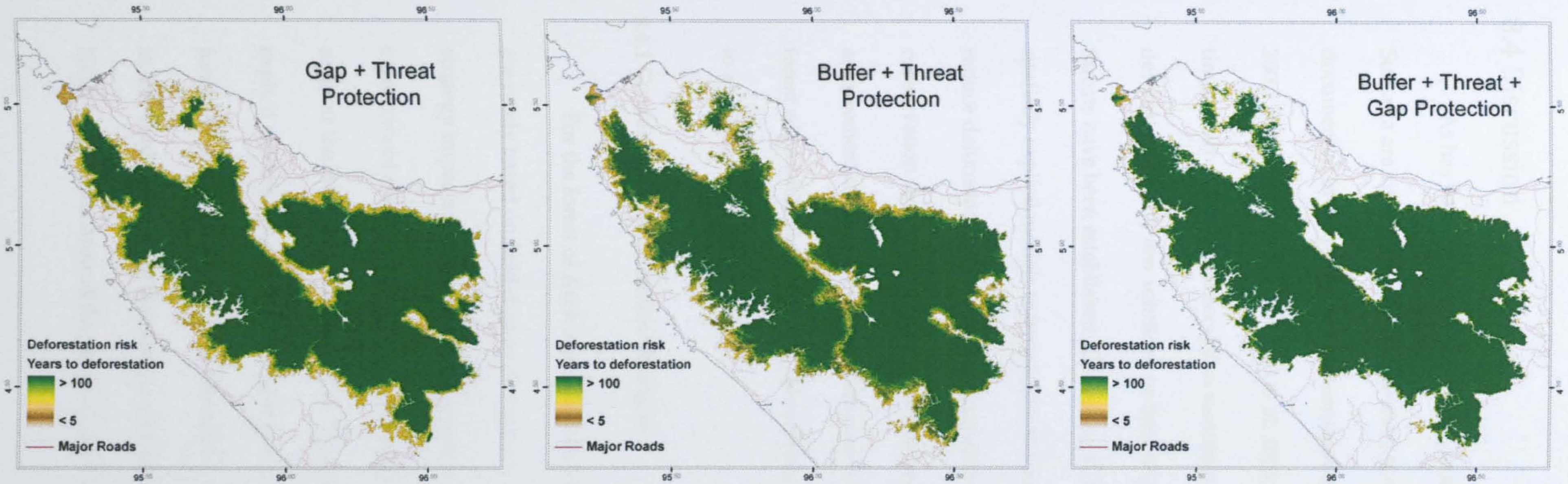

Figure 8.4 Spatially explicit maps of deforestation patterns modelled using different protection scenarios. See text for explanations on specific protection strategies.

8.4 Discussion

As has been shown by the results presented in chapter 7, annual deforestation rates in Sumatra are amongst the highest observed in the tropics, a fact that has been extensively documented in the peer-reviewed conservation literature, (Achard et al. 2002; Gaveau et al. 2007; Hedges et al. 2005; Kinnaird et al. 2003; Linkie et al. 2004, 2006). Although considerable time, effort as well as conservation resources are being spent to study the causes of deforestation, very few solutions on how to reverse these deforestation trends and species threats have been established (Gaveau et al. 2009; Linkie et al. 2008; Linkie et al. 2010). Using spatially explicit models to investigate the potential of different conservation scenarios to reduce deforestation in Aceh, it was possible to gain novel insights on the effectiveness of conservation strategies to reduce deforestation. The models presented here showed that a law enforcement strategy aimed at limiting access to the forest by increasing the distance to the forest edge while simultaneously preventing the expansion of newly opened gaps, predicted to avoid the most deforestation.

8.4.1 Conservation intervention strategies

For the forest of Aceh and most other Indonesian protected areas, protection strategies are rarely based on field data or on rigorous Spatio-temporal assessments of conservation strategy impacts as presented here . Current practices in the province often simply allocate enforcement resources according to the total area of forest present and do not take into account the local threats to the forest. The results presented here have shown that spatially explicit models can contribute to generate knowledge and increase insight on how *in situ* forest conservation should be implemented. From the different protection scenarios presented in this study, it appears that using a strategy aimed to limit access to the forest by applying a 500m wide buffer around the forest edge, considerably reduced the forest loss. Moreover, combining a forest buffer with law enforcement efforts concentrating on the prevention of gap expansion, rather than protecting forest patches based on the current threat of deforestation, was predicted to offset the most forest loss.

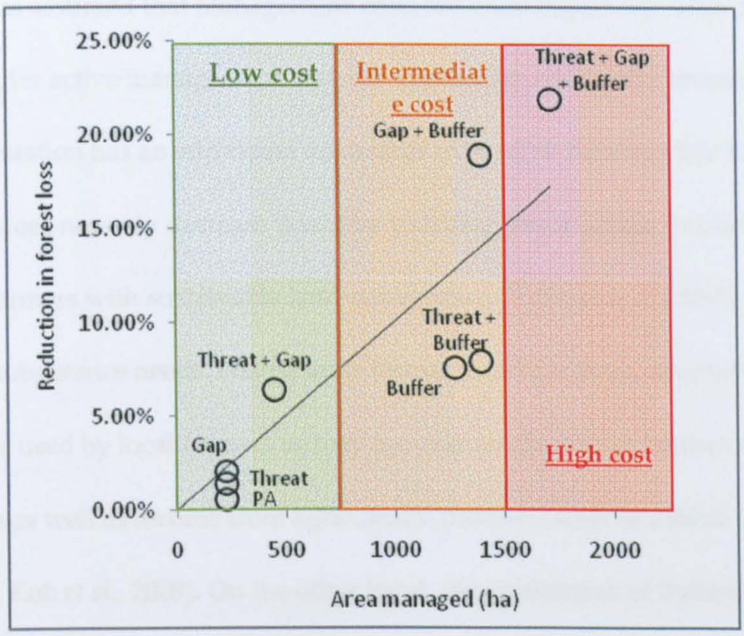

Figure 8.5 Cost-benefit plot showing the net reduction in deforestation plotted against the total area under active management under each conservation strategy (open circles). The estimated reduction in forest loss per hectare managed land is based on the linear correlation between the total area actively managed and the reduction in deforestation realised (solid black line R=0.77). Protection strategies located above this line have a proportionally higher reduction in deforestation than expected and vice versa. Relative operational costs are also shown: low cost=green; intermediate cost=orange; high cost=red.

Comparing the relative costs associated with protecting a certain area of land as presented in this study shows that protection strategies focussing on (1) limiting forest encroachment (gap protection), (2) a combined strategy protecting the most threatened areas while limiting gap expansion, (3) a combined strategy reducing forest access by applying a buffer while simultaneously limiting gap expansion or (4) a combination of all scenarios provided proportionally higher benefits to forest preservation then expected based on the overall model average (figure 8.5). Hence, increasing the current patrol effort aimed to the protection of the two main protected areas present in the region (i.e. low cost, figure 8.5), along with applying alternative patrolling strategies can effectively reduce deforestation in this region.

In absence of robust data on the associated costs of the different strategies proposed in this study, it was assumed that management costs are unambiguously related to the land surface area under active management. Yet, the application of a buffer around forest edges to mitigate deforestation has an additional advantage over other conservation intervention strategies as it does not only decrease threat by reducing forest access, but can also can provide local farmers with sustainable land-use systems (DeFries *et al.*, 2007a), which addresses the subsistence needs. Hence, agro-forests could present a alternative substitute for forest resources used by local farmers as they provide an alternative source of wood and other forest products as well as income from agricultural products such as rubber or coffee (Nyhus & Tilson, 2004; Koh *et al.*, 2009). On the other hand, the application of buffers around protected forest could lead to an increasing deforestation rate in adjacent areas also known as "neighbourhood leakage" (Ewers & Rodrigues, 2008). This process would initiate a landscape in which forest patches are completely isolated or in which unprotected patches would be compromised (DeFries *et al.*, 2005).

Preventing entry to the forest by allocating buffers is sensible, as it should increase the costs associated with clearance, e.g. travel time to market from the location. Such a strategy is also anticipated to increase the probability of encroachers being detected which, for wildlife protection, has been shown to act as a greater deterrent in mitigating illegal activities, such as poaching, than indirect intervention, such as fines or protected area status (Leader-Williams *et al.*, 1990). The effectiveness of a conservation strategy would therefore depend on the ability of local authorities to limit potentially detrimental activities in protected forest as well as creating a situation in which local communities are able to meet their economic requirements without needing to rely on forest resources.

8.4.2 Model validation

The result presented in this chapter provide new insights on the potential of different conservation strategies to safeguard the forest estate of Aceh. Yet the significance of these conclusions should be interpreted given the possible limitations of the modelling framework used. The baseline deforestation rate used to predict future patterns of deforestation was modelled based on historical deforestation patterns observed across the province. Hence, it is assumed future deforestation processes would continue at the same pace as observed during the five years (see chapter four). Yet exogenous factors, such as local economic forces and the prevailing political climate in the province, are likely to influence the demand for timber and hence the deforestation pressure. Other incentives coming form forest conservation such as PES and REDD schemes could potentially lead to a shift in forest exploitation from a source of timber to a source of environmental services or carbon storage (Van Beukering *et al.*, 2008).

The incorporation of a deforestation threshold enabled the models to simulate a reduction of deforestation rates over time, leaving forest patches in the most remote areas untouched. Other parts of Sumatra have shown similar patterns in which sub-montane and montane areas were less likely to be converted to farmland (Gaveau et al. 2007; Linkie et al. 2010). Still, changing timber markets will eventually lead to a situation where the profits of exploiting even the most remote areas to meet timber demands will eventually exceed the costs of operating in these areas. Hence, future models should investigate the use of additional parameters in order to realistically incorporate dynamic market forces into deforestation models.

Factors driving deforestation are likely to change over time and, to reflect these changes, models predicting deforestation over time should explicitly incorporate this variation to model temporal trends. In this study, this was partially controlled for through the construction of revised distance to forest edge covariate after each annual forest loss stage. Although useful, other, more complex relations should be investigated to realistically predict future deforestation patterns. The use of autocorrelation functions in which covariate coefficients are modelled as a function of forest availability could form a valuable first step to realistically incorporate temporal changes in deforestation pressure.

Likewise, a spatial component could be added to model the spatial interaction of drivers of deforestation. The protection scenarios presented in this study assigned full protection to the focal areas through a minimum risk threshold value. Even though such generalizations are useful to study the effect of different intervention strategies, this could be enhanced through modelling the gradual effects of forest patrols and spatial shifts in deforestation pressure resulting from intervention strategies.

GENERAL CONCLUSIONS

The results presented in this thesis disclose key information regarding the current status of elephant habitat and populations in Aceh and Sumatra and bring to light the threats elephants are currently faced with throughout their range. Hence it provides a framework from which conservation strategies can be implemented, enabling different stakeholders to develop and endorse conservation plans based on timely and ample data. In the past, such information often remained ambiguous or was based on subjective judgments rather than factual field data, making conservation strategies unlikely to engage the true problems threatening species survival (Meijaard & Sheil, 2007). Hence, this thesis contributes fundamental knowledge valuable for the preservation of wild ranging elephants and promotes effective protection strategies to safeguard the future survival of elephants and their habitat in Aceh and Sumatra.

To ensure the future survival of Sumatran elephants within a landscape increasingly dominated by a mosaic of forested and agricultural areas, an urgent need exists to understand how elephants respond to alterations of their existing habitat. Therefore, this thesis has provided a firm and robust framework based on which conservation strategies can be developed. New insights on elephant niche dynamics, habitat use and the provision of a spatially explicit habitat map have allowed to asses how elephants utilize their niche while simultaneously assessing the current status of the remaining elephant range. Additionally, a better understanding of the effects of deforestation and habitat encroachment on elephant persistence and the instigation on human-elephant conflict has provided valuable knowledge to encourage the coexistence of human and elephants in a multi use landscape matrix .

Deforestation in Aceh

Over recent years, conservationists working in various tropical regions have emphasized the importance of forest ecosystems to counteract the degradation of ecosystem services and loss of biodiversity (Brookes, 2002; DeFries, 2007; Gaveau *et al.*, 2007; Koh *et al.*, 2009). The enforcement of protected areas, however, has often failed to successfully preserve tropical forests (Curran *et al.*, 2004) and its biodiversity (Peh *et al.*, 2006). In Aceh, government financed transmigrations from Java to the northern parts of Sumatra in the early 1990s led to massive amounts of forest being converted to small-scale farmlands (Holmes, 2002). Moreover, economic expansion of the Asian continent and Aceh likewise led to a reduction in forest cover of no less than 60.4% of total forest loss between 1984 and 1997 (Holmes, 2002).

Even though deforestation rates as high as 5.50%/yr (Achard, 2002) have been observed over the last thee decades, the results presented here have shown Aceh to be an exception from such practices. Mapping forest cover, based on remotely sensed imagery, has shown that over the period 2005-2009 only a marginal increase in deforestation was observed, but annual forest losses did not significantly exceed the total period average of 1.1%/year (chapter 4). This observation is largely explained by the fact that the absence of an extensive and well maintained timber network in this province has limited access to areas rich in timber resources. The armed conflict, which lasted from 2001 until the tsunami in December 2004, withheld commercial enterprises to harvest timber in the province and prevented large areas from being cleared. Yet, the renewed peace in Aceh and the strong economic interest in non sustainable forest exploitation for the development of estate crops, including palm oil, have recently increased pressure on the remaining forest estate. In addition, unclear protected area demarcation, insufficient funds for protection, conflicting benefits and large scale corruption of funds by local authorities are now jeopardizing forest conservation efforts in Aceh. Hence alternative forms of sustainable land use practices which also meet local economic interests will need to be investigated.

Potential financial incentives to preserve forest, including revenues from carbon trade and ecosystem services such as water sanitation and natural pest control, have recently received increasing international attention (Blom *et al.*,2010). The current estimates of forest cover and deforestation for the northern forest of Aceh presented in this thesis involve a first step in realizing a framework for the implementation of carbon-financing based on reduced deforestation schemes (e.g. REDD) in Aceh. The provision of forest cover maps and forest cover estimates allow to determine baseline deforestation rates for this area. Consequently, they provide information necessary to assess the baseline loss of forest cover and carbon stocks likewise which can be used for the REDD purposes. These result therefore provide valuable information on the current state of the forest in Aceh which are critical to meet REDD guidelines outlined in the IPCC COP 13 action plan. Since, REDD schemes accredited under the CoP 13 convention (UNFCCC, 2007) require robust estimates of background deforestation to determine net losses of carbon stocks, temporal trends in deforestation rates can be used to establish baseline deforestation rates. Hence baseline deforestation rates, as presented in this thesis, allow to produce estimates of the potential benefits from reduced deforestation which can be realised.

As has been shown in this thesis, deforestation rates in northern Aceh (1.1%/year) are amongst the lowest found in Sumatra and Borneo (2.56%/year: Gaveau, 2007, 1.7%/year Langer, 2009). This causes a controversy when establishing financial gains from reduced deforestation if these would be determined based on local deforestation rates. Lower deforestation rates in Aceh would lead to a reduction of the net benefits generated from prevented deforestation when compared to other provinces where the forest is rapidly cleared. Yet, REDD schemes based on estimates of future deforestation rates can provide a viable alternative to compensate deforestation agents (Gaveau 2009). Also, with an estimated forest cover of 9920.0 km^2 this part of Aceh remains one of the most forested areas in the Indonesian archipelago. A REDD scheme in which a nation wide deforestation rate would be used to determine carbon profits are therefore be highly beneficial to the region.

Elephant habitat use

The changing landscape across northern Aceh and the use of elephants of this area presents a conservation dilemma. Whilst elephants did indeed reside at forested edges rather than at the primary forest interior, it is unclear how deforestation will affect elephants in the long-term. In Aceh, elephant habitat use was found to be limited by the availability of lowland forest. Hence, continuous conversion of forests and the further deterioration lowland habitats accordingly, is expected to decrease the survival chances of the remaining elephant populations. Elephants were shown to have a strong preference for forest edges with a high productivity located within valleys. Yet, as secondary regrowth is often abundant along forest edges, these areas are generally rich in elephant foliage, which in return could benefit elephants living on the forest non-forest interface. Yet, while elephants might prefer flatter, lowland area and topographic depressions, empirical data has shown that elephants did utilize mountainous and rugged terrain. Terrain ruggedness, however, does seem to constrain elephant niches to some extent, as elephant presence was less profuse within highly rugged terrain .

From these results it has become clear that elephants frequently occupy a wide range of optimal as well as sub-optimal habitat. Many of these areas, however, are unlikely to support viable elephant populations for extended periods of time. This idea is supported by the results presented in chapter 7, where it was shown that elephant populations are prone to extinction if large stretches of formerly suitable habitat had been cleared in the past. The prevalence of elephants within marginal habitats is therefore believed to therefore merely reflect a delayed population response to the conversion of previously suitable habitat. On the other hand, elephant presence in areas currently classified as comprising suitable elephant habitat has not been verified in all cases. Future work, should therefore aim to identify the minimum amount of habitat as well as the spatial configuration of suitable habitat patches which is needed to support a viable elephant population.

Human Elephant Conflict

One of the most prominent opinions among conservation biologists studying the effect of anthropogenic influences on wildlife distributions is that habitat alterations is the biggest threat to the survival of wildlife today (Laurance, 1999; Achard et al., 2002; Brook et al., 2003). As elephant habitat increasingly becomes encroached, the competition between humans and elephants for suitable living is likely to escalate the frequency of conflicting encounters between humans and elephants, decreasing the willingness of local farmers to participate in conservation schemes necessary to ensure the future survival of elephants in this and other Sumatran provinces (Nyhus et al., 2004; Rood et al., 2008; Uryu et al., 2008). Forest encroachment and elephant habitat destruction accordingly, was found to be common in areas of high habitat suitability. At least four percent of suitable elephant habitat was found to be lost on a yearly basis, four times exceeding the annual amount of forest loss observed (1.1% /year; chapter 3). Hence if forest encroachment continues at its current rate, elephants will be forced to survive within a landscapes completely dominated by humans (Rood et al., 2008; Rood et al., 2009).

Faced with the continuous conversion of lowland habitat over the last decade, elephants do not inevitably respond by moving away into new, undisturbed, areas. Rather they were often found to reside within smaller patches of less suitable habitat. In those cases where natural habitat has been totally converted into a matrix of secondary forest, pastures or estate crop plantations, crop raiding behaviour by elephants is now common. The incidence of crop raiding by elephants was shown to be concentrated in areas which both recently have become deforested, have a low forest cover remaining and are in the direct vicinity of human populations (e.g. close to roads). A decrease in forest cover, however, does not unambiguously lead to an increase of crop raiding. Yet, elephants which inhabit degraded forests or areas with high levels of secondary forest regrowth, are highly likely to raid crops. Likewise, as the remaining forest patches are being cleared for agricultural expansion, the frequency of crop raiding by elephants is likely to increase . The occurrence of crop raiding in areas where both undisturbed as well as secondary forest habitats are common emphasizes the fact that these habitats, to some extent, encompass natural elephant habitat.

Elephant conservation

Since the mid 1980s, the Indonesian governments' response to mitigate human elephant conflict by capturing large numbers of elephants and moving them to Elephant Training Centers (ETCs) has greatly impacted wild elephant populations . The lack of proper management and the absence of funds have in many cases resulted in high mortality rates within ETCs (Mikota *et al.* 2008). More strikingly, government institutions including the ETCs have frequently been associated with the illegal trade in elephant products. Poaching and trading of elephant products are known to be widespread in Sumatra (Shepherd, 2009; *pers. comm.*). Hence, the government regulated captures of wild elephants justified by the occurrence of human elephant conflict are believed to put an tremendous pressure on the remaining populations.

In many parts of Sumatra, habitat destruction, reprisal killings of elephants by local farmers, elephant poaching for ivory and government regulated captures have lead to the eradication of isolated subpopulations (Hedges *et al.*, 2005; Uryu *et al.*, 2008; Rood et al., 2006). Accordingly, twenty three distinct elephant populations or 52% of the populations known to exist in 1990 were found to have gone extinct by 2005, corresponding to a decline of 45% in total elephant range size between 1990 and 2005. Elephant subpopulation extinctions observed between 1990-2005 were found to be most strongly affected by the presence of anthropogenic influences, but to be also closely related to the amount of forest available to elephants in 1990. Yet elephant populations occurring in areas which experienced high deforestation rates over the last decade were did not show an increased extinction probability. Even though these results lead to believe that it is unlikely that forest destruction *per se* strongly affect elephant populations directly, deforestation has been shown to be the foremost cause of alterations of ecosystem integrity (Meijaard *et al.*, 2005; DeFries *et al.*, 2007a). Compromising ecosystem properties such as nutrient cycles, resource availability, natural migration routes and landscape configuration could eventually lead to a reduction of the system carrying capacity and an increased mortality within elephant populations (Sukumar 1989). Hence, while forest reduction in itself might not significantly influence elephant survival, land use and land cover change can have a considerable impact on wild elephant populations which can ultimately lead to local extinctions.

Concluding remarks

As land use planning for conservation landscapes within and outside established conservation areas is becoming a new standard in large mammal conservation practices (Nyhus & Tilson, 2004; Linkie et al., 2006), the effects of land use configuration, elephant behaviour and human response are amongst the most important issues to account for when setting long-term elephant conservation priorities. The habitat analysis and spatially explicit habitat suitability model presented in this thesis therefore provide an initial step to identify and prioritize core areas for elephant conservation. Hence, local authorities have been provided with the foremost tools to incorporate species conservation priorities to be built on when future spatial plans for the region are developed.

In order to effectively address elephant conservation issues, active law enforcement, and sound forest management will be critical. In order to develop effective conservation strategies as to guarantee the survival of wild ranging elephants in the future, conservation management strategies should aim to halt further forest encroachment and elephant habitat conversion. Ignoring to do so would lead to a further loss of the natural carrying capacity of the area and an escalation of human elephant conflict resultantly. This in turn would make conservation efforts less likely to find support amongst local stakeholders and reduce the willingness to preserve the last remaining patches of suitable elephant habitat. Such vicious circle could eventually lead to a complete suppression of wild elephant populations occurring in Aceh and the rest of Sumatra as has been witnessed with other wildlife species in Indonesia such as the Java Elephant and the Javan tiger (Corlett 2010).

The protection of forest in order to conserve wildlife has historically been marginally successful as the species richness living in Asian forests has continuously declined over the last century (Sodhi et al., 2010). However, other sustainable forest commodities such as carbon revenues and agro-forests could present an apt alternative livelihoods for local farmers as they can provide an alternative source of income making forest conversion a financially less encouraging (Beukering *et al.* 2008; Koh *et al.*, 2009; Blom *et al.*, 2010). Constraining access to the forest by allocating agroforest buffers is sensible, as it should increase the costs associated with clearance. Such a strategy is also expected to increase the probability of encroachers being noticed which, has been shown to act as a greater deterrent in mitigating illegal activities, such as poaching, than indirect intervention, such as fines or land tenure(Leader-Williams *et al.*, 1990). The effectiveness of a conservation strategy therefore depends on the ability of local authorities to stop activities jeopardizing forests from happening while simultaneously creating opportunities for local communities to meet their economic requirements without needing to rely on forest resources.

SYNTHESIS

Over the last three decades forest loss has been rampant throughout Indonesia where more than 17% of forest was cleared between 1990 and 2005. Here, the spatial pattern and rates of deforestation were derived by analyzing remotely sensed imagery covering the Ulu Masen forest block, Aceh, Indonesia. Annual forest loss in Ulu Masen was lower than observed over the whole Sumatra (1.1%/year vs. 2.56%/year). Deforestation was most likely around the forest edges and was strongly correlated to local infrastructure.

Elephant habitat use was assessed by means ecological niche modeling. According to their optimal niche requirements elephants were found to be mainly confined to closed canopy forest with a high productivity located along the forest edges. Elephant distribution did not appear to be constrained to lowland flats as they were also found at higher elevations and in rugged terrain. A comparison of the occurrence of suitable elephant habitat throughout Aceh, revealed that elephants occur at the margins of the ecological conditions present in the area. This could indicate that as forest conversion continues, elephants are slowly being displaced from their natural habitat.

When compared to the elephants optimal niche, crop raiding elephants were found at ecological conditions located at the margins of the elephants niche. Crop raiding was found to be most likely in areas which recently had been deforested but still hold patches of secondary forest. Yet, cop raiding was to a lesser extent correlated to landscape topography and occurred at a large elevation gradient. Hence, unsustainable logging practices throughout the province will instigate an increase in the occurrence of human-elephant conflict. As crop-raiding by elephants, amongst other crop raiding species, can have a large economic impact on small communities living near the forest, a further reduction of forest habitat could eventually lead to an explosive increase in crop raiding and hence a lower tolerance toward elephants.

Since the alterations of elephant habitat are believed to have a negative impact on the prevalence of wild elephant populations across Sumatra, the occurrence of population extinctions was investigated using Sumatra wide deforestation data as well as anthropogenic influences. Elephant populations were found to have experienced a severe decline since 1980s. Out of the forty-four populations recognized in the 1980's, twenty-three populations are believed to had gone extinct by the year 2005. Likewise 45% of elephant ranges disappeared over this period. Extinctions were found to be strongly related to human presence and past forest cover, indicating a delayed response to deforestation and hence more extinctions are expected within the near future.

Finally the effectiveness of several conservation strategies to reduce deforestation were explored by means of predictive modeling. From this work it appears that deforestation is unlikely to significantly reduced by the enforcement of current protected area, or by providing full protection to the most threatened forest patches. A protection strategy in which access to the forest is reduced limiting access by, for example, erecting a buffer around the forest estate was found to be the most effective forest conservation strategy. This scenario is most likely to be successfully enforced since it would not rely on the high investment needed to chase loggers throughout the forest, but is limited to surveying the forest edges. Moreover, the establishment of buffer zones, in which agricultural exploitation would be possible, would additionally provide alternative livelihoods for local residents, diverting incentives from unsustainable logging practices.

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