A Spatial Ecological Assessment of Fragmentation and Disturbance Effects of Infrastructure Construction and Land Conversion in Gunung Halimun Salak National Park, Indonesia

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Habitat loss and fragmentation have been identified as one of the sources of biodiversity loss. Those issues are predominantly triggered by human activities such as the development of settlements, conversion of forest habitat into agricultural areas, and the development of infrastructure such as transportation and pipelines. Gunung Halimun Salak National Park (GHSNP) as one of the protected areas in Indonesia is deforested each year by approximately 1,473 ha or 1.3% of the total area while the capability of GHSNP to rehabilitate the degraded forest is only 500-800 ha annually because of a limited budget and at the shortage of human resources.

The focus of this study was primarily on an ecological assessment of landscape scale habitat impacts on mammals and birds caused by infrastructure construction and land conversion. This was accomplished, by firstly identifying the current land cover and land cover change over a 15-year periods within the study area, GHSNP, Indonesia, as a case study; secondly, by assessing a class-level landscape metrics and their changes to detect forest fragmentation over a 15-year period within GHSNP; and finally, by analysing changes in the habitat network caused by fragmentation, land conversion, and disturbance. The potential effects on wildlife were also discussed, including how to derive and interpret such fragmentation effects and disturbance by applying Geographical Information System (GIS)-based quantitative modelling.

Results of this study show that land cover in GHSNP has changed drastically with Forest and Agriculture experiencing the biggest decrease and increase by 35.63% and 463.74% respectively. It was also found that landscape metrics for forest cover in GHSNP show changes with a decrease in class area (CA), patch size (MPS), patch size coefficient of variation (PSCoV), mean shape index (MSI), mean core area (MCA) and mean proximity index (MPI) and an increase in number of patch (NP) and

edge density (ED). These suggests that the forest cover in GHSNP has been fragmented for the past fifteen years.

Regarding the effects of fragmentation and disturbance on mammals and birds, the responses of the forest-grassland mammals (high area and medium area demands), forest mammals (area below 1500 metres above sea level and area above 1500 metres above sea level and bird profiles (forest area) were similar to each other, with class area (CA) and number of patches (NP) being reduced as a response to both effects, except for small mammals requiring forest area below 1500 metres above sea level, which underwent a relatively huge increase in the number of patch for the disturbed area. Nevertheless, among all Ecological profiles established, mammals having a high area demand are considered to be most impacted from fragmentation and disturbance effects.

Keywords: Geographical Information System, frgamentation, disturbance, road, biodiversity, impact, land cover, landscape metrics, national park, forest, birds, mammals

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Chapter 1

Introduction

1.1 Background

Indonesia is the highest ranked among the mega-biodiversity countries and thus, biologically, is one of the wealthiest nations in the world (Mittermeier, 2004; Myers, Mittermeier, Mittermeier, Da Fonseca, & Kent, 2000). Myers et al. (2000) states that 80% of its area is a world biodiversity hot spot (Figure 1.1). In terms of biodiversity richness, Indonesia has some 12% of the world's mammals (515 species, 39% endemic), 16% of the world's reptiles and amphibians (511 reptile species, 29% endemic, and 270 amphibian species, 40% endemic) and 17% of the world's birds (1531 species, 26% endemic) (Mittermeier, 2004). However, landscapes, land uses and land cover in Indonesia are undergoing a massive transformation in response to a variety of economic, demographic and policy factors, especially after the economic and political crises of 1997. Those enormous changes in its landscape have led to environmental degradation, and have resulted in a decrease of green open spaces; an increase in water, soil, and air pollution; and a loss of biodiversity, predominantly in the most populated island of Indonesia, namely Java (Arifin & Nakagoshi, 2011).

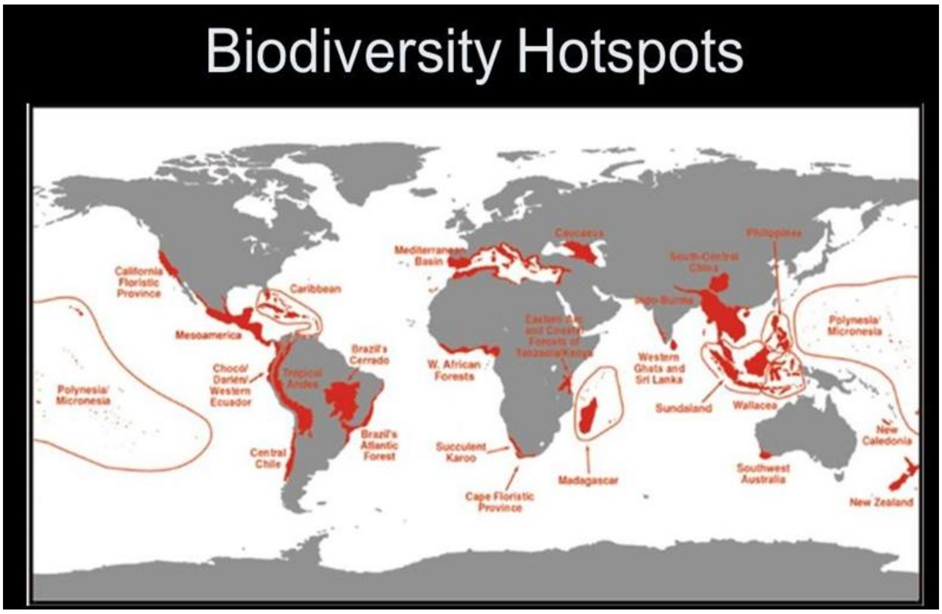


Figure 1.1 The world's 25 biodiversity hotspots (Myers, et al., 2000)

In Indonesia, national parks as protected areas are the basis of biological conservation. However, most of the protected areas are experiencing external pressures from both encroachment and degradation. One of the main threats in these protected sites is habitat loss and degradation due to land conversion for pastureland and agricultural purposes (Alers, Bovarnick, Boyle, Mackinnon, & Sobrevila, 2007). Due to such issues, Indonesia has been recognized by all international conservation priority-setting exercises as a top priority for actions to conserve biodiversity (Kitchener, Brown, Merrill, Dilts, & Tighe, 2004).

Habitat loss and fragmentation have been identified as one of the sources of biodiversity loss (Chaves & Arango, 1998; Terborgh, 1989). According to Heywood and Stuart (1992), the rule of thumb for a habitat-species relation is a 90% loss of habitat leads to a 50% loss of species.

Habitat loss is mostly caused when large areas of natural landscapes are impacted by human activities (Bogaert, Farina, & Ceulemans, 2005), such as the development of settlements, conversion of forest habitat into agricultural areas or the development of infrastructure such as transportation and pipelines (Forman, 1995; Llausàs & Nogué, 2012; Slonecker et al., 2012). Such activities, however, in the long run will impact biodiversity richness and abundance at genetic, species and ecosystem levels and thus, each level should be considered in an environmental assessment (Slootweg & Kolhoff, 2003). As an example, on Borneo, species richness and diversity of butterflies have declined significantly with the increase in fragmented habitat and thus, endemic species were not recorded within small remnants (less than 4000 ha). However, this remaining area still plays a significant role, due to its contribution to regional biodiversity (Benedick et al., 2006).

In environmental assessment, particularly regarding habitat loss and fragmentation effects, the limited use of quantitative methods is still a major hurdle. Moreover, recent studies show that current practice depends mainly on expert knowledge, and that not even fundamental landscape characteristics such as habitat amount and the number of habitat patches are used to inform decision making. Indicators such as habitat amount and connectivity, as well as anticipated habitat loss and fragmentation, are important considerations when addressing the impacts on ecological processes and biodiversity and thus, quantitative methods could complement current methods for such assessment (Karlson & Mörtberg, 2015).

Since biodiversity issues related to habitat fragmentation have raised deep concerns, methods for modelling and quantifying the effects of fragmentation are necessary in impact valuation. In this case, Geographical Information System (GIS) are efficient tools for impact assessment (Şahin & Kurum, 2002) and the combination of GIS and quantitative environmental modelling has offered new perspectives in integrated science (Clarke, Parks, & Crane, 2000; Gontier, 2007; U. Mörtberg, Zetterberg, & Balfors, 2012; Zetterberg, Mörtberg, & Balfors, 2010). However, some issues still require deep attention in reserve design and include the viability of protected populations and spatial considerations such as habitat fragmentation, connectivity and the spatial distribution of each species (Cabeza, 2003). For ecological impact assessment efforts that apply quantitative approaches, species with similar habitat requirements are classified by their traits and thus, ecological profiles can be developed to represent each group's response to environmental change (Angelstam, Edman, Dönz-Breuss, & DeVries, 2004; U. Mörtberg et al., 2012; Vos, Verboom, Opdam, & Ter Braak, 2001). This approach could bridge the gap between ambitions and current practice, and allow for quantified predictions and a more systematic ecological impact assessment. In summary, exploring the use of GIS-based quantitative methods for modelling the ecological effects of human impact activities is seen as beneficial for generating baseline environmental information and can provide coarse predictions for evaluating conservation options.

National park management in Indonesia faces major challenges in preventing further loss of biodiversity. If remote sensing and GIS are to be effective, then an appropriate method for ecological impact assessment is vitally important.

The focus of this research was primarily on a quantitative ecological assessment of landscape scale habitat impacts on mammals and birds caused by infrastructure construction and land conversion. This was accomplished by firstly identifying the current land cover and land cover change over a 15-year periods within the study area, Gunung Halimun Salak National Park (GHSNP) Indonesia, as a case study; secondly, by assessing a class-level landscape metrics and their changes to detect forest fragmentation over a 15-year period within Halimun Salak National Park; and lastly, examining changes in habitat network due to fragmentation, and disturbance; along with its potential effects on wildlife, including how to derive and interpret such fragmentation effects and disturbance by applying GIS-based quantitative modelling.

1.2 Study Aims and Objectives

The aims of this study were:

1. To establish the current land cover and analyse the land cover and forest cover changes within GHSNP;

2. To assess spatial patterns of forest fragmentation in GHSNP by utilizing landscape indices (metrics) to analyse fragmentation;

3. To assess the overall impact of land conversion and transportation infrastructure on mammals and birds in the study area by modelling the effects of infrastructure construction identified in the literature, and;

4. To explore potential effects of fragmentation and disturbance on selected groups of mammals and birds, by analysing changes in the habitat amount and connectivity of habitat networks for designated ecological profiles.

The specific objectives to be addressed in this study were:

1. The magnitude and the rate of land cover changes that have occurred between 2001 and 2016, based on remotely sensed data;

2. The magnitude and the rate of forest cover changes that have occurred between 2001 and 2016;

3. Comparative analysis of class-level landscape metrics within Gunung Halimun Salak National Park between 2001 and 2016;

4. Comparison of road effect zones within forest areas by firstly calculating Mean Species Abundance (MSA) of birds and mammals, and;

5. Comparison of the landscape metrics, namely number of patch (NP) and class area (CA) which describe the structural properties of habitat networks of each ecological profile established.

The conceptual approach of the study was to explore the utility of GIS-based quantitative modelling methods by integrating biodiversity components for ecological modelling, which support impact assessment of fragmentation and disturbance due to land conversion. To accomplish this objective, the study comprised several tasks:

1. Conduct ground truthing to test the accuracy of image interpretation at research study and to clarify interpretation assumptions of land cover;

2. Derive and build thematic maps based on databases obtained from the Ministry of Forestry and Environment as well as GHSNP Office;

3. Collect, analyse and organize biodiversity data either from field investigation and reviews of current literature review;

4. Collect, classify, and assess the accuracy of LANDSAT satellite images obtained from of 2001 and 2016 to generate the land cover and land cover change using ERDAS and ArcGIS;
5. Calculate class-level landscape metrics of GHSNP forest cover in 2001 and 2016 using Patch Analysis;
6. Construct and run the modelling of spatial distribution of road effects in ArcGIS;
7. Create ecological profiles of mammals and birds through a review of the literature and information obtained from GHSNP;
8. Construct habitat network layers for those ecological profiles by using focal statistics in ArcGIS;
9. Generate such fragmented and disturbed habitat network layers using ArcGIS, and;
10. Calculate the landscape metrics, namely total habitat area (CA) and number of individual patches (NP) using Patch Analysis.

In general, the study generated an ecological impact assessment at a landscape level that illustrates how forest habitat and wildlife are impacted by land cover changes in the study area. Therefore, it is expected that the research will help or lead to the development of GIS-based quantitative modelling that will be beneficial in assessing the ecological impact, particularly such impacts occurring in GHSNP. Finally, the output can be a reference in formulating appropriate conservation measures at GHSNP and other similar national parks.

1.3 Thesis Structure

This thesis is structured as follows:

Chapter 1 presents the general outline of the thesis.

Chapter 2 describes the study area and the geography, demography and current issues occurring at the study area.

Chapter 3 reviews relevant literature covering introduction to landscape, land cover, fragmentation and disturbance, impact assessment and the use of Geographic Information System (GIS) and Remote Sensing for ecological assessment.

Chapter 4 explains the general research design and the methodology adopted to achieve the study objectives.

Chapter 5 presents the study results, including land cover patterns and changes, forest cover fragmentation analysis and how such fragmentation and disturbance impact the mammals and birds.

Chapter 6 presents the discussion of each outputs of the study and further discusses the implication and management of GHSNP area and its mammals and birds.

Chapter 7 draws the conclusions and provides the recommendations for further studies.

Chapter 2

The Study Area

2.1 Overview

The study area is GHSNP, located in the West Java Province of Indonesia. This national park was established in 1992 with total area of 113,357 ha. Since its establishment, this protected area has often been used for the purposes of research, education, breeding enhancement, recreation, and tourism.

2.2 Indonesia

Indonesia is renowned as the largest archipelagic state in the world, stretching 5,110 km along the equator from east to west and 1,888 km from north to south. It is comprised of five major islands (See Figure 2.1) (Java and Bali, Sumatra, Kalimantan, Sulawesi, and Irian Jaya) and about 30 smaller groups, with over 17,000 islands in total (Sugiyarto, Blake, & Sinclair, 2003). Culturally, it is home to 300 ethnic groups and over 500 different traditional languages and dialects (Sabandar, 2004). In terms of biodiversity, Indonesia is biologically the world's most diverse country since being located between the two major biogeographical regions of Australasia and Indo-Malaysia (Baines & Hendro, 2002).

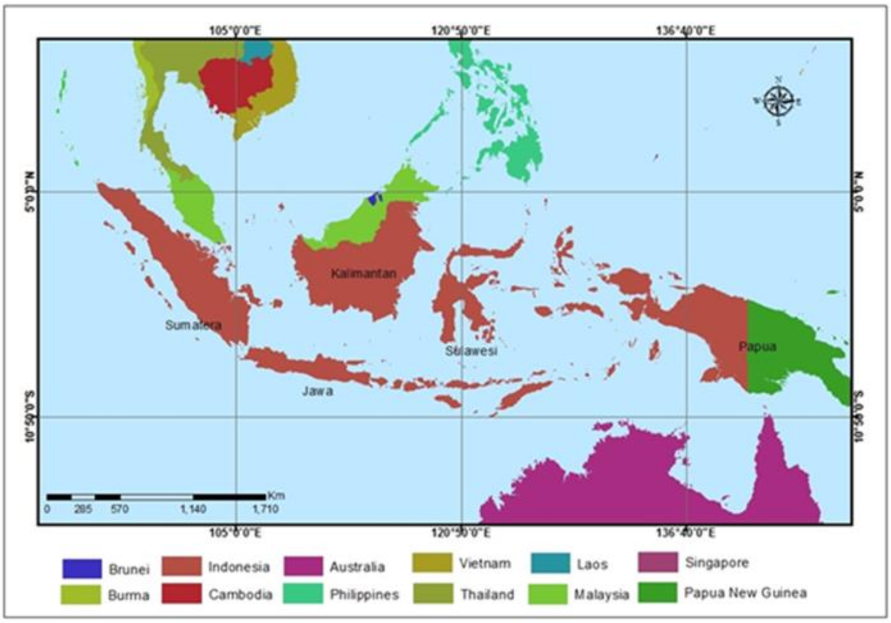


Figure 2.1 The five main Islands of Indonesia (B. I. Geospasial, 2016; Institute, 2011)

2.3 Java Island

Java Island, Indonesia's political and industrial centre, is one of the most densely populated areas in the world. The very fertile soils sustain about 114 million inhabitants, at an average population density of 862 people per km^2 (Whitten, 2000). Java has basically been deforested and the majority of the remaining forest fragments cover the many volcanoes on the island. Due to this, less than 10% of the original forest remains. Currently, deforestation has slowed down, yet fragmentation and forest degradation still persist (Smiet, 1992).

High population numbers on Java have driven forest clearing especially in lowland areas. The remaining natural forests that were destroyed were mostly situated in remote mountain areas with less human activity (Thiollay & Meyburg, 1988; Smiet, 1992; Galudra, 2003). Whitten (2000) estimated that more than 1.5 million ha had already been lost to farmland and teak plantations by 1000 A.D. Prior to World War II, the forests in Java had been reduced to 23% of their original extent (Seidensticker, 1997). By 1973 this figure dropped to 11 %, and by 1990, to an estimated 7% — currently only 0.96 million ha of forest remnants (FAO 1990). Most of the natural forests remaining today are located in national parks or other forms of protected areas, including those for watershed conservation. Those national parks existing in Java include Gunung Halimun Salak National Park (van Balen, 1988).

2.4 Gunung Halimun Salak National Park

2.4.1 Location

Administratively, GHSNP (Figure 2.2) is located in two provinces namely West Java and Banten, covering one district in Banten province and two districts in West Java province, namely Sukabumi and Bogor. It is geographically situated between 106°13'and 106° 46' E and 06°32 'and 06° 55' S (Rinaldi, 2003).

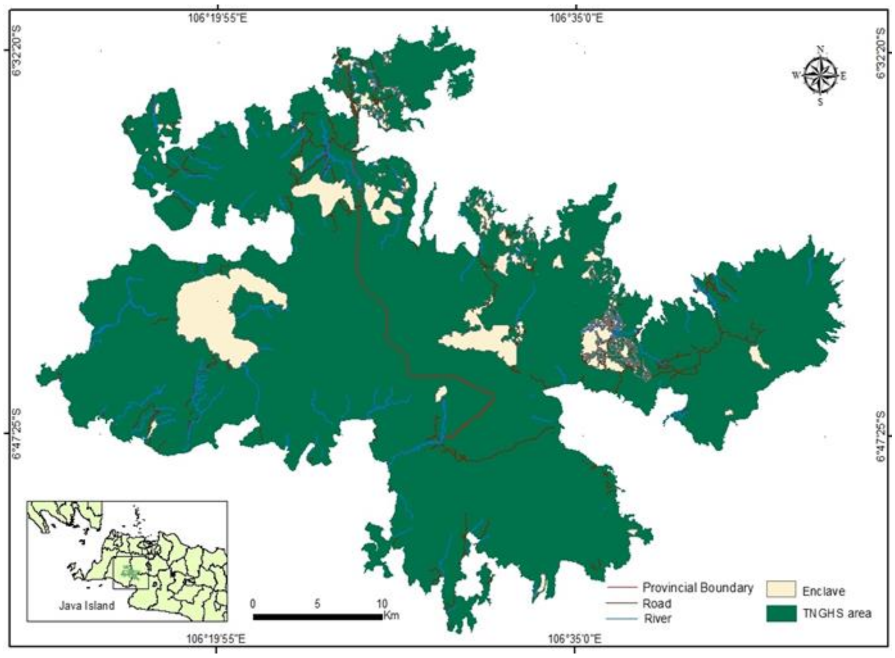


Figure 2.2 Gunung Halimun Salak National Park (T. N. G. H. Salak, 2016)

2.4.2 History of the National Park

The forests in this national park were historically controlled under the successive authorities of the Dutch colonial government, the Japanese colonial government, and the Indonesian government (Harada, 2003). Galudra (2005) identified that in Gunung Halimun-Salak, deforestation began in the 1700s during the colonial era to establish coffee plantations. However, these plantations failed due to plant diseases and initiated the degradation of natural forest. Before the establishment of the park, GHSNP had been recognized as protected forest known as a Strict Nature Reserve (Cagar Alam) from 1977. The objectives of the park are to protect and conserve the flora and fauna within its boundaries.

Historically, by the Ministry of Agriculture No. 40/Kpts/Um/1/1979, it was stated that about 40,000 ha of the Halimun area was declared as a nature reserve for conservation – this was then transformed into the Halimun National Park in 1992 and the remaining forest areas (73,357 ha) were declared as production and protected forests. However, in an effort to reduce forest loss, the Indonesian government ratified a new conservation area by merging Halimun National Park and Salak Reservation area in 2003, including the production forests. Thus, under the Ministry of Forestry decree No 175/Kpts-II/2003 on June 10, 2003 those merged areas officially became GHSNP, with a total size of 113,357 hectares (Team, 2008), making it the national park that has the largest tropical rainforest ecosystem on the island of Java. This large national park area does not merely encompass the forest but also villages, tea plantations, agriculture and grassland, reflecting the land use history (Rinaldi et al., 2008). Thus, those areas are categorized as enclaves since they are not part of GHSNP, yet they are situated surrounding. GHSNP

2.4.3 Biotic Condition

2.4.3.1 Flora

GHSNP is believed to have the highest biodiversity for both plants and animals in Java (Team, 2008). Forest ecosystems in GHSNP can be classified into three zones based on its altitude, namely the Collin zone, the submontane and the montane zone (JICA, 2007). Collin zone is the area which lies at an altitude of 500-1000 metres above sea level. It is dominated by the plant types of *Schima wallichii* (Puspa), *Quercus sundaicus* (Pasang), *Altingia excels* (Rasamala), and *Castanopsis acuminatissima* (Saninten); the submontane zone is defined as the area that is situated at an altitude of 1000–1500 metres above sea level and is predominantly occupied by the species of *Elaeocaldrus ganitrus* (Ganitri), *Saurauia pendula* (Kileho), and *Weinmania blumei* (Kimerak). The montane zone is an area in the range between 1500–2211 metres above sea level. This zone is dominated by *Dacrycarpus imbricatus* (Jamuju), *Podocarpus blumei* (Kibima), *Podocarpus neriifolius (*Kiputri*)*, and *Vernonia arborea* (Hamirung). It is also recorded that there are 258 species of orchids, 12 species of bamboo, 13 species of rattan, ornamental and medicinal plants such as *Nepenthes* sp (Kantung Semar) and *Dipterocaldrus hasseltii* (Palahlar), which are unique and rare plants found only in GHSNP (JICA, 2007).

2.4.3.2 Fauna

GHSNP has a wide range of habitat of as well as endangered species. Primate mammals found in this park include *Hylobates moloch* (the Javan gibbon), *Presbytis comate* (Surili), *Trachypithecus auratus* (langur), and *Macaca fascicularis* (the long-tailed macaque). Some species of deer such as, Indian muntjac; *Tragulus javanicus* (Java mouse-deer); *Sus scrofa* (wild boar); carnivorous animals, such as *Panthera pardus melas* (the *Javan leopard*) and *Prionailurus bengalensis* (the wildcat) also exists in this park (JICA, 2007).

In addition, this area also supports many species of unique insects and butterflies, beetles, and birds. Up to 244 species of birds have been recorded in this region, 32 of which are endemic to the island of Java, such as *Spizaetus bartelsi* (the Javan hawk-eagle), *Cochoa azurea*, *Otus angelinae, Haldractes reinwardtii*, and *Bucheros rhinoceros* (the rhinoceros hornbill) that is officially declared as an endangered species (JICA, 2007).

2.4.4 Demographic and Socio-Economic Conditions

There are 314 sub-villages (or settlements) located within the national park area, and around 100,000 people rely on the natural resources in the park for their daily livelihoods (Kubo & Supriyanto, 2010). Some parts of GHSNP area and its surrounding are also home to several indigenous groups of people who occupy Kampung Urug, Citorek Bayah, Ciptamulya, Cicarucub, Cisungsang, Sirnaresmi, Ciptagelar and Cisitu (JICA, 2007).

Widada (2004) stated that the value of economic benefits of GHSNP, based on the analysis of the total economic value (NET), reached NZ$ 47.28 million per year, consisting of a carbon sink value of NZ$ 46.21 million (97.73%). If the value of carbon sinks is not taken into account, then the value of GHSNP is NZ$ 1.02 million, with the economic value of water (domestic and agricultural) showing the highest proportion of 66.58% (Ilyas, 2014).

Social conflicts related to land ownership, intensive land use, and ongoing timber exploitation by the rural community are major problems for the management of this national park (Rosleine, Suzuki, Sundawiati, Septiana, & Ekawati, 2014). Concerning deforestation, approximately 1,473 ha or 1.3% of the total GHSNP area is deforested each year (Halimun-Salak, 2007). Worse still, these degraded areas are mainly located in the corridor between the Mount Halimun and Salak areas (Rosleine et al., 2014) where this is thought to enable movement of animals between connected patches of habitat (Tewksbury et al., 2002). The capability of GHSNP to rehabilitate the degraded forest is only 500 to 800 ha annually because of a limited budget and lack of human resources. One of the causes of such deforestation is the use of forest products by villagers. This finding is in a study conducted by Hani and Rachman (2016) in Cimapag Kampong (a village close to GHSNP). Based on their study, it was found that more than 85% of the villagers collect firewood from the national park area to fulfil their needs and 33% of them cut timber wood to build their houses and for other purposes.

2.4.5 Physical Characteristics

2.4.5.1 Topography

The topography of this region is normally hilly and mountainous with the highest peak being Mount Salak 1 at 2,211 metres above sea level (all elevations are metres above sea level unless otherwise stated), situated in the southeast of the national park. Some of the highest mountains located in the western part of the park are Mount Halimun Utara (1929 m), Mount Sanggabuana (1,920 m), and Mount Botol (1850 m); while the mountain situated in the north-east of GHSNP is Mount Kendeng Utara (1,377 m) (Halimun-Salak, 2007).

The slope of the land ranges from 1% to 32% with the dominant elevation ranges from 500 to 2,211 m. Information regarding the slope of GHSNP is presented in Table 2.1 and its spatial distribution is shown in Figure 2.3

Table 2.1 Class of slope within the area of Gunung Halimun Salak National Park

Number	Class of Slope (%)	Area (ha)	Percentage (%)
1	1-2	27,04	25.71
2	> 2-4	37,39	35.55
3	> 4-6	26,06	24.78
4	> 6-8	12,23	11.63
5	8-32	2,431	2.311
	total	105,17	100

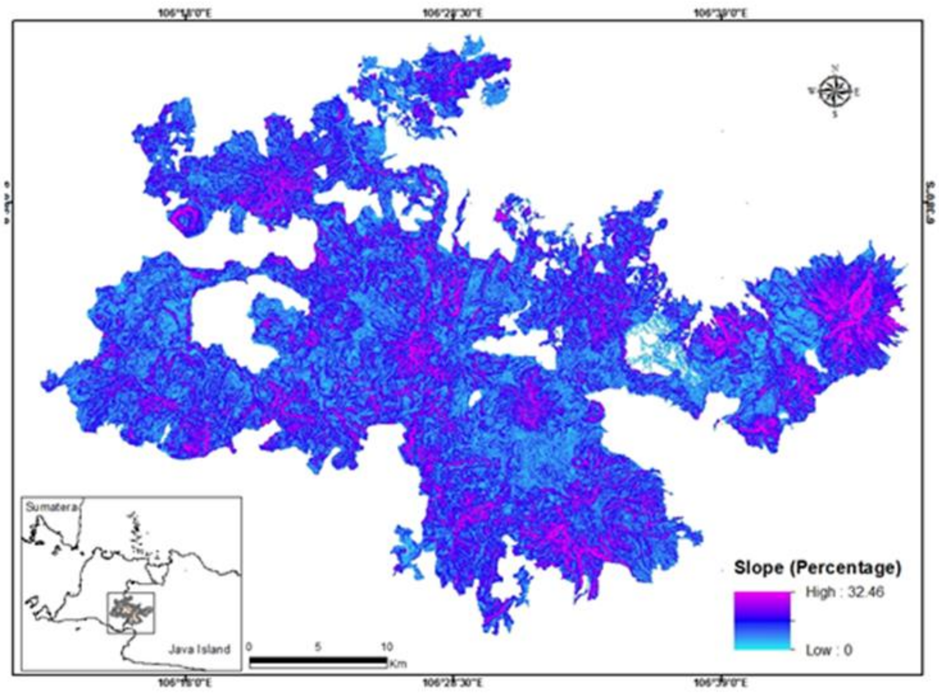


Figure 2.3 Slope maps of Gunung Halimun Salak National Park (P. P. d. P. I. Geospasial, 2014)

Based on the map of the slope above, it appears that the dominant slope in GHSNP is in the range of > 2-4% or 35.5% of the total area of GHSNP. The second dominant slope is in the range of > 4-6% or 24.8% of the total area.

Table 2.2 shows that in general, 49.91% of the total area of GHSNP consists of elevations ranging from 500 to 1000 m; 40.50% of the area has an elevation from 1000 to 2000 m, and 8.84% of the area has an elevations from 250 to 500 m. The spatial distribution of elevation in GHSNP is presented in Figure 2.4.

Table 2.2 Class of elevation within the area of Gunung Halimun Salak National Park

Number	Class of Elevation (m)	Area (ha)	Percentage (%)
1	0 - 250	706	0.62
2	> 250 - 500	9994	8.84
3	> 500 -1000	56369	49.91
4	> 1000 -2000	45743	40.50
5	> 2000	120	0.10
	total	112812	100

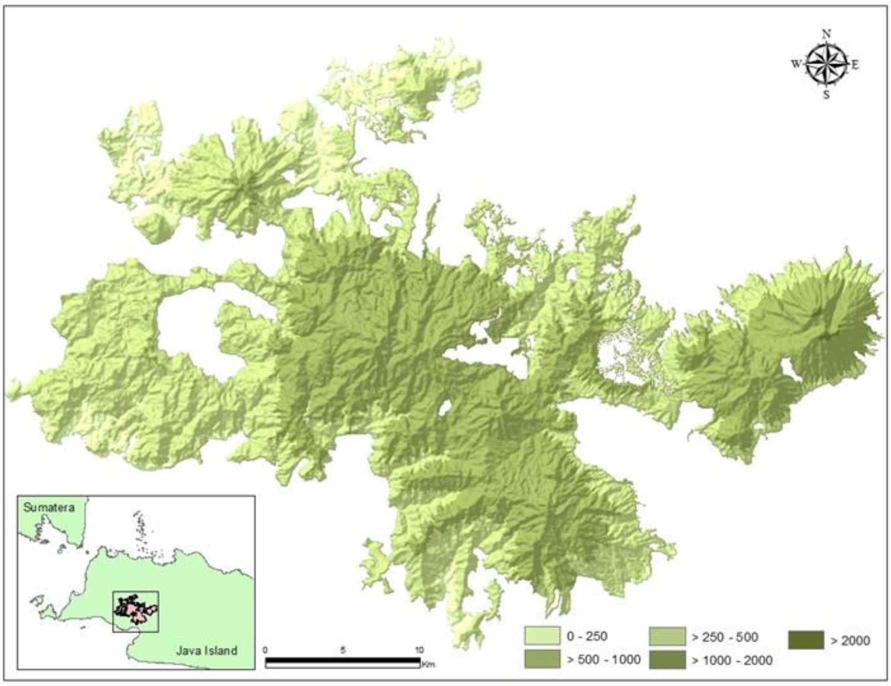


Figure 2.4 Elevation maps of Gunung Halimun Salak National Park (P. P. d. P. I. Geospasial, 2014)

2.4.5.2 Types of Soil

The type of soil in the GHSNP area consists of a combination of Andosol Brown and Regosol Brown; a mix of Latosol Brown Yellowish; a combination of Reddish Brown Latosol with Latosol Brown; a combination of Latosol Red, Reddish Brown latosol and Laterite groundwater; a mix of latosol Redness and Latosol; and associations of Brown latosol and Regosol Gray (JICA, 2007).

2.4.5.3 Climate

The climate in the GHSNP area and its surroundings is classified as a Type B climate with a Q value of 24.7%. The average rainfall is 4000–6000 mm/year and a rainy season occurs during the months of October to April. The dry season takes place from May to September with rainfall of about 200 mm/month (JICA, 2007). The number of days of rain each year is approximately 203. The average daily temperature is between 20°C to 30°C, with average air humidity around 80%. Wind conditions are affected by monsoon winds that change the wind direction based on the season. Throughout the dry season, the wind is typically blowing from the north-east at low speed with an average air humidity of 80% (Halimun-Salak, 2007).

Chapter 3

Literature Review

3.1 Introduction to Landscape Ecology

Landscape ecology is the study of landscape patterns, the influences of human and natural environmental relations on a landscape mosaic, and changes in the landscape pattern and environmental processes over time (McGarigal & Marks, 1995). Landscape ecology considers vegetation as a mosaic of patches with unique landforms, species compositions and disturbance gradients (Ravan, Roy, & Sharma, 1998). Thus, various definitions of the term landscape have evolved from landscape ecology research. As an example, Diaz and Apostol (1992) defined Landscapes as "...aggregates of homogeneous patches of vegetation or land use types and landforms that come into being through climatic influences, geomorphic processes, natural disturbances, human activities, and plant succession."

Similarly, Forman (1995) defined landscape as the mosaic repetition of local ecosystems or land uses that is identical in terms of forms over the extensive area. Some of the examples of landscape include, suburban, forested, cultivated, and dry landscapes. Moreover, landscapes, according to Ndubisi (2002), are the combination of natural and cultural features that include, hills, fields, and forests, throughout the land.

Forman (1995) stated that the mosaic pattern of landscapes consists of three spatial elements: patches, corridors, and matrices. Patches are areas that differ from the surrounding context (Forman, 1995) and are heterogeneous when compared to the entire area (T. G. Barnes, 2000). The form of patches varies from large to small, elongated to round, and convoluted to smooth. A corridor is a kind of a patch though they connect patches to other areas. They can be wide to narrow, meandering to straight, and have high to low connectivity (Dramstad, Olson, & Forman, 1996; Forman, 1995). Corridors have been shown to be beneficial for wildlife to facilitate movement and survival habitat in many situations, such as forest, urban, or agricultural landscapes (MacDonald, 2003), in spite of the fact that the need for corridors of individual species is varied. A number of studies have suggested that corridors are beneficial to the conservation of wildlife and vegetation. These include studies of arboreal marsupials in Queensland (S. G. Laurance & Laurance, 1999), small mammals and frogs in Amazonia (De Lima & Gascon, 1999), butterflies in North America (Haddad & Baum, 1999), and carabid beetles in Scotland (Petit & Usher, 1998).

A matrix is the "background ecological system" of a landscape. It is the most extensive and connected landscape type, and plays the foremost role in landscape functioning (Forman, 1995). Thus, it is the dominant component in the landscape (T. G. Barnes, 2000). Forest, grassland, rice fields, or another land cover often forms a background matrix, while the individual trees, shrubs, rice plants, and small buildings are aggregated to create the pattern of patches, corridors, and matrix on land (Forman, 2014). Figure 3.1 shows landscape composition as the union of patches, corridors, and matrices that exist in a landscape (Betts, 2000; Forman, 1995).

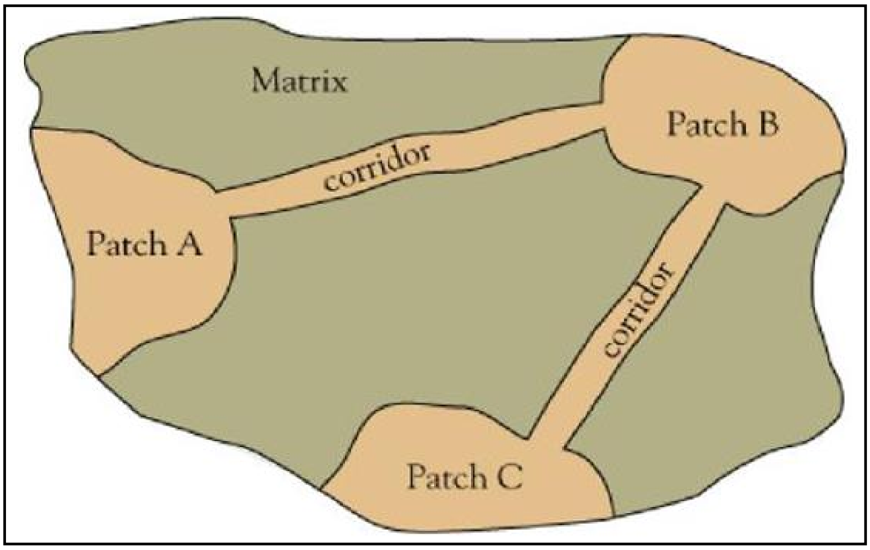


Figure 3.1 Landscapes consist of patches, corridors, and matrices (T. G. Barnes, 2000)

According to Brown, Hickey, Harrington, and Gill (2001) landscape elements can be classified as either tangible or intangible. Tangible landscape elements include transportation corridors and junctions, utilities, and land cover. Intangible landscape elements include political boundaries, eco-regional boundaries, ownership boundaries, and land use. For this study, the tangible elements of landscapes taken into consideration were areas, roads, and land cover while intangible elements in this study include the eco-regional boundary and the regional boundary.

Tangible elements change the physical character of the landscape and often have a direct impact on ecosystems (Brown et al., 2001). For instance, a road network established in undisturbed forest decreases the quality of habitat for large mammals. Similarly, the installation of utility networks, such as pipelines, requires the clearing of vegetation which can result in fragmentation (Brown et al., 2001). Forests as a landscape, for example, could also be fragmented. The fragmentation of the forest could be described as the breaking up of a forest unit, where the number of patches and the amount of expose edge increase while the core area decreases (Meddens, Hudak, Evans, Gould, & González, 2008). Core area in this case is defined as a forest area that is free from edge effects (Moreno-Sanchez et al., 2012).

3.2 Land Cover

Land use is the term that is used to describe human uses of land, or actions modifying or changing land cover and the term land cover originally referred to the kind and state of vegetation, such as forest or grass cover, but it has broadened in subsequent usage to include other things such as human structures, soil types, biodiversity, surface and ground water (Meyer, 1995).

Land cover can be altered by forces other than anthropogenic ones. Natural phenomena such as flooding, weather, fire, climate change, and ecosystem dynamics may initiate modification upon land cover. Globally, land cover today is transformed mainly by direct human use, by agriculture and livestock increases, forest harvesting and management, and urban and suburban construction and development (Meyer, 1995). The driving forces behind this human activity could be economic, technological, demographic, scenic and or other factors. A good illustration regarding the causes and consequences of the changes in land use and land cover (see Figure 3.2) is best explained by Reid et al. (2000).

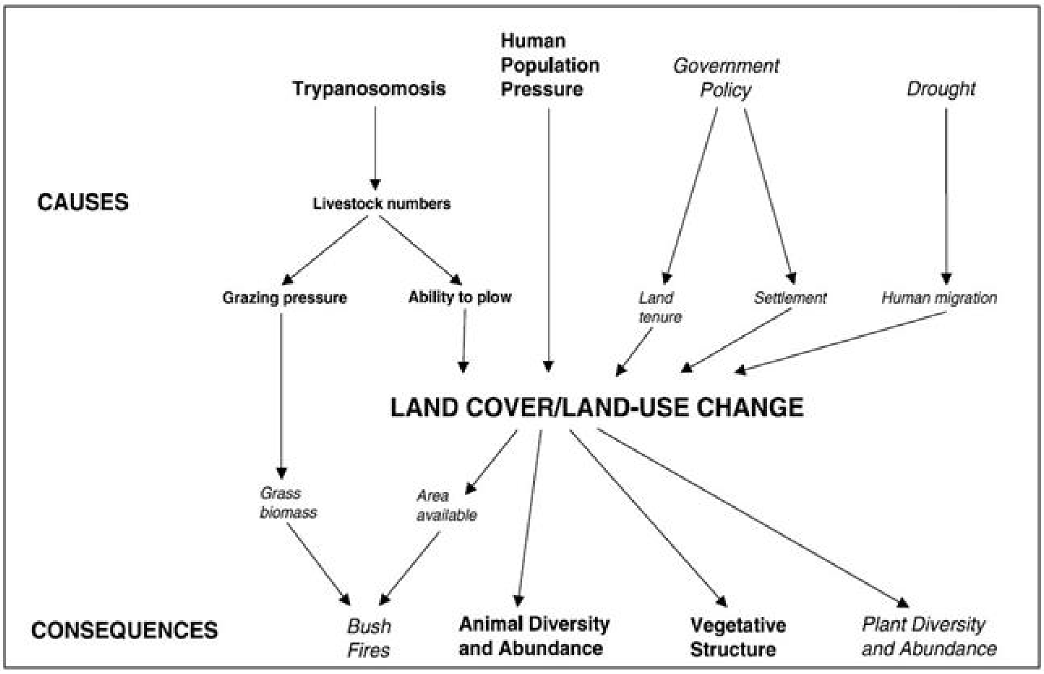


Figure 3.2 Hypotheses (in bold) and final (in bold and italics) concerning the causes and ecological consequences of land use/land cover change (Reid et al., 2000)

Based on their study, it was found that rapid land use or land cover change was due to the combined effects of drought and migration, alteration in settlement and land tenure policy, and changes in the severity of livestock disease. The scale of the sources and consequences of land use or land cover change were wide ranging from local to regional to international; and the correlation between causes and consequences across scales. At the landscape scale, each cause affected the location and pattern of land use or land cover differently.

In summary, land use and land cover dynamics are a result of complicated interactions between several biophysical and socio-economic conditions that may ensue at a wide range of temporal and spatial scales (Reid et al., 2000). Land use and land cover changes are so prevalent that, when accumulated globally, they significantly affect vital aspect of Earth system's functioning. They directly influence biodiversity around the globe (Sala et al., 2000), and contribute to local regional climate change (Dale, 1997).

3.3 Habitat Fragmentation and Disturbance

Landscapes change over time. The causes of those changes can include fires, flooding, windstorms, earthquakes, volcanic eruptions, tsunami, firestorms, climate change and anthropogenic activities, such as forest clearing (Dale et al., 2001). This forest exploitation action is just one clear example of how a landscape changes from one form to another.

Generally, there are five major spatial processes (see Figure 3.3) resulting from transformation of land from one form to another (Forman, 1995). These processes include:

- perforation: a process in which hoIes are created in a habitat or land type by disturbance features
- dissection: the subdivision of an area by equal-width line, such as roads or utility corridors
- shrinkage: the decrease in size of landscape patches, and
- fragmentation: the breaking up of a land cover type into pieces that are commonly and unevenly distributed across space.
- attrition: the disappearance of landscape patches.

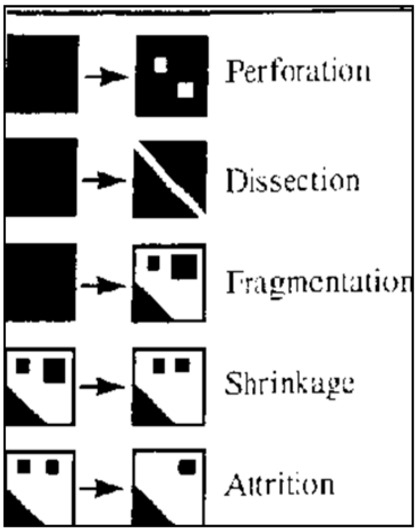


Figure 3.3 Spatial process as a result of land transformation which includes perforation, dissection, fragmentation, shrinkage, and attrition (Forman, 1995).

Concerning fragmentation, Forman (1995) argues that habitat fragmentation is one stage within this broader sequence of landscape processes that transform the natural land cover type of a region as a result of human or natural disturbance events. The five processes mentioned above overlap in both sequential order and importance, with perforation and dissection peaking in importance in the early phases of Landscape change. Fragmentation and shrinkage are typical of the middle phases and attrition predominates at the end of the landscape transition cycle.

Generally, a fragmentation process is characterized by a decrease in the number of habitats (habitat loss) and an alteration in the spatial features and configuration of the remaining patches (Forman, 1995; Saunders, Hobbs, & Margules, 1991). Besides a reduction in habitat amount, fragmentation has an implication for habitat patterns as well and lead to: (a) an increase in the number of habitat patches, (b) a decrease in sizes of habitat patches, and (c) an increase in isolation of patches. These effects are the basis of most quantitative measures of habitat fragmentation. However, fragmentation measures vary widely; some include only one effect (e.g., reduced habitat amount or reduced patch sizes), whereas others include two or three effects but not all four (Fahrig, 2003).

Over time, human activities are the primary reason for fragmentation (Bogaert et al., 2005). It occurs when large areas of natural landscapes are intersected by human activities, such as the development of settlements, conversion of forested land into agricultural land use, and development of infrastructure, such as transportation, pipelines and so on (Forman, 1995; Llausàs & Nogué, 2012; Slonecker et al., 2012). Roads and railways, for example, change hydrological patterns due to their physical structure, with subsequent effects on erosion and sedimentation levels and may act as blockades to animal movement or as dispersal conduits (Forman, 1995; Gelbard & Belnap, 2003). Utilization of transport infrastructure further indicates disturbance such as traffic noise and artificial light, increased animal mortality, introduction of exotic species and chemical contamination. These impacts extend outwards from the road corridor producing effect zones within which environmental conditions are altered (Forman & Alexander, 1998; Trombulak & Frissell, 2000).

Substantial amounts of experimental research support the existence of these effect zones (Benítez-López, Alkemade, & Verweij, 2010; Biglin & Dupigny-Giroux, 2006; Eigenbrod et al., 2009; Forman & Deblinger, 2000) and studies have assessed that 15% to 20% of the USA (Forman & Alexander, 1998) and 16% of the Netherlands (Reijnen & Foppen, 1995) are within effect zones. The “road effect zone”, first coined by Forman and Alexander (1998), is the distance perpendicular to any point at a road, within which environmental changes can be significantly distinguished from a control location (see Figure 3.4).

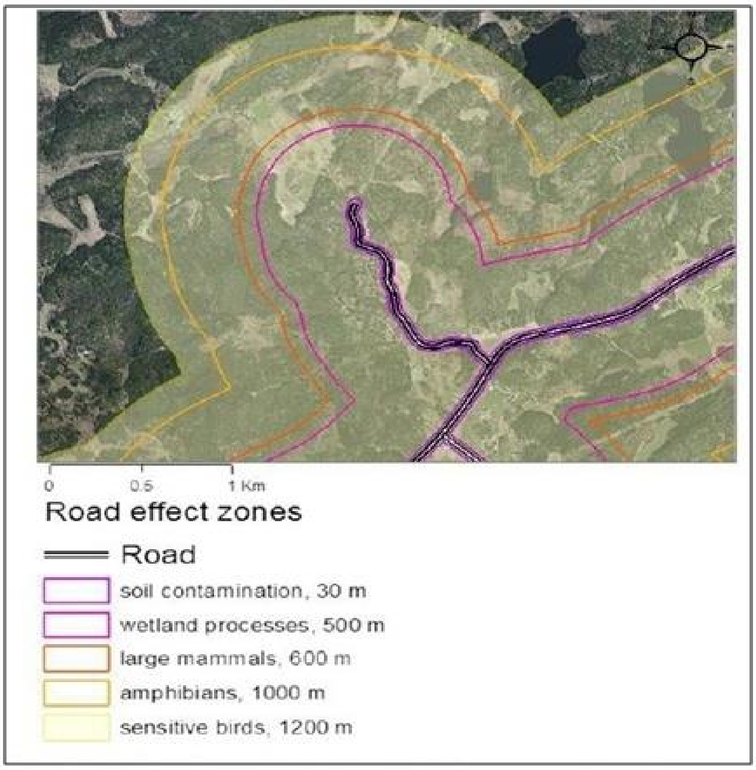


Figure 3.4 Illustration of road effect zones based on empirical studies (Karlson, 2015)

The effects of fragmentation are likely due to two main causes. First, fragmentation per se indicates a larger number of smaller patches. At some point, each patch of habitat will be too small to support a local population or perhaps even an individual territory. Species that are unable to cross the non-habitat portion of the landscape (the "matrix") will be restricted to many small patches, ultimately reducing the overall population size and probability of persistence. The second main cause of fragmentation impact is edge effects. According to Fahrig (2003) more fragmented landscapes contain more edge for a particular of habitat. This can escalate the probability of individuals leaving the habitat and entering the matrix. Moreover, the time spent in the matrix will be longer in a more fragmented landscape, which may increase overall mortality rate and subdue overall reproductive rate of the population (Fahrig, 2002). In addition, negative edge effects might appear due to species interactions. For instance, some comprehensive studies have pointed out the increased predation on forest birds at forest edges (Chalfoun, Ratnaswamy, & Thompson, 2002)

Overall, it is widely thought that fragmentation could have extremely damaging impacts on biodiversity. Some studies have shown that the abundance, diversity and breeding activities of bird species in the forest and grassland habitat are significantly diminished in the areas up to 300–1000 m from roads (Forman, Reineking, & Hersperger, 2002; Reijnen, Foppen, & Meeuwsen, 1996; Rheindt, 2003; Seiler & Helldin, 2006).These authors argue that this may be caused by disturbance regimes like traffic noise and light pollution. Other authors noted road mortality as the most likely overall cause for depressed densities of mammal and bird populations in the same distances (Eigenbrod et al., 2009; Fahrig & Rytwinski, 2009; Summers, Cunnington, & Fahrig, 2011).

Wilcox and Murphy (1985), for example, have claimed that forest fragmentation is the most serious threat to biological diversity and the foremost cause of the present extinction crisis. A major consequence of forest fragmentation is loss of connectivity between residual forest patches. Although the value of travel corridors for wildlife populations has been debated (Beier & Noss, 1998) and the value of corridors differs among species (Haas, 1995; Hannon & Schmiegelow, 2002), connectivity of habitat patches is considered to be highly important to maintain. A study conducted by Sieving, Willson, and De Santo (2000) clearly illustrated how a distribution of bird populations and metapopulation structure are affected by habitat connectivity. Opdam (1991) stated that in metapopulation theory, the survival of a spatial network of subpopulations of a species relies on a dynamic equilibrium between local, coincidental disappearances (extinction), and re-appearances (colonisation). Once populations are divided into smaller subpopulations, the chances of extinction of these subpopulations increases (Forman & Alexander, 1998). This is due to the higher susceptibility of smaller populations to ‘normal’ variations (Jaarsma & Willems, 2002).

3.4 Fragmentation Quantification

A quantification and comparison of landscape indices have been acknowledged as the most effective way to assess forest fragmentation at the landscape and patch levels of analysis (Ripple, Bradshaw, & Spies, 1991; Singh, 1989; Southworth, Nagendra, & Tucker, 2002; M. A. Wulder et al., 2008). Due to the complex nature of the fragmentation process, it has been suggested that a single landscape metric is insufficient to capture all aspects of fragmentation (Davidson, 1998; Ripple et al., 1991).

The set of metrics to quantify habitat fragmentation consists of a rationale for each spatial metric, along with a brief description (Table 3.1). Based on the literature reviewed, each of these metrics appear to capture the essential changes that are assumed to have occurred in the past few decades because of habitat fragmentation. The use of a well-documented set of metrics also allows the results of this research to be compared with similar investigations on the quantification of forest fragmentation in other regions.

Table 3.1 Landscape metrics that are generally used to quantify habitat fragmentation

No	Spatial Metrics	Description/Responses to Fragmentation
1	Mean patch size (ha)	Average size of patches is expected to decrease with increasing fragmentation (McGarigal & Marks, 1995)
2	Patch density	Number of class patches relative to total landscape area. As the number of patches increases on the landscape, the density of suitable patches is expected to be positively correlated and also increase with habitat fragmentation (McGarigal & Marks, 1995)
3	Number of patches	Total number of patches within each individual class To measure patch abundance and determine if suitable patches are becoming more or less numerous over time (McGarigal & Marks, 1995)
4	Class area (ha)	Total area of all patches per class. The total amount of suitable habitat is expected to decrease as a result of fragmentation due to timber harvest and wildfires (McGarigal & Marks, 1995)
5	Patch size coefficient of variation	The average relative variability. It is about the mean for each class. It is generally preferable to Patch Size Standard Deviation for comparing patch size variability between landscapes. Patch size coefficient of variation is expected to decrease as patches become less variable or more similar in patch size over time (McGarigal & Marks, 1995)

6	Mean shape index	Average shape index of all class patches. Patch shape equals the perimeter divided by the square root of the patch area, adjusted to a square standard. Patches are expected to become less geometrically complex in a managed Landscape (McGarigal & Marks, 1995)
7	Edge density	Amount of edge relative to the landscape area is expected to increase in the initial stages of habitat fragmentation (McGarigal & Marks, 1995)
8	Mean proximity index	Average proximity index for all patches in a class. Proximity index is calculated as the sum of the ratio of patch size to nearest neighbour edge-to-edge distance for all patches within a specified search radius. It measures the isolation of patches in a landscape and is expected to decrease over time as there are fewer patches of the same class in close proximity to each patch type (Gustafson & Parker, 1992)
9	Mean core area	Average size of disjunct core patches or interior patch areas remaining after specifying an edge effect buffer area. One of the main effects of forest fragmentation is the conversion of interior habitat to edge habitat (Tinker et al. 1998). it is expected that the amount of interior habitat will decrease as a result of fragmentation (McGarigal & Marks, 1995)

Several studies analysing the fragmentation of forested habitat have been conducted by applying these metrics. For example, Luque, Lathrop, and Bognar (1994) documented the extent of landscape fragmentation in the New Jersey Pine Barrens region by showing significant changes in selected spatial metrics between a forested landscape in 1972 and the 1988 landscape. As a result of human disturbances, such as suburban and exurban development and logging activities over the 16 year period, a range of landscape metrics including fractal dimension, diversity, and contagion decreased while dominance, disturbance and edge indices increased at the landscape level. At the patch level of analysis, the mean size of forested patches decreased significantly. These results emphasised a trend to a more dissected or fragmented landscape over time because of human disturbances.

Another study related to fragmentation was conducted by Hargis, Bissonette, and Turner (1999) who calculated a suite of landscape metrics to determine the effects of forest fragmentation on American marten *(Martes americana*) population counts in the Uinta Mountains of northern Utah. The results indicated that martens were prone to changes in landscape pattern resulting from natural openings and timber harvesting, and that a reduction in forest interior area may endanger the survival of future populations.

3.5 Impact Assessment Quantification

The assessment of the magnitude of impacts is often undertaken by the application of simulation models (Fedra, 1993). The result is often presented in the form of a map showing the value of a given environmental descriptor (e.g., concentration of an air pollutant) at any location within the study area. The extension of environmental impacts can therefore be estimated from the spatial distribution of environmental quality values predicted for each alternative (Antunes, Santos, & Jordao, 2001).

In general, selecting which models to use depends on the aim and scope of the study and the context in which the results will be used. Further issues that need to be considered are, for instance, what biodiversity components are to be modelled, availability and quality of data and expert knowledge, time frame, available resources, and competence of those carrying out the analyses. In addition, the limitations and constraints of the different types of ecological models must be taken into consideration (Gontier, 2006). Geneletti (2004), for instance, proposed a modelling method based on ecosystem rarity to introduce criteria for protection and preservation of nature, and he applied it to compare the impact of different alternatives for a road project. Another modelling-based land cover assessment is provided by Treweek and Veitch (1996) who looked at the spatial distribution of land-cover categories and their proximity to existing or planned developments. Such modelling approaches can be supported by Geographical Information Systems (GIS).

In addition, to understand the impact of fragmentation effects, a good understanding of the biology and habitat use of species is also required (Wiegand, Revilla, & Moloney, 2005). Hence, the biggest issue is how to integrate the habitat network requirements of an array of species— which greatly differ in their response because of different spatial requirements and different movement capacities —with landscape pattern and change (Lord & Norton, 1990; Opdam, 1991; Opdam, van Apeldoorn, Schotman, & Kalkhoven, 1993; Rosenberg, Noon, & Meslow, 1997b).

In order to predict and assess the consequences of fragmentation caused by urbanisation and infrastructure, biodiversity needs to be quantified, and this requires biodiversity indicators that are sensitive to fragmentation processes (Lambeck, 1997; Noss, 1990). Useful indicators can be the habitat networks of focal species that are related to a certain habitat type, have large area requirements, or have low dispersal ability (Hansson, 2001; Vos et al., 2001). Vos et al. (2001) proposed ecological profiles and ecologically-scaled landscape indices (ESLI), which are both useful tools for integrated fragmentation assessment and spatial conditions assessment of the landscape for sustainable conservation of biodiversity. As the concept of ecological profiles aims to model the behaviour of a larger system through the behaviour of prioritised biodiversity components, it is imperative to ensure that the formulation of species requirements and attributes is realistic and relevant for the specific context (Karlson, 2015). The resulting matrix of ecological profiles, classified according to individual area requirements and dispersal distance, encompasses relevant elements of this specific-scale dependent fragmentation sensitivity.

3.6 The use of Geographic Information System (GIS) and Remote Sensing

Recently there has been a revolution in the availability of information and in the development and application of tools for managing information and undertaking ecological assessment using Geographic Information Systems (GIS). These have enabled users, including ecologists, to organize information gathered across broad geographic regions in a spatial database and to perform analyses at a scale that was previously problematic to achieve (Miller & Wu, 2000). This tool has proved to be effective in accommodating large varieties of spatial attribute data. Additionally, it is beneficial to handle many layers of map information relating to one area. Generally, each layer describes a different aspect of its geography. One layer might hold data on geology, another on soils. Subsequent layers might include data on land cover, in a particular area, species distributions, or the socio-economic characteristics of the human population in the area. Thus, the power of GIS lies in the fact that data from any combination of these layers might be used to solve a particular problem. Furthermore, as problems change, the data can be processed in various ways to address different issues in a highly flexible way. Moreover, not only the ability to handle spatial information in the form of maps is important, GIS can also hold nonspatial attribute information, which can be associated with the various map features in a database management system of some kind (Miller & Wu, 2000).

As an example, the information embedded in a GIS could be used to target surveys and monitoring schemes. Data on species and habitat distribution from different dates allow monitoring of the location and the extent of change (Powell, Accad, & Shapcott, 2005; Salem, 2003). Another example of how GIS is used in the study conducted by Green and Baker (2003) about urbanisation impacts on avian habitat. In their research, they used a GIS to measure habitat structure and species composition at each sample point, and a variety of landscape –habitat metrics were measured from aerial photographs and their location recorded (Miller & Wu, 2000).

Increasingly, ecologists are also using new technologies to collect field data. Remote sensing from aircraft and satellites has allowed them to collect data at various scales that can include many interacting ecosystems and even whole biomes. Remote sensing is the measurement of reflected, emitted, or back-scattered electromagnetic radiation from the earth's surface, using instruments placed at a distance, most often on a satellite, and occasionally aircraft are also used (Southwood & Henderson, 2000). Hence, the multispectral data provided by such on-board sensors led to an improved understanding of crops, forests, soils, urban growth, land changes and many other earth features and processes (Peres & Terborgh, 1995).

The integration of GIS and remotely sensed image analysis have indicated that a combination of data sources and techniques may provide more information about environmental change. A review by Wilkinson (1996) identifies three ways in which remote sensing and GIS technologies are complementary: (a) remote sensing techniques can be used to acquire GIS data sets; (b) GIS data can be incorporated as additional information to improve remote sensing products; and (c) remote sensing data and GIS data can be used in conjunction for environmental modelling and impact analysis. A diagram as shown in Figure 3.5 shows how GIS plays an important role in land cover and impact assessment.

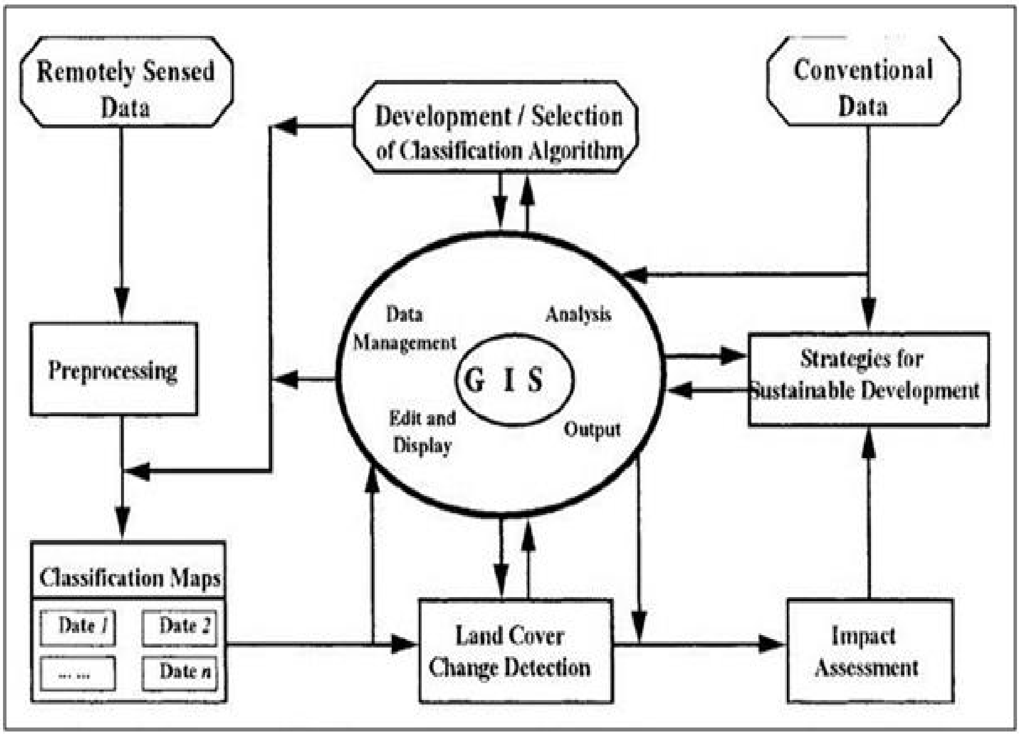


Figure 3.5 Diagram of land cover change analysis based on remote sensing and GIS (Xiuwan, 2002)

In conjunction with spatial metrics applied in fragmentation assessment, many GIS and image analysis software packages (e.g. IDRISI or GRASS) include landscape metric algorithms or other landscape metric software. Packages such as FRAGSTATS (McGarigal & Marks, 1995) and Patch Analyst (Elkie, Rempel, & Carr, 1999) have been developed to operate in conjunction with a GIS. FRAGSTATS is a computer software program designed to compute a wide variety of landscape metrics for categorical map patterns. It is a stand-alone program written in Microsoft Visual C++ for use in the Windows operating system. Alternatively, Patch Analyst works as an extension within ArcGIS and contains a more recent update of software code relative to FRAGSTATS. Either of these two programs can compute various indices of hierarchy.

The integration of GIS, remote sensing, and landscape metric software provides the opportunity for advanced analysis of landscape changes and disturbances. As an example, Tinker et al. (1998) used FRAGSTATS to calculate spatial pattern metrics to compare the effects of clear cutting and road building on the landscape pattern of the Bighorn National Forest, Wyoming. A principal components analysis was conducted to group the FRAGSTATS metrics into three uncorrelated components. Based on the results, the authors suggested the effects of timber harvesting and road construction may be easily monitored by the analysis of a few key metrics: patch core area, patch size and number of patches, edge density, and patch shape. Each of these metrics reported the highest loadings or scores across the three different principal components. Measures of inter-patch distance, such as mean nearest neighbour distance (MNN) and the mean proximity index (MPI), were also included in the Tinker et al. (1998) study. Results showed that either the MNN or MPI or inter-patch distance measures could contain unique information about landscape structure based on low correlations with the other metrics.

3.7 Summary

Forest ecosystems have long been known to have global conservation importance due to their vital economic, social and environmental benefits. However, these ecosystems are being rapidly changed or degraded in many areas of the world. Such forest changes are linked to forest fragmentation, which is generally defined as the process of splitting up a continuous habitat type into smaller patches, resulting in various impacts on ecological processes.

Fragmentation ensues when huge areas of natural landscapes are intersected by human activities, such as development of human settlements, conversion of forested land into agricultural area, and the development of infrastructure, for instance, roads and pipelines.

Roads accelerate fragmentation by dissecting previously large patches into smaller ones, and then produce edge habitat in patches along both sides of the road, potentially at the expense of interior habitat. Moreover, the use of transport infrastructure shows disturbance effects such as traffic noise and artificial light, increased animal mortality, exotic species invasion and air pollution. These impacts extend outwards from the road corridor generating effect zones, within which environmental conditions are changed.

One of the most alarming aspects of forest loss and fragmentation is the unparalleled threat to biodiversity that can escalate the risk of species extinction. Thus, exploring the effects of forest loss and fragmentation on ecological processes and function at various level (local, regional, national and global) is of primary concern for sustainability when managing forests around the world. In addition, biodiversity also needs to be quantified in relation to habitat fragmentation and this assessment requires biodiversity indicators that are sensitive to fragmentation processes. Useful indicators can be habitat networks of focal species that are related to specific habitat type, and have large area requirements, or have low dispersal ability.

Increasingly, ecologists are using new technologies to collect field data. Remote sensing from aircraft and satellites has allowed them to compile data at various scales, which can include many interrelating ecosystems and even whole biomes. With the integration of GIS and remotely sensed image analysis, it seems that a combination of data sources and techniques for spatial data analysis may provide more information about environmental change as well as landscape changes and disturbances. In regards to this, many GIS and image analysis software packages (e.g. IDRISI, GRASS), including landscape metric algorithms or other landscape metric software packages such as FRAGSTATS (McGarigal & Marks, 1995) and Patch Analyst (PC Elkie, RS Rempel, & AP Carr, 1999), have been developed to operate in conjunction with a GIS.

Chapter 4
Methodology

This chapter describes the data and methodology used in this study. Section 4.1 presents a brief background of the study area. Section 4.2 discusses the research design which was comprised of three phase: (1) analysing the changes in land cover from 2001 to 2016 (2) analysing forest fragmentation for a fifteen year periods (3) analysing fragmentation and disturbance effects on mammals and birds. Section 4.3 presents materials and data collection methods, and Section 4.4 describes data processing and analysis used to answer research questions.

4.1 Study Area

GHSNP was chosen as a study area since this conservation area is the largest and the remaining tropical mountain forest in West Java-Banten, Indonesia (Galudra, 2005). Figure 4.1 shows the distribution of conservation areas in Indonesia, including GHSNP.

Figure 4.1 Gunung Halimun Salak National Park as one of the conservation areas in Indonesia (Watch, 2010)

4.2 Research Design

This research was conducted in three phases. Figure 4.2 shows the general flow of this study. The first phase was to analyse the changes in land cover between 2001 and 2016. This consisted of some initial steps namely, the interpretation of Landsat imagery, the classification of maps of land cover, ground truthing of the research site, land cover accuracy assessment and overlaying the maps of land cover in 2001 and 2016. This was conducted Using ArcGIS ver. 10.3 and ERDAS, a GIS software. Then, in the second phase, forest cover maps for 2001 and 2016 obtained from the first phase was used to analyse forest fragmentation by quantifying landscape metrics of GHSNP forest cover in 2001 and 2016. The final phase was to analyse the fragmentation and disturbance effects on mammals and birds. This was conducted in several stages, namely (1) modelling the spatial distribution of road effects, (2) the construction of ecological profiles, (3) construction habitat network for each ecological profiles, (4) construction of fragmented and disturbed habitat network, and (5) quantification of the impact on habitat networks using Patch Analyst.

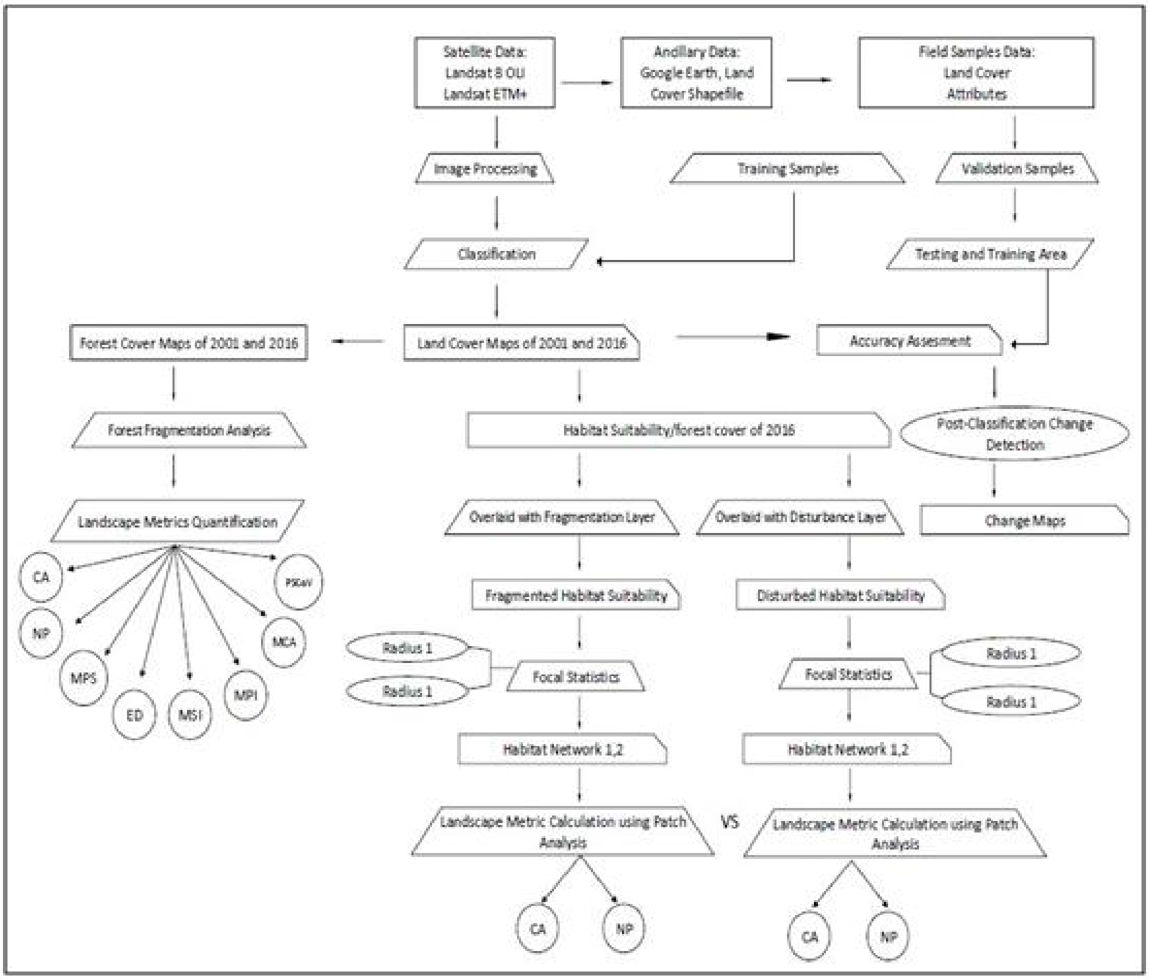


Figure 4.2 General flow of data processing and analysis

4.3 Data Collection

In general, data collected in this study consisted of primary data and secondary data. Primary data includes remotely sensed data and ground truthing data for accuracy of the interpreted land cover. Secondary data that encompassed spatial information was sourced from related institutions or websites. For the spatial data collected, there are two types of data which represent the geographical information of the real world: vector data and raster data. The former provides a vector view, which allocates coordinates (x, y) in the form of points, lines or area (polygons) to form a map (O'Sullivan & Unwin, 2010). Polygons represent areas that have boundaries (countries, lakes and forest areas), lines represent linear objects (roads, rivers and pipelines), and points represent subjects with limited spatial extent (this depends on map scale, but can include cities, schools and individual trees). Polygons, lines and points are called vector data (Ormsby, Napoleon, Burke, Groessl, & Bowden, 2010). Raster data define objects on the ground using a grid of small units, called pixels (O'Sullivan & Unwin, 2010) (see Figure 4.3).

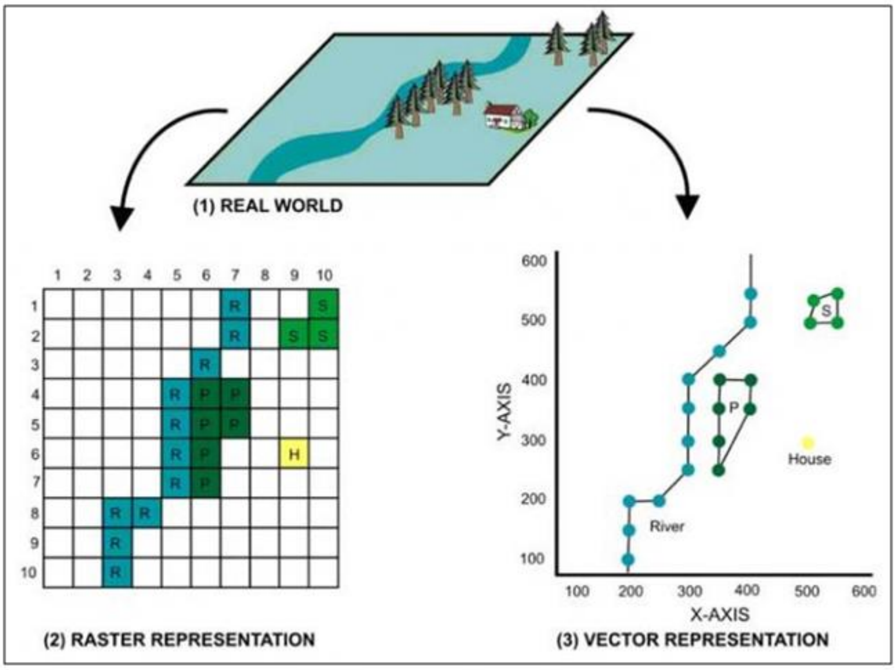


Figure 4.3 Vector and raster data (Isabelle, 2011)

To obtain such primary and secondary data required for the study, field research was conducted from June to August 2016. A special arrangement was made to visit GHSNP office and the Ministry of Environment and Forestry in Jakarta, the capital of Indonesia. Visits to Gunung Halimun Salak

National Park were arranged to collect sufficient field data for validation of the land cover of GHSNP and wildlife in the study site. Prior to this, Landsat image interpretation was conducted and Google earth was used to assist the interpretation of image.

4.3.1 Primary Data

Landsat 8 OLI data and Landsat Enhanced Thematic Mapper Plus (ETM+) imagery for 2001 and 2016 (path/row: 122/065), were used in this study. Both of the images were obtained from the United States Geological Survey (USGS) website (http://glovis.usgs.gov/). This platform was chosen because it has two sensors: the Operational Land Imager (OLI) and the thermal infrared sensor (TIRS). OLI collects data at a 30-m spatial resolution with eight bands located in the visible, near-infrared and in the shortwave infrared regions of the electromagnetic spectrum, plus an additional panchromatic band at 15-m spatial resolution (Jiménez-Muñoz, Sobrino, Skoković, Mattar, & Cristóbal, 2014). During field work, a set of 150 points was sampled using a handheld Global Positioning Systems (GPS) GARMIN Etrex 30 and a mobile phone (IPhone 5). The sampling points, however, were not equally distributed within each land cover due to the sampling strategy applied. At the sampling units, which were about 30 by 30 metres, visual estimates of the biophysical attributes and coordinate points were made and recorded in the field observation sheet. Digital photographs of the sites were also taken. These data were later entered into Microsoft Excel 2010 spreadsheet at the end of the field survey (see Appendix A).

4.3.2 Secondary Data

Secondary data encompassing spatial data from relevant institutions, including land cover maps, maps of administrative boundaries of GHSNP and others are summarized in Table 4.1. In addition, Topographic Maps of Indonesia (TMoI) was also used as secondary data. This dataset consist of vector data of natural and man-made features, including administrative boundaries; transportation and utilities; hydrographic maps with elements such as rivers, lakes and others. TMoI was obtained from the Geospatial Information Agency (BIG) by downloading the data via http://www.bakosurtanal.go.id.

Table 4.1 Data collection from field visits

Data/Information	Name of Institution/Organization	Acronym	Type of Data
Maps of road network	- The Directorate of Inventory and Monitoring of Forest Resources (*Direktorat Inventarisasi dan Pemantauan Sumber Daya Hutan*)	DIPSDH	Spatial
Land cover (time series 1990, 2000, 2003, 2006, 2009 and 2011)	- The Directorate of Inventory and Monitoring of Forest Resources (*Direktorat Inventarisasi dan Pemantauan Sumber Daya Hutan*)	DIPSDH	Spatial
Maps of Administrative Boundaries, Geographic Condition of GHSNP	- Gunung Halimun Salak National Park Office	GHSNP Office	Spatial
Other data and information related to environmental condition and biodiversity in GHSNP	- Gunung Halimun Salak National Park Office	GHSNP Office	Nonspatial

4.4 Data processing and Analysis

This section provides information on the data processing and analysis of the land cover classification, fragmentation and disturbance analysis.

4.4.1 Analysis of Land Cover and Land Cover Change

The land cover data from the field survey were analysed based on the biophysical attributes. The eight classes were used for classification as well as change analyses. Thus, steps being taken in the interpretation of Landsat imagery for the area of GHSNP are described in the following sections:

4.4.1.1 Determination of Research Area

Since geometric and radiometric correction of the satellite data had already been conducted by the suppliers, clipping the Landsat image to determine the study area was necessary before the area could be analysed. The method used for this step was *extract by mask* using ArcGIS, conducted by cutting the Landsat image with the GHSNP administrative boundary layer. These were afterwards reprojected, using ArcGIS software, to UTM zone 48 south (map projection) and WGS 84 (datum and ellipsoid).

4.4.1.2 Classification of the image

The classes for land cover classification was based on the classification system of the Planning Agency of the Ministry of Forestry through Regulation of Ministry of Forestry No. 67/ Menhut-II/2006b regarding Criteria and Standards of Forest Inventory (see Table 4.2). Classification was carried out by using visual interpretation (digitising on screen) at a scale of 1: 25,000, with an approach using the elements that include colour, texture (the frequency of changes in colour), pattern (the compilation of spatial objects), size , form (directly related to the general shape, configuration or shape or the framework of a single object), and the shadow and the site (the location of an object to another object) (Lillesand, Kiefer, & Chipman, 2014). In this classification, industrial plantation forest was merged with primary and secondary dry forest as industrial plantations no longer occur within the national park. Dryland and mixed dryland agriculture were also merged to reduce the number of classes. In this step, Google Earth and spatial land cover data from the previous year were used as a reference.

Table 4.2 Land cover description and classification system according to Ministry of Forestry and Environment of Indonesia

Number	Land Cover		Code	description
1	Forest	Primary dry forest	2001	Forests that grow on dry land habitats. It can be either forests occupied lowland, hills and mountains and high plains or such tropical forests that are still original and clearly visible indications of no human activities and the ecological processes are not significantly disturbed.
		Secondary dry forest	2002	Forests that grow on dry land habitats. It can be either forests occupied lowland, hills and mountains and high plains or such tropical forests that have been altered by humans or have been cleared by natural or man-made causes, such as agriculture or ranching.
		Industrial	2006	The entire area of industrial plantation forest

		plantation forest		(IPF) that has either already been planted or the area is going to be planted.
2	Agriculture	Dryland agriculture	2009	All agricultural activities such as, mixed farms and fields.
		Mixed dryland agriculture	2009	All agricultural activities in the dry land, together with bushes, shrubs and logged forest around the agriculture area
3	Grassland		2007	Former dry forest area that has been cleared and that has grown back, dominated by a small to medium-sized vegetation and no longer showing the forest area look.
4	Bare lands		2014	Non-built-up land with no, or insignificant, vegetation cover. The lands could be former fires area or lands overgrown with grass/reeds.
5	Plantations		2010	Land that is used for agricultural activities without crop/plant replacement for about two years. Harvest usually occurs after a year or more.
6	Paddy field		2009	Agricultural areas flooded by water using the technology of irrigation, rain-fed, lowland or tidal characterized by patterns of embankment, with cultivated types of short-lived food crops (rice).
7	Water body		5001	Open water – all areas of open water, generally with less than 25% cover of vegetation or soil.
8	Built-up		2012	Acreage or land used as a residential area or neighbourhood occupancy and for activities that support people's lives.

A supervised classification approach was used in this image classification. The supervised classification approach involves, firstly, training the classification algorithm with a number of sites where the classification output is known. This involves polygons around areas of known land cover. Then, summary information about the spectral characteristics of the training sites of the various classes of output, derived from the training samples, was applied to all cells within an area to assign them to one of the pre-defined types of landscape. The maximum likelihood classification (MLC) were selected for this supervised land cover classification of Landsat data.

The MLC has been a popular parametric classifier used for remote sensing data classification (Foody, 2002; Jia et al., 2014). The MLC assumes that a hyper-ellipsoid decision volume can be used to approximate the shape of the data clusters. Moreover, for a given unknown pixel, the probability of membership in each class is calculated using the mean feature vectors of the classes, the covariance matrix and the prior probability (Duda & Hart, 1973). The unknown pixel is considered to belong to the class with the maximum probability of membership. Based on the knowledge of land cover distribution characteristics, eight classes were identified as the final class types: forest, agriculture, grassland, bare lands, plantations, paddy field, water body and built-up. Representative sample collection is the most time-consuming and essential process in land cover classification efforts. Throughout the process of interpretation, RGB 543 composites were used for Landsat ETM+ and RGB 654 for Landsat-8 colour composition because it has the best information for the identification of land cover (M. Wulder, Skakun, Kurz, & White, 2004). Landsat images for 2001 and 2016 were then interpreted to finally become land cover maps of 2001 and 2016. To analyse the land cover change between 2001 and 2016, an overlaying process between 2001 land cover maps and 2016 land cover maps was carried out using ArcGIS software.

4.4.1.3 Landsat Assessment Accuracy

To validate the land cover classification performance, the classification results using the MLC were assessed via visual observations. Quantitative classification accuracy was conducted and an error matrix was created. An error matrix is a square array of numbers organized in rows and columns that express the number of sample units (i.e., pixels, clusters of pixels, or polygons) assigned to a particular category relative to the actual category as indicated by the reference data (see Appendix A). The columns usually represent the reference data while the rows indicate the classification generated from the remotely sensed data. An error matrix is such a very effective way to represent accuracy, that the accuracies of each category are plainly described along with both the errors of inclusion (commission errors) and errors of exclusion (omission errors) which are present in the classification (Congalton, Plourde, & Bossler, 2002).

The reference data were obtained from randomly selected sample points derived from interpretation, ground observation, and ground measurement. In this case, the test points were determined from the 150 points collected during the field research. Overall Accuracy and Kappa Accuracy were calculated by using the error matrix where the rows are a class of land cover results of image interpretation and columns are a class of land cover results of checks using high-resolution imagery (Google Earth).

Overall accuracy was computed by dividing the total number of correctly classified pixels (i.e., the sum of the major diagonal) by the total number of pixels in the error matrix. This accuracy measure indicates the probability of a reference sample being correctly classified. On the other hand, Kappa analysis is a discrete multivariate technique used in accuracy assessments

(Congalton, Oderwald, & Mead, 1983; Janssen & Vanderwel, 1994). Kappa analysis yields a K_{hat} statistic (an estimate of Kappa) that is a measure of agreement or accuracy (Congalton, 1991). The K_{hat} statistic is computed as:

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

where r is the number of rows in the matrix, x_{ii} is the number of observations in row $_i$ and column $_i$, x_{i+} and x_{+1} are the marginal totals for row i and column i respectively and N is the total number of observations.

The values of the K_{hat} statistic can range from +1 to –1. However, since there should be a positive correlation between the remotely sensed classification and the reference data, positive K_{hat} values are expected. Landis and Koch (1977) characterized the possible ranges for K_{hat} into three groupings: a value greater than 0.80 (i.e., 80%) represents strong agreement; a value between 0.40 and 0.80 (i.e., 40–80%) represents moderate agreement; and a value below 0.40 (i.e., 40%) represents poor agreement. Using this technique, it is possible to test if an individual land cover map generated from remotely sensed data is significantly better than if the map had been generated by randomly assigning labels to areas. The second test allows for the comparison of any two matrices to see if they are statistically significantly different (Congalton et al., 2002). Figure 4.4 presents an example error matrix.

Reference Data

	D	C	BA	SB	row total
D	65	4	22	24	115
C	6	81	5	8	100
BA	0	11	85	19	115
SB	4	7	3	90	104
column total	75	103	115	141	434

Land Cover Categories

D = deciduous
C = conifer
BA = barren
SB = shrub

OVERALL ACCURACY = 321/434 = 74%

PRODUCER'S ACCURACY

D = 65 / 75 = 87%
C = 81 / 103 = 79%
BA = 85 / 115 = 74%
SB = 90 / 141 = 64%

USER'S ACCURACY

D = 65 / 115 = 57%
C = 81 / 100 = 81%
BA = 85 / 115 = 74%
SB = 90 / 104 = 87%

Figure 4.4 an example of error matrix (Congalton, 1991)

4.4.1.4 Post-Classification Analysis

With the information derived from remotely sensed data and conventional data, land cover change and its impacts can be identified for further national park management. The most obvious method of change detection is a comparative analysis of spectral classifications for times t_1 and t_2 produced independently (Landis & Koch, 1977). This is also suggested by Stow, Tinney, and Estes (1980) who suggest that a change map of two images will only be generally as accurate as the product of the accuracies of each individual classification. Hence, accuracy of relevant class changes depends on the spectral reparability of classes involved and in this study, both Landsat 8 OLI and Landsat ETM+ data of both dates were independently classified using the maximum likelihood classifier and was then overlaid to detect the changes.

4.4.1.5 Quantification and Assessment of Changes in Forest Configuration in GHSNP

The term 'fragmentation' has been defined as simultaneous reduction of forest area, increase in forest edge, and subdivision of large forest areas into smaller non-contiguous fragments (W. F. Laurance, Delamônica, Laurance, Vasconcelos, & Lovejoy, 2000). The degree of fragmentation has been illustrated as a function of the varying size, shape, spatial distribution, and density of patches (Jorge & Garcia, 1997). Ecologist have been using metrics for assessing forest fragmentation and its impact on landscape structure (Lele & Joshi, 2009; Reed, Johnson-Barnard, & Baker, 1996); The most common metrics applied to analyse forest fragmentation at the class or patch type level include class area, number of patches, mean patch size, edge density, mean core area, mean proximity index, mean shape index, and patch size coefficient of variation (Hargis et al., 1999; Leitão & Ahern, 2002; Luque et al., 1994; Reed et al., 1996; Zheng, Wallin, & Hao, 1997). In the present study, details of the data and software used for computation of selected metrics at the class level to quantify fragmentation in the forest area of GHSNP are presented in Table 5.5

Table 4.3 Landscape indices used for spatial pattern analysis of forest fragmentation

No	Metrics	Code	Description
1	Class area (ha)	CA	The sum of the areas (m^2) of all patches of the corresponding patch type. It is a measure of landscape composition representing how much a particular patch type occupies a landscape; a class with greater density of patches indicate that it is subdivided into many patches and thus could be considered more fragmented (Cushman, McGarigal, & Neel, 2008; McGarigal, Cushman, Neel, & Ene, 2002).
2	Number of patches	NP	Measures the extent of fragmentation of the entire landscape. It represents the number of patches for the class(McGarigal & Marks, 1995).
3	Mean patch size (ha)	MPS	The average area of a patch of a particular class, and depends on data resolution; sensitive class to addition/deletion of small patches. Mean patch size can serve as a habitat fragmentation index; smaller mean patch size indicates more fragmented forest (McGarigal & Marks, 1995)
4	Edge density	ED	A measure of landscape configuration. It standardises edge to a per unit area basis that is based on the ratio between total edges and total area, facilitating comparisons among landscapes of varying size. Larger edge density represents more spatial heterogeneity and thus the class is less compact; amount of edge relative to total area is expected to increase in initial stages of fragmentation (McGarigal & Marks, 1995)
5	Mean shape index	MSI	The average shape index of patches of corresponding forest type, adjusted by a constant for a square standard (raster); patches are expected to become less geometrically complex in managed forest (McGarigal & Marks, 1995)
6	Mean core area (ha)	MCA	The average core area of the patches of the corresponding forest type; one of the main effects of fragmentation is the conversion of interior habitat to edge habitat. It is estimated that the amount of core area will decrease as a result of fragmentation (McGarigal & Marks, 1995)
7	Mean proximity index	MPI	Average proximity index for all patches in a class. Proximity index is calculated as the sum of the ratio of patch size to nearest neighbour edge-to-edge distance for all patches within a specified search radius (Hargis, Bissonette, & David, 1997).

8	Patch size coefficient of variation	PSCoV	Patch size coefficient of variation is generally preferable to patch size standard deviation for comparing patch size variability between landscapes; it is expected to decrease as patches become less variable or more similar in patch size over time (McGarigal & Marks, 1995)

In this study, landscape metrics have been calculated at the class level for total forest cover of GHSNP. In this case, some metrics applied in this study consist of class area which is considered to be the most important statistical parameter; number of patch and edge density which have been widely used to analyse the extent of fragmentation of forest classes. Both patch and density metrics are important indices as they provide information regarding the pattern of fragmentation. The level of fragmentation was also examined by calculating mean proximity index and mean shape index which are widely used indices in landscape ecological studies.

Quantification and comparison of the spatial configuration of forest patches in GHSNP between two time periods was conducted based on selected important landscape metrics and other related forest fragmentation studies (Hansen, Franklin, Woudsma, & Peterson, 2001). In this study, Patch Analyst 4.0 for ArcGIS 10.3 was used to generate the landscape metrics or indices at class level (McGarigal & Marks, 1995). This spatial software provides a wide-ranging choice of landscape metrics at both landscape and class levels and computes spatial statistics on both raster files (e.g. raster grids) and polygon files (vector format such as shape files).

Six groups of statistics are available in Patch Analyst:

- Patch density and size metrics that analyse landscape fragmentation and configuration;
- Area metrics quantify landscape or class area;
- Edge metrics which attribute the amount, length, and distribution of edges between specific patch types;
- Core area metrics which measure the extent of the patch deprived of its outer belt;
- Shape metrics which measure the geometric complexity;
- Diversity and interspersion metrics which measure patch isolation.

Some indices are only applicable at the landscape level, while others are only applicable on shape themes or on grid themes. Both nonspatial composition and spatial configuration were commonly used to analyse spatial heterogeneity and measure the fragmentation of natural ecosystems

(Geneletti, 2004; Tischendorf, 2001; Trani & Giles Jr, 1999). A summary of certain landscape metrics used for forest landscape analysis in this study is shown in Table 4.3

Table 4.4 Summary of relevant landscape metrics used for the study and their abbreviations (McGarigal and Marks, 1995).

Selected Spatial metrics	Abbreviation	Unit
Number of patches	NUMP	ha
Mean Patch Size	MPS	ha
Edge Density	ED	m/ha
Mean Shape Index	MSI	
Class Area	CA	ha
Patch Density	PD	
Patch Size Coefficient of Variation of Variation	MPCoV	
Mean Proximity Index	MPI	
Mean Core Area	MCA	

4.4.2 Impact Analysis of Fragmentation and Disturbance

There were several steps taken in the process of assessing the overall impacts of fragmentation and disturbance on mammals and birds.

4.4.2.1 Modelling the spatial distribution of fragmentation effects

The spatial distribution of transportation infrastructure (road) effects of different intensity were modelled in order to enable the analysis of their consequences. The two selected models published by Benítez-López et al. (2010) on the ratio between the species abundance at varying distances to infrastructure — referred to disturbance or effect distance— on mammals and birds were used to quantify the impact of roads. These models were selected because the study was conducted using a meta-analysis —a method of synthesizing results from multiple studies.

The models used for this analysis demonstrates a binomial prediction of effects on mammals and birds, expressed as Mean Species Abundance (MSA). MSA was used as the effect size measure. These are derived from generalized linear mixed effect models and a logit link function. For this study, two spatial datasets on the predicted effects on mammals and birds ($MSA_{mammals}$ and MSA_{birds}) were created in ArcGIS (Esri, 2011) at a particular resolution by applying a logit transformation:

$$MSA_{(estimated)} = \frac{e^{u}}{1 + e^{u}}$$

where MSA (estimated) is the predicted MSA at the observed distance from the altered area ranging from 0 to 1 and u is the linear exponent describing the log-transformed probability of the presence of a species at a certain distance x from the area:

$$u = \ln\left(\frac{P_i}{1-P_i}\right) = a + b\mathrm{x}$$

Where “a” is the estimated value of u when x = 0 and “b” is the regression coefficient for the independent variable x. The values of regression coefficient “b” and “a “were taken from (Benítez-López et al., 2010). The distance variable x took the value of each cell in a raster containing the Euclidian distance from the altered area. The $MSA_{mammals}$ and MSA_{birds} layers were calculated from an altered area layer with an Average Daily Traffic of 100 or more vehicles. These ADT thresholds were identified from the studies analysed in Benítez-López et al. (2010), and set according to the lowest traffic volumes with significant influence on MSA. The derived MSA datasets were then reclassified into four effect intensity zones with break values of 0.5, 0.7 and 0.9 (Table 4.4).

Table 4.5 Classification of Mean Species Abundance (Karlson, 2013)

	MSA Intervals			
Mammals	0.35 - 0.5	0.5- 0.7	0.7- 0.9	0.9- 1.0
Birds	0.3 - 0.5	0.5- 0.7	0.7- 0.9	0.9- 1.0

The MSA values obtained, ranged from 0 to 1.0. This means that MSA in an unaltered control location is close to 1, while a low MSA value indicates a high disturbance effect (Figure 4.5). None of the models predicted zero MSA at the actual altered road but instead 0.35 for mammals and 0.3 for birds (Table 4.4). This is interpreted as a realistic value for these predictions, as transport infrastructure can create attractive habitat for some species and have insignificant or no effects on others (Fahrig & Rytwinski, 2009; Forman, 2003).

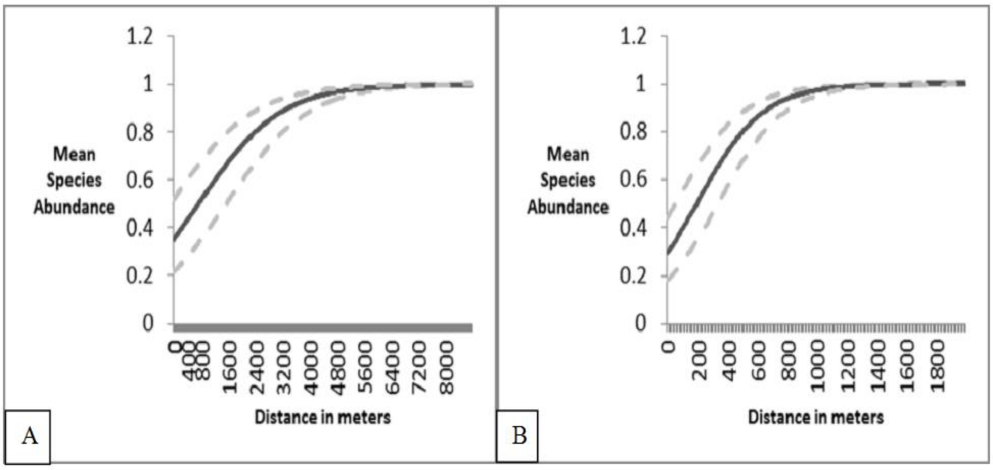


Figure 4.5 Predictions of the Mean Species Abundance of mammals (A) and birds (B) as an effect of distance to infrastructure, based on statistical analyses of empirical data (Benítez-López et al., 2010)

A generic effect of roads on birds and mammals was then estimated as the areal intersection between forest habitat and the four effect intensity zones. A good example of different road effect on a habitat type was illustrated by Karlson (2015) who examined different road effect intensity intervals relative to the distribution of pine forest older than 70 years in a Swedish pine forest (Figure 4.6).

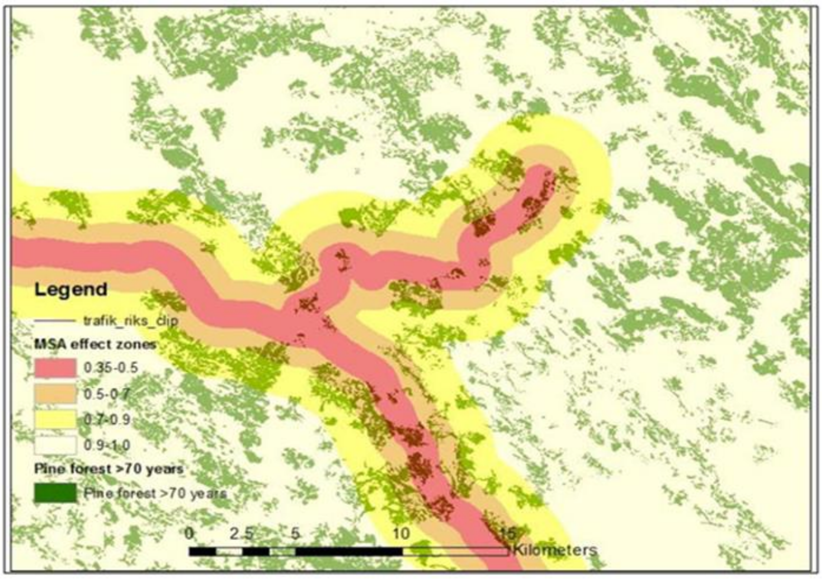


Figure 4.6 Illustration of the different road effect intensity intervals relative to the distribution of pine forest older than 70 years in a Swedish landscape (Karlson, 2015)

4.4.2.2 Assessing the overall impact of fragmentation and disturbance effect

This was conducted by quantifying the forest cover in GHSNP, localized within zones of different effect intensity. Hence, it was assumed that all forest and grassland cover types —a scale from 0 to 1— are a suitable habitat for mammals and birds in GHSNP. For this purpose, the MSA layers were overlaid on the spatial datasets of forest cover, and then the forest area per effect intensity zones were calculated.

4.4.2.3 Exploring fragmentation and ecological effects

Analysing the effects of fragmentation and disturbance on habitat networks for a selected ecological profiles consisted of several stages. These are described in sections 4.4.2.3.1 to 4.4.2.3.4.

4.4.2.3.1 Construction of ecological profiles

To construct metrics that encompass ecological processes and can serve as ecological indicators on a landscape scale, the concept of ecological profiles was applied (Angelstam et al., 2004; Mörtberg, 1998; Vos et al., 2001). The concept is similar to that of umbrella species, and follows the same recommendations on selection of model species: high degree of habitat specialization, high sensitivity to disturbance and area demands representative of the scale of context (Angelstam et al., 2004; Edman, Angelstam, Mikusiński, Roberge, & Sikora, 2011; Fleishman, Murphy, & Brussard, 2000). In other words, the species with similar habitat requirements and dispersal capacity were categorized into groups with similar ecological profiles, on the assumption that these groups would respond in a similar way to specific environmental changes, such as habitat loss and fragmentation (see Figure 4.7) (Vos et al., 2001).

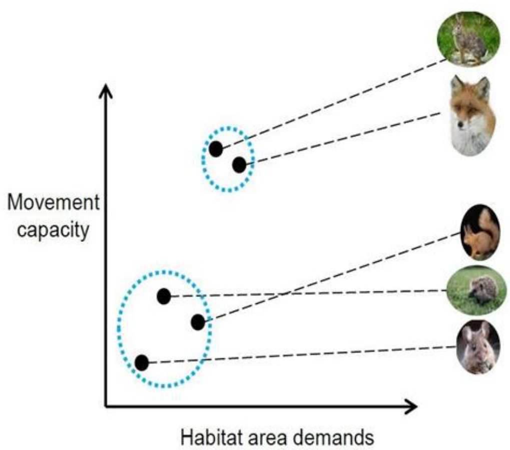


Figure 4.7 Conceptual model of the ecological profile system. A single species might be chosen to represent others, if they share, to a minimum degree, a selection of traits and resource requirements (Karlson, 2015)

In this study, the analysis encompassed similar habitat, between mammals and birds, and uncertainties concerning habitat requirements and movement capacity. Among all the animal taxa, mammal and bird populations were chosen for the analysis. The first reason is that both have been widely reported to be declining at rates related to their distance from infrastructure. Some studies show that bird populations seem to be affected within a few hundred metres from infrastructure, whereas a reduction in mammal populations has been found at distances of a few hundred metres up to several kilometres from infrastructure (McLellan & Shackleton, 1989; Nellemann, Vistnes, Jordhøy, Strand, & Newton, 2003; Ortega & Capen, 1999). The second reason is that road traffic is likely to play a big a role in the decline of both bird and mammal populations near roads (Gagnon, Theimer, Dodd, Manzo, & Schweinsburg, 2007; Reijnen & Foppen, 1995; Reijnen et al., 1996; Rheindt, 2003; Van der Zande, Ter Keurs, & Van der Weijden, 1980).

To achieve this, a literature review was carried out to collect information about habitat requirements (habitat type and home-range size) for mammals and birds and these were then classified into ecological profiles. The literature review mainly included studies in which home-ranges are estimated. The home-range sizes used in this study were retrieved from local conservation organizations such as GHSNP office or literature. In this way, a set of model species were selected to help define the ecological profiles. Such profiles were based on the habitat requirements of these species, habitat type, home-range size and related average movement distance. The requirements of the model species were considered to represent realistic ecological profiles that could be used for creating representative habitat networks (Karlson & Mörtberg, 2015). The ecological profiles represented different forest mammals and birds in GHSNP, with area demands from approximately 15 ha to 3500 ha.

4.4.2.3.2 Habitat networks for ecological profiles

Habitat networks were generated using focal statistics assuming a “patch” to be all available pixels within the daily activity range. For each ecological profile, a set of habitat networks were created in order to take uncertainties into account. Furthermore, the windows, reflecting the habitat area demands of each ecological profile, were assumed to be circular in shape and the movement distance to be the radius of that circle, then pixels with suitability < 0.5 were excluded from the datasets. Focal statistics were calculated for each pixel based on all the pixel values within a chosen window. For each ecological profile, two different radii were used to include uncertainties in home-range size, resulting in two habitat networks with slightly different spatial properties. To include eventual edge and contrast effects between habitat types, the average habitat suitability of the pixels in the window was also calculated, and pixels with a suitability of > 0.5 were reclassified into habitat patches (Karlson & Mörtberg, 2015). In sum, 10 habitat networks were created. The GIS-analyses were performed at a resolution of 30 x 30 m.

Table 4.6 Habitat Network of Each Ecological Profile

No	Ecological profile	Species	Habitat Network 1 (m)	Habitat Network 2 (m)
1	Forest-Grassland mammals High area demands	leopard dhole	3605.6	5916.1
2	Forest grassland mammals -medium area demands	leopard cat wild boar small Indian civet	1581.1	2323.8
3	Forest mammals less than 1500 asl area demands	slow loris lesser mouse-deer	141.4	500.0
4	Forest mammals more than 1500 asl area demands	Javan langur silvery gibbon	387.3	685.6
5	forest birds	javan hawk-eagle	1593.7	2000

4.4.2.3.3 Construction of fragmented and disturbed habitat networks

Each of the original habitat suitability layers were overlaid by (1) the national road network layer to represent fragmentation effects, from here referred to as the fragmentation layer, and (2) the effect intensity zone layer to represent disturbance effects, from here on referred to as the disturbance layer. The zones with highest disturbance (MSA < 0.5) were selected for this analysis to enable a clear demonstration of the potential effects. Consequently, when the original habitat suitability layers were overlaid by the fragmentation layer, the habitat suitability value of the affected pixels was reduced to zero to represent the fragmentation effect. By contrast, the value of pixels affected by the disturbance layer was reduced by a factor 0.5 times the original habitat suitability; this was done in order to represent a reduction by at least half where MSA < 0.5 due to road effects (Karlson, 2015). New habitat networks were then created. To summarize, for each original habitat suitability layer, one "fragmented" and one "disturbed" habitat suitability layer was created. For each of these, two "fragmented" habitat networks and two "disturbed" habitat networks were created. In total, for each of the ecological profiles, three habitat suitability layers and six habitat networks were created. The raster analysis was performed at resolution of 30 m × 30 m; thus, roads were considered to be 30 m wide.

4.4.2.3.4 Quantification and comparison of the impacts on habitat networks

Disturbance and fragmentation effects were generally assessed by quantifying the relative change in the amount of habitat network and connectivity of the fragmented and the disturbed habitat networks. Hence, it was conducted by calculating three landscape ecological metrics. After that, a comparison was performed by evaluating the differences in these metrics between "fragmentation" of an ecological profile and "disturbance" of the same ecological profile. The comparison was essentially based on the hypothesis that disturbance effects to some extent might alter the physical properties of an ecological profile network by altering its quality. The software program, Patch Analyst (Rempel, Elkie, & Carr, 1999) was used to calculate two ecologically important quantities: habitat amount, expressed by the landscape metrics of class area (CA), and connectivity which is expressed by the metric number of patches (NP) (Table 4.5). The metric scores are normalized against those calculated for the unaltered habitat networks.

Applying a set of different home-range sizes for each ecological profile resulted in a set of habitat networks, which consequently resulted in a range of responses. Thus, such a response range was calculated as the difference in metrics between home-range sizes used for each ecological profile.

Table 4.7 Description of landscape metrics used in fragmentation analysis

Landscape Metrics		
Name	Total Class Area (CA)	Number of Patches (%)
Description	Total area of all habitat patches in hectare	Number of individual patches

Chapter 5

Results

This chapter presents the results of the land cover of GHSNP analysis, forest fragmentation analysis, and in particular, road fragmentation and disturbance effects on mammals and birds. Specifically, section 5.1 describes the classes used for the classification, accuracy assessment of land cover classification, land cover change and forest cover change analysis between 2001 and 2016; section 5.2 presents a quantification and assessment of changes in forest configuration of GHSNP between 2001 and 2016; section 5.3 presents the fragmentation and disturbance results due to the road network in GHSNP. The potential road effects on mammals and birds in GHSNP are also presented in more depth.

5.1 Land Cover of Gunung Halimun Salak National Park

5.1.1 Land Cover Classification

In this study, the land cover in GHSNP was classified into eight categories in which primary dry forest, secondary dry forest, and industrial plantation forest were merged as forest cover; dryland agriculture and mixed dryland agriculture were merged into the agriculture class; and the other land cover classes in this study were grassland, bare land, plantations, paddy field, water body, and residential area. Land cover classification maps of GHSNP plus enclaves for 2001 and 2016 are presented in Figure 5.1 and 5.2.

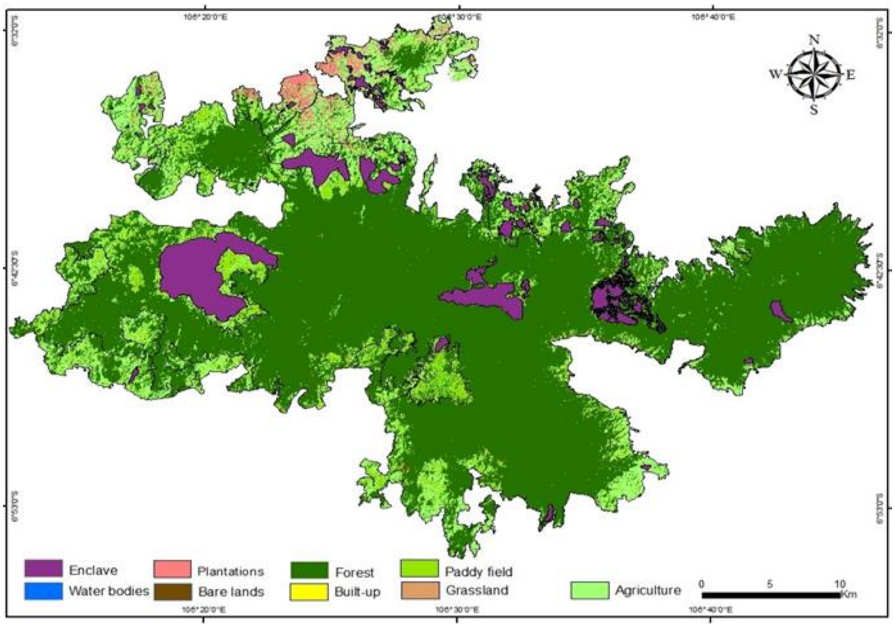


Figure 5.1 Land cover classification map of GHSNP in 2001

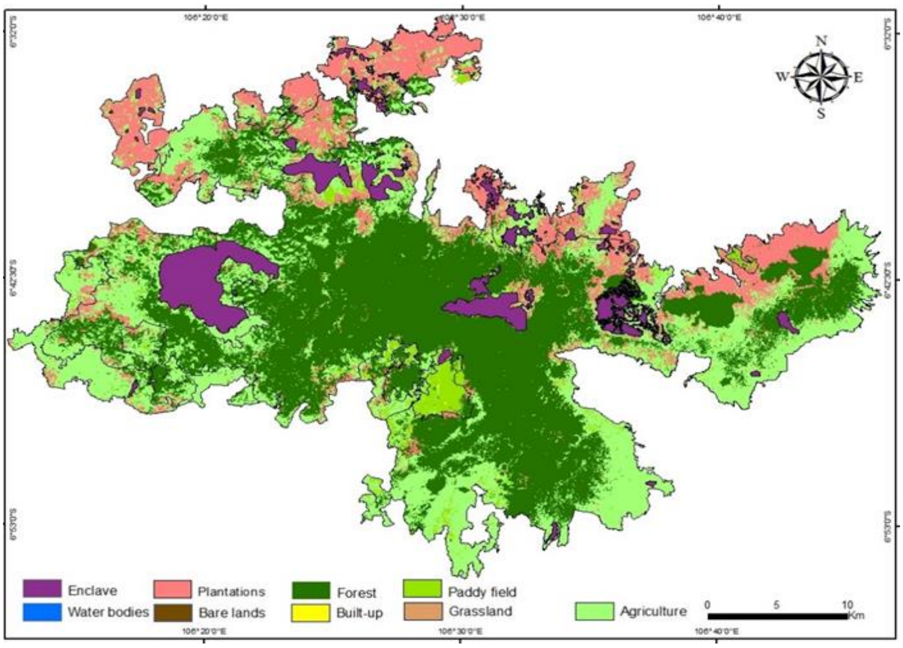


Figure 5.2 Land cover classification map of GHSNP in 2016

Once the initial classifications were done, land cover data derived from the classification process were used then to gain information about forest fragmentation and disturbance effects. Southworth et al. (2002) also derived landscape metrics from the classifications to analyse any changes in diversity and fragmentation of the landscape. Mapping the percentage of impervious surface area, an alternative way of monitoring urban growth, has also been conducted using urban masks generated from the land cover classification maps (Bauer, Yuan, & Sawaya, 2003). In addition, similar classifications have been used as inputs to an environmental impact analysis project by the U.S. Environmental Protection Agency and in a land use transformation model to project future land use change in the Detroit and the Twin Cities Metropolitan Areas (Pijanowski, Shellito, Bauer, & Sawaya, 2001).

5.1.2 Assessment of Land Cover Classification Accuracy

An acceptable overall accuracy for land cover maps is set > 85% (J. Anderson, Hardy, Roach, & Witmer, 1976) and >90% (Lins & Kleckner, 1996). An error matrix (Congalton, 1991; Story & Congalton, 1986) was used to calculate overall classification accuracy (P), a confidence interval for P, producer's accuracy, and user's accuracy. P is a simple, intuitive measure of the proportion of total sampling units that were correctly classified; it indicates the overall probability that a unit on the ground was correctly classified. User's accuracy is a measure of commission errors, indicating the probability that a unit within an individual category is correctly classified. Producer's accuracy is a measure of omission errors, indicating the probability that a reference data sample is correctly classified. All of those accuracies are useful for the map reader because it measures the degree to which a land cover type on the earth can be distinguished and mapped using remote sensing data. Meanwhile, Kappa calculation tends to over-estimate the level of chance agreement. So, Kappa will consistently underestimate the overall classification (Foody, Campbell, Trodd, & Wood, 1992).

In this study, error matrices were used to assess classification accuracy and are summarized in Table 5.1. The overall classification accuracy was found to be 91.3% and 92.7% for overall kappa statistics. For a map derived from satellite imagery, the measures of overall classification accuracy (91.3%) indicate that the map classes were properly distinguished using the classification method applied (Landis & Koch, 1977).

The user accuracy compares the number of pixels of a classified image that are correct based on the reference data. User's and producer's accuracies of individual classes for the 2016 Landsat image were relatively high, ranging from 66% to 100%. In general, forest, grassland and built-up area had the highest overall accuracy of 100%. In contrast, agriculture had the lowest overall accuracy at 66.6%.

The high accuracy of the classification and the relevance of the accuracy assessments results are attributable to the following factors: spatially accurate reference data; a well-planned sampling strategy; a simple and spectrally distinct classification based on detailed phytosociological information, and prior knowledge; and experience in generating land cover maps for the region (Muller et al., 1998).

Table 5.1 Overall accuracy, producer's accuracy, user's accuracy, and Kappa coefficient for the classification of Landsat image 2016

No	Class of Land Cover	Reference									Producer's accuracy (%)	User's accuracy (%)	Overall accuracy	Overall Kappa Statistics
		Forest	Grassland	Plantation	Built-up	Open Space	Water Body	Agriculture	Rice Field	Total				
1	Forest	40	0	0	0	0	0	0	0	40	100.0	100.0	91.3%	92.7%
2	Grassland	0	18	0	0	0	0	0	0	18	90.0	100.0		
3	Plantation	0	0	10	0	0	0	2	0	12	83.3	83.3		
4	Built-up	0	0	0	10	0	0	0	0	10	100.0	100.0		
5	Open Space	0	0	0	0	10	1	0	0	11	100.0	90.9		
6	Water Body	0	0	0	0	0	11	0	0	11	84.6	100		
7	Agriculture	0	2	2	0	0	1	20	5	30	90.9	66.6		
8	Rice Field	0	0	0	0	0	0	0	18	18	78.2	100		
	Total	40	20	12	10	10	13	22	23	150				

5.1.3 Land Cover Change Analysis: an Overview between 2001 and 2016

The land cover and land cover changes in GHSNP for the period of 2001 and 2016 are shown in Figure 5.3, while each class area and its percentage changes for the fifteen years are summarized in Table 5.2. In 2001, forest land cover classes dominated the GHSNP landscape (72.8%); however, 15 years later, the forest cover had declined by 35.4% by 2016. In addition, paddy fields, bare lands, and grassland decreased by 46.4%, 22.9% and 19.4% respectively.

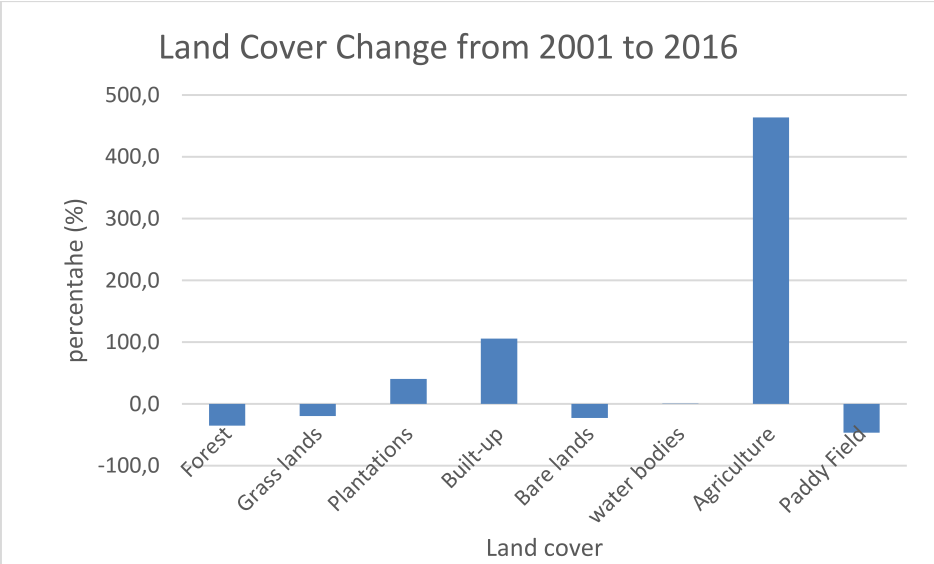


Figure 5.3 The percentage of land cover changes in GHSNP between 2001 and 2016

Table 5.2 Areas and percentages of each land cover class in 2001 and 2016

Class	2001		2016		Changes	
	ha	%	ha	%	ha	%
Forest	76,580.1	72.8	49,500.4	47.1	-27,079.7	-35.4
Grassland	1,743.9	1.7	1,405.2	1.3	-338.6	-19.4
Plantations	11,993.0	11.4	16,862.8	16.0	4,869.8	40.6
Built-up	511.3	0.5	1,051.1	1.0	539.9	105.6
Bare lands	1,349.4	1.3	1,040.0	1.0	-309.4	-22.9
water bodies	168.5	0.2	168.4	0.2	0.0	0.0
Agriculture	5,541.3	5.3	31,238.2	29.7	25,697.0	463.7
Paddy Field	7,286.5	6.9	3,907.5	3.7	-3,379.0	-46.4
Total	105,174.0	100.0	105,174.0	100.0		

In 2001 agriculture (5.3%) and built-up areas (0.5%) covered a small percentage within the GHSNP territory, but by 2016 it was found that agriculture and built-up areas experienced a marked increase of 463.7% and 105.6% respectively. In addition, plantations also experienced an increase by 40.6% over the study period (Table 5.2).

5.1.4 Forest Cover Change Analysis between 2001 and 2016

Post-classification change detection was carried out for the forest cover results of 2001 and 2016 and enclave areas —which are not part of GHNSP areas— were not included in this analysis. Based on this analysis (Figure 5.4) it shows that deforestation had been occurred over a fifteen-year period, particularly around the edge of GHSNP area. The finding is emphasised by Kubo and Supriyanto (2010) who stated that the forest cover of the GHSNP area steadily decreased, with the annual deforestation rate being around 1.2–2.3%. Furthermore, it appears that the interior forest of GHNSP was not much affected by deforestation since the topography of this area is mostly hilly and mountainous (Halimun-Salak, 2007).

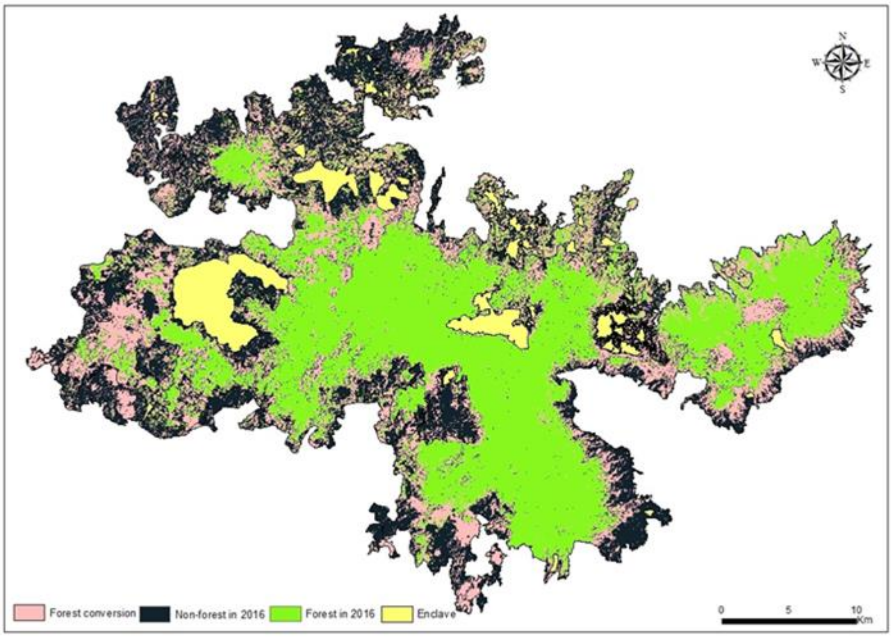


Figure 5.4 Map of forest conversion occurred in GHSNP for 2001 and 2016

There are various types of forest ecosystems in GHSNP which are identified according to the altitude of their locations: the Collin zone, which lies between 500 and 1,000 m; the submontane zone, which lies between 1,000 and 1,500 m; and the montane zone, which lies between 1,500 and 2,400 m (van Steenis, Hamzah, & Toha, 1972).

Between 2001 and 2016, substantial forest conversion occurred in GHSNP. Detailed forest conversion between 2001 and 2016 is presented in Table 5.3. Correspondingly, Table 5.4 presents the forest conversion based on the altitude for the period of 2001 and 2016.

Table 5.3 Areas and changes of forest and non-forest classes of GHSNP in 2001 and 2016

Class	2001		2016		Changes	
	ha	%	ha	%	ha	%
forest	76,580.6	72.8	49,500.4	47.0	27,079.6	35.3
non-forest	28,594.4	27.1	55,673.5	52.9	-27,079.1	-35.3
Total	105,175.0	100.0	105,174.0	100.0		

Table 5.4 Areas and changes of forest classes based on altitude in GHSNP between 2001 and 2016

area	2001		2016		Changes	
	Ha	%	Ha	%	Ha	%
Collin	24,889.9	37.8	13,706.1	29.2	11,183.8	44.9
Submontane	35,083.1	53.3	27,565.7	58.8	7,517.4	21.4
Montane	5,910.4	9.0	5,606.0	12.0	304.5	5.2
Total	65,883.4	100.0	46,877.8	100.0		

For the period 2001 to 2016, changes in forest land-cover occurred for 35.3% of GHSNP area (Table 5.3). Specifically, Collin forest that was about 24,889.9 ha in 2001, underwent a significant decrease of 44.9% by 2016 (Figure 5.5). For the same period of time, montane forest slightly decreased by 5.2 % (Table 5.4). Such results indicate that deforestation and forest encroachment mainly occurred in the Collin zone that lies between 500 to 1000 m.

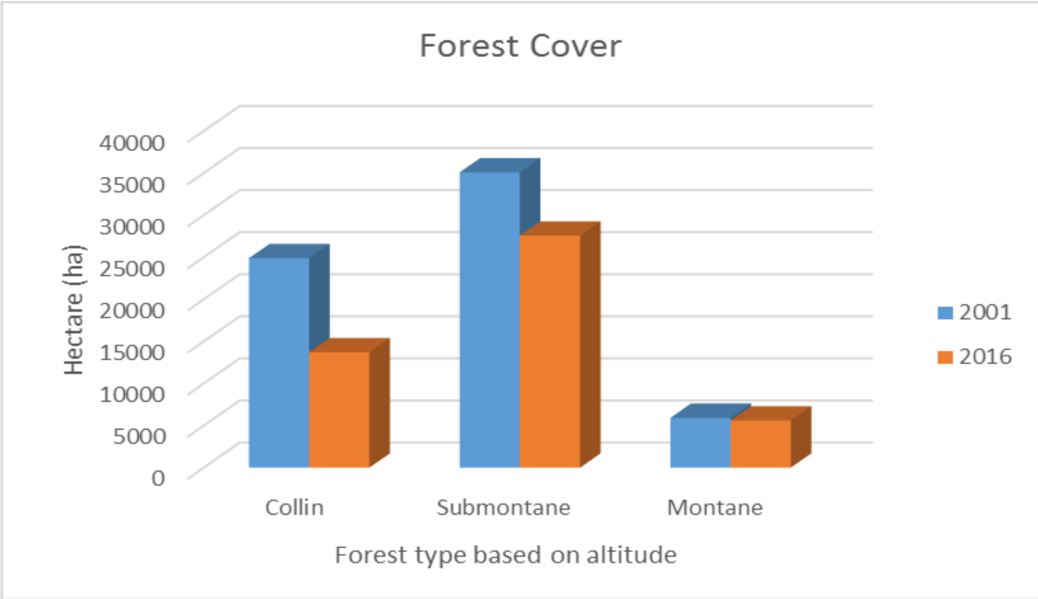


Figure 5.5 Areal of each forest type (Collin, submontane and montane zone) of GHSNP between 2001 and 2016

5.1.5 Changes in Forest Configuration between 2001 and 2016 in Gunung Halimun Salak National Park

The results of spatial patterns (2001 and 2016) of forest landscape indices are presented in Figure 5.6. In general, for several landscape metrics applied in this study, there were significant decreases for class area of 30.3% followed by other matrices that also underwent a relative decrease, namely mean patch size (MPS), patch size coefficient of variation (PSCoV), mean shape index (MSI), mean core area (MCA) and mean proximity index (MPI). On the contrary, number of patches (NP) and edge density (ED) increased substantially by 55.0 % and 25.1% respectively.

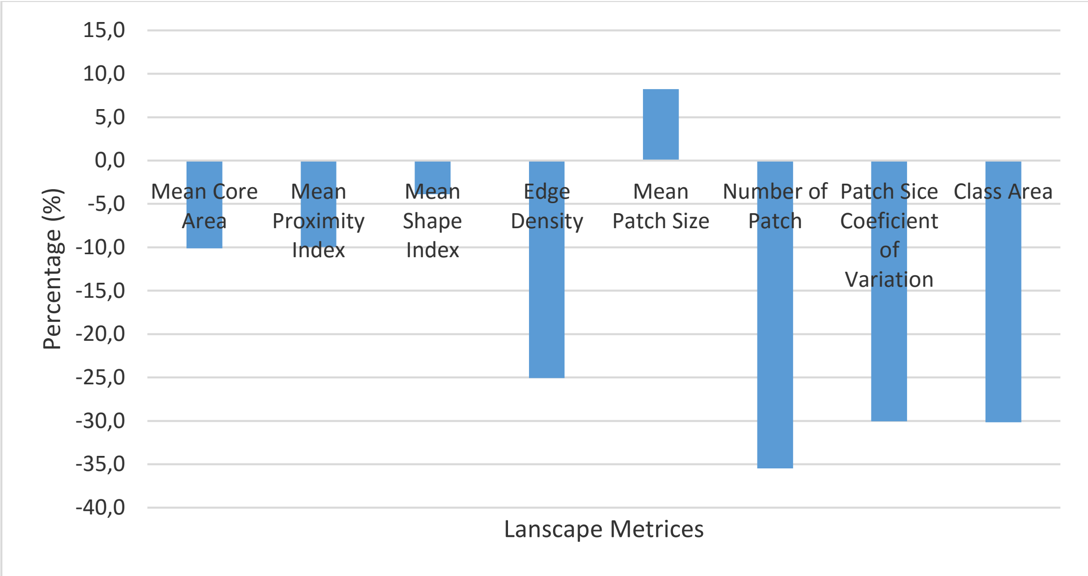


Figure 5.6 The percentage change of forest landscape indices between 2001 and 2016

This result also shows that the total area or amount of forest area habitat decreased massively by 30% for over fifteen years. Thus, this indicates that the forest area of GHSNP is more fragmented.

In contrast, the total number of forest patches has shown an increase of 5,519 patches (55%) during the 15-year period. The number of patches at landscape level has increased significantly during the period because of changes in the land cover distribution. It increased from 10,034 patches in 2001 to 15,553 patches in 2016 (Table 5.6). This indicates that forest in GHSNP has become more fragmented.

Table 5.5 Comparative analysis of forest landscape indices in GHSNP between 2001 and 2016

Landscape metrices	2001	2016	Change	Percentage
Mean Core Area	8.6	7.7	-0.9	-10.1
Mean Proximity Index	211662.9	190564.6	-21098.3	-10.0
Mean Shape Index	1.3	1.2	-0.1	-3.9
Edge Density	66.4	88.6	22.2	25.1
Mean Patch Size	4.7	4.3	0.4	8.2
Number of Patch	10034.0	15553.0	5519.0	55.0
Patch Size Coeficient of Variation	10856.7	7592.9	-3263.8	-30.1
Class Area	68010.6	47500.5	-20510.2	-30.2

Mean patch size (MPS) decreased by 8.2% during the 15 years. Again, this shows that the forest of GHSNP became more fragmented over the 15-year period. MPS, which declined over time, indicates a progressive reduction in the size of forests and along with the increase in number of patches, it thus confirms a higher rate of fragmentation.

Patch size coefficient of variation (PSCoV) is expected to decrease as patches become less variable or more similar in patch size over time (McGarigal & Marks, 1995). PSCoV in this study shows a significant decrease of approximately 30% from 10,857 ha in 2001 to 7,593 ha in 2000 (Table 5.6). This means landscapes with greater PSCoV are more heterogeneous while landscapes with lower PSCoV are more uniform (McGarigal & Marks, 1995).

Edge density at class level is directly related to the degree of spatial heterogeneity among classes (McGarigal & Marks, 1995), showing that the study area became more heterogeneous by 2016, since the edge density increased from 66.4 to 88.6 (by 25%) over the 15 years.

A slight decrease in mean size index (MSI) value was recorded in the forest of GHSNP (1.2 in 2016 to 1.2 in 2001). The decrease value in MSI indicates that the forest patches became simpler (Table 5.6). The mean shape index indicates that forest habitat patches are becoming less geometrically complex, although the amount of change in this index is not large. The mean shape index may also serve to quantify the location or position of disturbance features within natural vegetation patches. For example, the placement of a clear cut in the trend portion of a habitat patch may increase the perimeter in relation to the patch area. Consequently, the geometric complexity and shape index of the patch would actually increase (McGarigal & Marks, 1995).

Mean core area is the average core area of the patches of the corresponding forest type. This particular edge zone width has been chosen by several researchers investigating fragmentation issues within the forests (Mladenoff, White, Crow, & Pastor, 1994; Ripple et al., 1991) to represent the areas which are not influenced by the edge effect. Edge effects are a consequence of biotic and abiotic factors such as increased wind speed, solar insulation, and an altered soil moisture regime that combine to alter the environmental conditions along patch boundaries compared to the interior or core conditions. Based on the analysis, mean core area in GHSNP has decreased by 10.10%, indicating that fragmentation has occurred over the study period.

Mean proximity index (MPI) used for fragmentation analysis also demonstrated a decrease of 9.97%. This decrease in MPI also emphasises that, over time, forest patches become smaller and more isolated (Gustafson & Parker, 1992) as a consequence of fragmentation in those forest landscapes. The mean proximity index (MPI) decreased substantially during the last 15 years in GHSNP, suggesting that the isolation and degree of fragmentation for forest patches has increased as a result of deforestation. The proximity index was first introduced by Gustafson and Parker

(1992) and modified in the FRAGSTATS software (McGarigal & Marks, 1995) to distinguish between sparse distributions of small habitat patches (indicating a fragmented landscape) from a situation in which habitat patches form a complex cluster of larger patches (less fragmented). The metric equation for MPI is a product of both the average patch area of a class and the mean edge-to-edge distance between patches. Based on this formula, it follows that a larger MPI value indicates a class in which patches are distributed over larger, more contiguous areas that are located in closer proximity to patches of the same type (McGarigal et al., 2002). Thus, the decrease in MPI shown in Table 5.6 appears to confirm that areas of suitable habitat are being fragmented into smaller patches that are more dispersed or isolated from each other over time.

5.1.6 Changes in Forest Configuration Based on Altitude (Collin, submontane, and montane forest) in Gunung Halimun Salak National Park

The results of the variation of forest indices in the three zones of GHSNP forest are presented in Figure 5.7. In general, for several landscape metrics applied in this study, there were significant decreases for mean patch size (MPS) in the Collin and submontane forest zone, by 70.5% and 65.5% respectively. Other matrices that underwent a decrease in both zones are mean core area (MCA), class area (CA) and mean proximity index (MPI). Conversely, number of patches (NP) and edge density (ED) increased substantially in both zones with the highest ones occurring in the submontane zone: 175% for number of patch and 51.2% for edge density (EG). Patch size coefficient of variation also experienced an increase in submontane and Collin Zone by 77% and 74.8% respectively. Thus, in terms of fragmentation, those two zones are found to be much more fragmented compared to the montane zone which is the least fragmented forest in GHSNP.

	Mean Core Area	Mean Proximity Index	Mean Shape Index	Edge Density	Mean Patch Size	Number of Patch	Patch Size Coeficient of Variation	Class Area
Collin Zone	-68.6	-47.1	-1.6	41.2	-70.5	166.0	74.8	-21.4
Submontane Zone	-63.5	-0.1	0.0	51.2	-65.5	175.0	77.0	-5.2
Montane Zone	-27.3	-46.7	-3.0	-35.1	-25.3	-26.4	-19.3	-44.9

Lanscape Metrics

Figure 5.7 The percentage of change in landscape metrics of Collin, submontane, and montane forest between 2001 and 2016

In detail, the submontane zone experienced the highest values of edge density (ED). This value indicates that fragmentation had occurred in this area, while the montane zone, which experienced a decrease in edge density, is the least fragmented compared with the other two zones.

The results for mean patch size (MSI) revealed that change values for all the forest classes in GHSNP were negative, indicating that the average patch shape in all forest zones in the GHSNP area was irregular. Comparison of mean patch size (MPS) in three different zones demonstrates striking differences (Table 5.7). Collin zone forest shows a massive decline at 70.5%, which indicates that this zone is much more fragmented.

Notably, the decreased values of MCA in the Collin zone were slightly higher than in submontane zones, by 68.6% and 63.5% respectively (Figure 5.7). This decrease could be a sign of more edge effect in the Collin and submontane zones, as highlighted above. In addition, it also gave the cue about the process of fragmentation in Collin and submontane zones due to the changes in forest configuration.

Mean proximity index (MPI) is expected to decrease over time as habitat patches become smaller and more isolated (Gustafson & Parker, 1992). In regards to this, the Collin zone forest experienced a massive decrease in MPI at 47.1% as did the montane zone which also underwent a decrease at 46.7%. Similarly, the class area (CA) in the three zones shows a wide range of decrease from 5.2% to 44.9%. In contrast, the number of patches at the class level increased significantly in the submontane and Collin zone by 175% and 166% respectively over the past fifteen years. The increase in number of patches (NP) shows that the forest area has been subdivided into more patches and it could also be an indication that the forest in these two zones has become more fragmented.

Table 5.6 Comparative analysis of forest fragmentation in Collin zone, submontane, and montane area of GHSNP between 2001 and 2016

Landscape metrices	Collin Zone forest		Change (%)	Submontane Zone		Change (%)	Montane Zone		Change (%)
	2001	2016		2001	2016		2001	2016	
Mean Core Area	97.5	30.6	-66.9	307.9	112.4	-195.4	6.1	4.4	-1.7
Mean Proximity Index	92,503.3	48,901.2	-43,602.1	6,434.4	6,430.0	-4.4	5,719.0	3,049.6	-2,669.3
Mean Shape Index	1.3	1.3	0.0	1.6	1.6	0.0	1.3	1.3	0.0
Edge Density	16.1	22.7	6.6	1.6	2.5	0.8	52.8	34.3	-18.5
Mean Patch Size	67.7	20.0	-47.7	295.5	101.9	-193.6	4.3	3.2	-1.1
Number of Patch	518.0	1,378.0	860.0	20.0	55.0	35.0	5,778.0	4,253.0	-1,525.0
Patch Size Coefficient of Variation	1,812.1	3,166.8	1,354.8	282.3	499.7	217.4	2,689.4	2,169.7	-519.7
Class Area	35,083.1	27,565.7	-21.4	5,910.4	5,606.0	-304.5	24,889.9	13,706.1	-11,183.8

5.2 Effect of Road Infrastructure in Gunung Halimun Salak National Park

This subsection specifically aimed to explore fragmentation by roads and its disturbance effects on forest cover. Moreover, the analysis of habitat networks of each ecological profile constructed is discussed to understand the impact of road fragmentation and its disturbance effects on mammals and birds. The output of these are mean species abundance (MSA) of birds and mammals in three areas: Collin, submontane and montane zone; as well as class area (CA) and number of patches (NP) of each ecological profile.

Roads and other transport infrastructure interact with ecological processes by fragmenting and converting natural habitats, creating barriers and disturbance regimes, and disturbing trophic structures through road mortality and the invasion of allien species (Fahrig & Rytwinski, 2009). Road network in GHSNP is presented in Figure 5.8

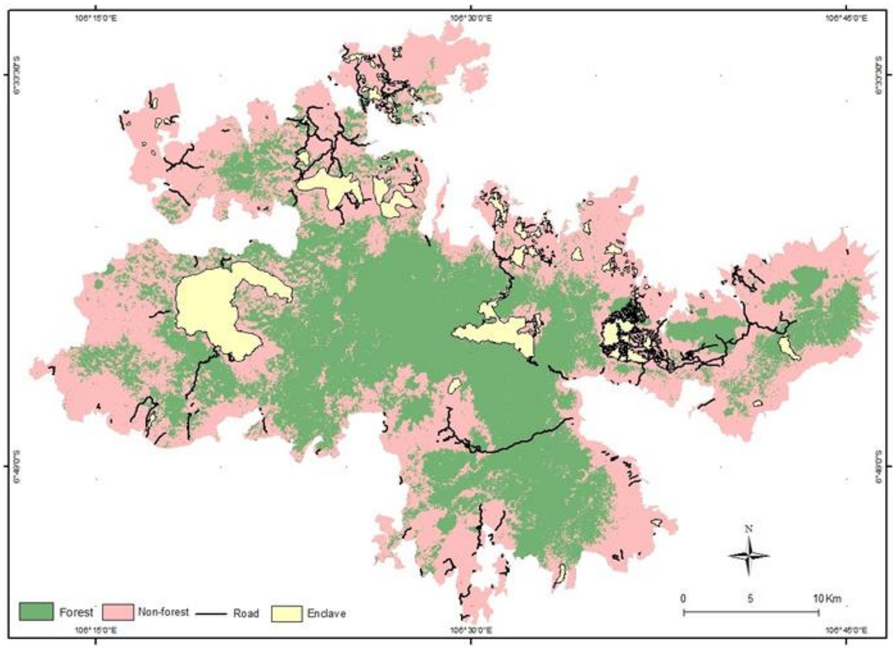


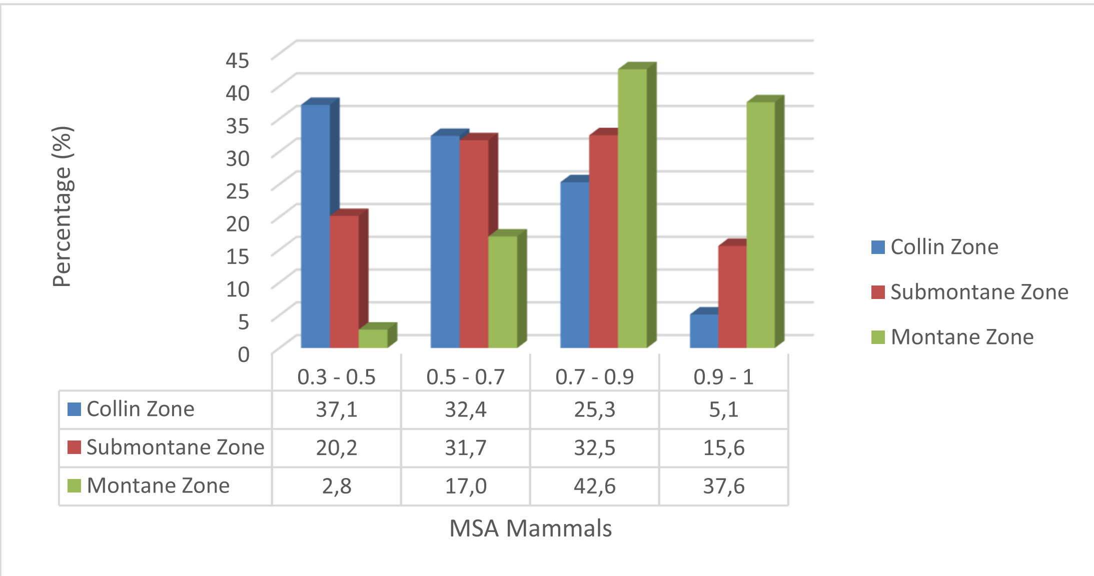
Figure 5.8 Road network in Gunung Halimun Salak National Park

The effects of transport infrastructure, for example the effect of the road network, on the landscape can be observed within a certain distance from a road, creating effect zones where environmental characteristics can be significantly distinguished from a control location (Forman & Deblinger, 2000). Several studies support the existence of such zones (Benítez-López et al., 2010; Biglin & Dupigny-Giroux, 2006; Bissonette & Rosa, 2009; Eigenbrod et al., 2009; Helldin & Seiler, 2003; Huijser & Bergers, 2000; Reijnen & Foppen, 1995).

5.2.1 Overall Impact of the Road Network

Figure 5.9 illustrates the predicted effects on mammals, and Figure 5.10 illustrates the predicted effects on birds. These figures show the proportion of different forest habitat types of high biodiversity value within the effect intensity zones, where the MSA in roadless area was close to 1 while a low MSA was considered as high road effect intensity (See also Appendix D). Both figures show that a larger proportion of Collin zone (37.1% for mammals and 4.9% for birds) and submontane zone forests (20.2% for mammals and 2.9% for birds) were situated in zones with a predicted MSA <0.5, —with the highest road disturbance intensity. By contrast, the proportions situated in areas with the highest road disturbance intensity were much lower for montane zone forest (2.8% for mammals and 0% for birds).

Concerning road effects on mammals, Figure 5.9 further shows that approximately 37.1 % of Collin zone forest and 20.2% of submontane forest in GHSNP are situated within the highly disturbed areas with MSA mammals < 0.5. This could be compared to MSA mammals between 0.9 and 1.0 of Collin and submontane forest that is 5.1% and 15.6% respectively. These results suggest that Collin forest is most affected by road disturbance effects so that they would support fewer mammals, compared to areas that were least qualitatively affected.



	0.3 - 0.5	0.5 - 0.7	0.7 - 0.9	0.9 - 1
Collin Zone	37,1	32,4	25,3	5,1
Submontane Zone	20,2	31,7	32,5	15,6
Montane Zone	2,8	17,0	42,6	37,6

Figure 5.9 Proportional share of forest habitat types (Collin, submontane, and montane zone) that are localized within each interval of road effect intensity, expressed as MSA of mammals MSA ranges between 0.35 and 1, where values close to 1 represents the MSA in a roadless area while a low MSA represents a high disturbance level.

Concerning road effects on birds, Figure 5.10 similarly shows that 4.91% and 2.9% of Collin and submontane forest were predicted to be situated within highly disturbed areas with $MSA_{birds} < 0.5$, compared with areas of montane forest. To summarize, the results indicated that the Collin and submontane zones were proportionally more exposed to road effects on birds than was montane forest, and that the Collin zone in general was predicted to be more detrimentally affected by road effects on birds than the submontane zone.

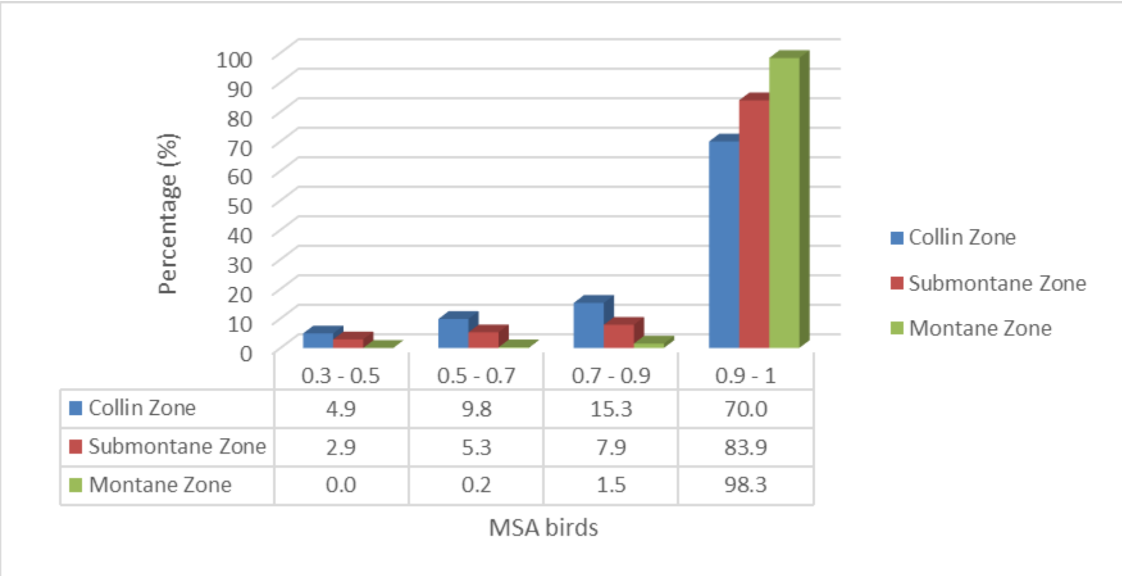


Figure 5.10 Proportional share of forest habitat types (Collin, submontane, and montane zone) that are localized within each interval of road disturbance intensity, expressed as MSA of birds. MSA ranges between 0.3 and 1, where values close to 1 represent the MSA in a roadless area while a low MSA represents a high disturbance level.

5.2.2 Effects of Fragmentation and Disturbance on Mammals and Bird

It is essential for ecological profiles to be constructed so as to include valuable biodiversity components that are targeted; such components should include species that are vulnerable to the development at stake and relevant for the study area (Mortberg, 1998). Moreover, they should capture properties that make those species vulnerable to road effects (Jaeger et al., 2005; Rytwinski & Fahrig, 2012). Such properties include habitat type and required habitat (Moreno-Sanchez et al., 2012) which were addressed in this study. Sensitivity to different types of disturbances, road mortality and other road-specific traits were not addressed in the current study, yet require further development. In this study, species representing birds and mammals in each ecological profile are sensitive and, as such, are keystone species in GHSNP. A keystone species is a species that has a disproportionately large effect on its environment relative to its abundance (V. H. Heywood & Watson, 1995). Therefore, the selected species are susceptible to habitat loss, fragmentation and disturbance. Below are the ecological profiles used for this study, which were constructed based on the data obtained from the literature review, as well as from Gunung Halimun Salak National Park office (see Table 5.8 and Table 5.9).

Table 5.7 Assumptions on home-range sizes, average movement distances, and habitat suitability of different habitat types for the five ecological profiles

Ecological profiles	Home-range size (ha)	Movement distance (m)
Forest-grassland mammals high area demands	1300–3500	3600–5900
Forest-grassland mammals medium area demands	250–540	1580–2525
Forest mammals below 1500 m area demands	2–25	141–500
Forest mammals above 1500 m area demands	15–47	387–685
Forest birds	254–400	1600–2000

Table 5.8 Model species that were selected in order to help define the ecological profiles for this study

No	Ecological profile	Scientific name	English name	References	Conservation status
1	Forest-grassland mammals	*Panthera pardus*	leopard	Malau (2013)	Vulnerable on the IUCN red list
	High area demands	*Cuon alpinus*	dhole	Grassman, Tewes, Silvy, and Kreetiyutanont (2005)	endangered on the IUCN red list
				Mustari, Setiawan, and Rinaldi (2016)	
2	Forest-grassland mammals	*Prionailurus bengalensis*	leopard cat	Gunawan, Chumsangsri, Lastini, and Gunawan (2007)	-
	medium area demands	*Sus scrofa*	wild boar	Graves (1984)	-
		Viverricula indica	small Indian civet	Shepherd (2008)	-
3	Forest mammals	*Nycticebus coucang*	slow loris	Nekaris, Blackham, and Nijman (2008)	Vulnerable on the IUCN red list
	below 1500 m area demands	*Tragulus javanicus*	lesser mouse-deer	Farida, Setyorini, and Sumaatmadja (2003)	
				van Schaik and Griffiths (1996)	
4	Forest mammals	*Trachypithecus auratus*	Javan langur	Karen Margaretha Kool (1989)	Vulnerable on the IUCN red list
	above 1500 m area demands	*Hylobates moloch*	silvery gibbon	Jatna Supriatna (2006)	endangered on the IUCN red list
				Rinaldi (2003)	
5	forest birds	*Nisaetus bartelsi*	javan hawk-eagle	Gjershaug (2006)	endangered on the IUCN red list
				(Prawiradilaga, 2006)	

After the ecological profiles were defined, another issue was how to best represent their integrity and response to change. The results of the analysis of fragmentation and disturbance took the form of two metrics describing the estimated change in total habitat area (CA) and in number of individual patches (NP), compared to a roadless habitat network. However, a multitude of landscape metrics have been suggested, of which many are redundant, while suggestions have been made on how to interpret them and how to select good indicators among them (Cushman et al., 2008; Schindler, Poirazidis, & Wrbka, 2008; Walz & Syrbe, 2013). Since normalization was carried out against the roadless habitat networks, an increase or a decrease in the metrics is considered as a percentage increase or decrease. The output, however, (Figure 5.11 and Figure 5.12) gave an indication of which types of species habitat were most impacted and on which groups of species the transport infrastructure effects had the most impact.

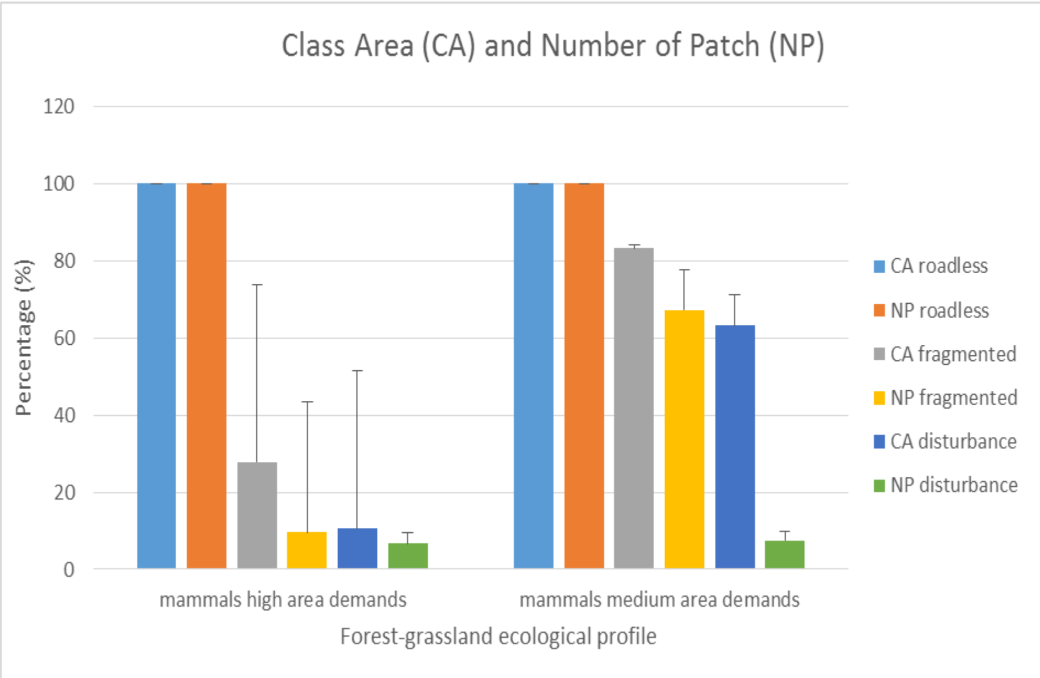


Figure 5.11 Forest–grassland ecological profiles: changes in total habitat area (CA) and number of patches (NP) in the fragmented and disturbed forest profiles compared to the roadless forest profiles. The bars represent the smaller home-range size for each profile, while the response range (vertical lines) represents the difference in results for the larger home-range size.

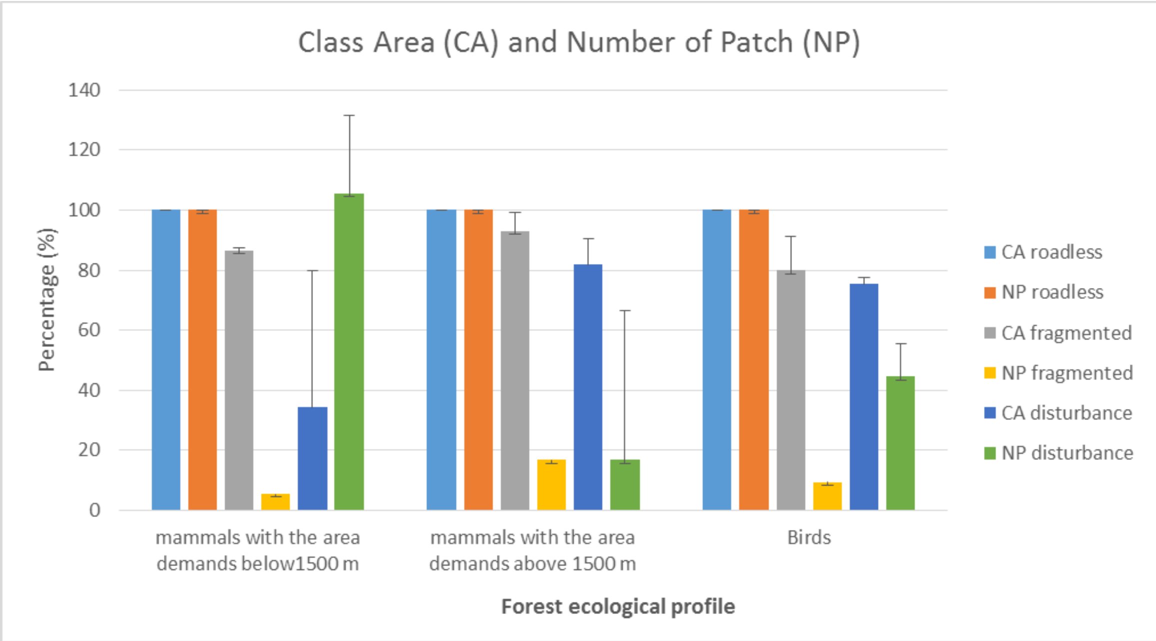


Figure 5.12 Forest ecological profiles: changes in total habitat area (CA) and number of patches (NP) in the fragmented and disturbed forest profiles compared to the roadless forest profiles. The bars represent the smaller home-range size for each profile, while the response range (vertical lines) represents the difference in results for the larger home-range size

In general, this analysis showed that the responses of the forest-grassland mammals (high area and medium area demands) and forest mammals (below 1500 m and above 1500 m) and bird profiles (forest area) were similar to each other, with CA and NP being reduced as a response to both effects, except for small mammals requiring forest area below 1500 m which underwent a relatively large increase in the number of patch for the disturbed area. Figure 5.11 and Figure 5.12 show how disturbance affects both metrics more than fragmentation does.

In particular, forest-grassland species with high area demands were most adversely impacted by road effects. For the ecological profiles representing forest and grassland habitat, compared to the roadless habitat networks, fragmentation reduced the amount of habitat in the study area (CA) by 26.2% – 72.2% for the profile with high area demands and 15.7% – 16.7% for the profile with medium area demands. Simultaneously, NP decreased by 56.6%– 90.4% for the high area demands profile, and it also decreased by 22.5% – 32.7% for the medium area demands profile (see Figure 5.11). In response to disturbance, CA decreased for the forest profiles by 48.5% –89.4%, 28.8% – 36.8% (high area demands and medium area demands respectively). At the same time, NP decreased in the high area demand profiles by 90.4% – 93.3% and 90.1% – 92.5% for medium area demand profile.

For the ecological profiles representing forest habitat, the results also showed that forest species living at altitude below 1500 m were adversely impacted by road effects (see Figure 5.12). Compared to the roadless habitat networks, fragmentation reduced the CA by 12.51% – 13.47% in the mammal profile for area below 1500 m, by 8.87% – 20.18% for forest bird profile and by 0.73% – 6.97% for mammals profile living in the area above 1500 m. The number of patches (NP) of fragmented area decreased by 79.63% – 94.64% for the mammal profile of forest demands at altitudes below 1500 m, decreased by 87.30% – 90.70% for the bird forest profile, and decreased by 0% – 83.33% for the eco-profile of above 1500 m area demands (Figure 5.13). In response to disturbance, CA decreased for eco-profile of area demands below 1500 m by 20.27% – 65.76%; by 22.44% – 24.75% and 9.58% – 18.11% for eco-profile of birds and eco-profile of above 1500 m area demands respectively. Similarly, number of patches for the disturbance area of birds experienced a decrease by 44.44% – 55.56% and for mammals requiring forest habitat of above 1500 m underwent a decrease by 33.33% – 83.33%. In contrast, NP increased by 5.63% – 31.48% for mammals requiring forest area below 1500 m.

In sum, for the forest eco-profile, disturbance reduced habitat amount (CA) in all eco-profiles and increased the number of patches (NP) in one eco-profiles, specifically mammals with area demands below 1500 m (Figure 5.12). For birds and mammals that occupy the area above 1500 m in the forest eco-profiles, NP declined as a response to disturbance. For the forest-grassland profiles, mammals with high area demands showed a proportionally greatest response to disturbance effect than fragmentation within each profile. Overall, it can be concluded that the larger home-ranges were more affected than the smaller one across all forest profiles. The results also suggest that fragmentation effects to be a greater concern for forest-grassland mammal's species with high area demands compared to forest mammals and bird species; and habitat loss to be a specific concern for forest-grassland species with high area demands and a general concern for forest mammals species requiring habitat below 1500 m.

Chapter 6

Discussion

This chapter presents a discussion of land cover change and forest cover change analysis; the forest fragmentation analysis; and also road fragmentation and impacts on mammals and birds in GHSNP between 2001 and 2016. Implications for management and conservation are also discussed in this chapter.

6.1 Land Cover Change and Forest Cover Change Analysis in Gunung Halimun Salak National Park

Land use and land cover changes as impacts of human activities contribute to land degradation through deforestation, removal of natural vegetation, and urban sprawl; unsustainable agricultural land use management practices, such as use and abuse of fertilisers, pesticides, and heavy machinery; and overgrazing, improper crop rotation, poor irrigation practices (Lambin, Geist, & Lepers, 2003).

Indonesia is one of the most important areas of tropical forests in the world. These forests are also of global importance because of their biodiversity and carbon sequestration capacities. Seventy-five percent of Indonesia's total land area of 191 million ha is classified as forest land, and the tropical rain forest component makes up the vast majority of forest cover (Hope, 2014). Total forest area is approximately 133.6 million ha and non-forest area is 54.3 million ha. Based on its function, about 15% of forest area is categorized as conservation forest, 22% as protection forest, 46% as production forest and 17% as convertible production forest. However, vast areas of rainforest are being lost every year due to high deforestation and thus, major land cover changes have occurred in Indonesia during the period of 2000 – 2010. As a result, the total area of forest decreased by more than two million hectares (1.7%) in that period, while the total area of shrubs, grassland and sparsely vegetated areas increased by 276,966.78 hectares or 3%. The total area of cropland increased by 1,810,485.16 ha or 2.7%. There was no change in wetlands and water bodies and artificial area during the period (Margono et al., 2012).

Overall, over the fifteen-year period agriculture, built-up areas, and plantations experienced an increase while forest, grassland, bare lands, and paddy field underwent a decrease. Kubo and Supriyanto (2010) suggest that the decreasing extent of forest in GHSNP was driven by illegal practices such as logging, agricultural expansion and mining that are conducted by local people. In contrast, Ilyas (2014) stated that the driving factor of the land cover changes from forest to non-forest was increased population.

In relation to the forest cover in GHSNP, the results of the study showed that 27,080.17 ha forest has changed and become another land cover type. In other words, 35.36% of forest cover has been converted to be mainly agricultural land or plantations. These forest conversion issues are mainly due to land encroachment and deforestation and, as a result, a gradual increase of settlement and nonforested land such as bush, upland, paddy field and settlement is now more predominant in the GHSNP area.

Deforestation is a term that was also used to describe a condition of degraded forests (Resosudarmo, 1996). It was from 2000 onwards that most international organizations defined and differentiated forest degradation and deforestation clearly (Contreras-Hermosilla, 2000; FAO, 2007). Direct causes of forest degradation and deforestation in Indonesia have been identified (Broich et al., 2011; Contreras-Hermosilla, 2000; W. D. Sunderlin, 1996; Verchot et al., 2010) mostly related to commercial plantation activities, transmigration programmess, infrastructure development, mining activities, commercial logging and natural forest fires.

Deforestation in GHSNP has been occurring since 1989. The highest rate of deforestation occurred primarily during the economic crisis due to the logging demand and not for plantation expansion (Acuña, Stimac, Sirad-Azwar, & Pasikki, 2008). In the case of settlement pressure, it is because long before the National Park was established, some inhabitants had been occupying villages within that area. Local people who live around GHSNP depended heavily on the forest to fulfil their basic needs. Collecting fuel wood and timber wood, hunting animals, using grass for feeding livestock, and forest encroachment for agricultural purposes are the highest pressures on the forest more recently. Consequently, these factors led to massive deforestation of the national park and conflict between local people's interests and conservation (Ahadi, Takao, & Sagala, 2013). Overall, it can be concluded that logging demand is the more dominant factor that leads to the deforestation in GHSNP because it has been occurring long before forest encroachment for agricultural purpose began.

Similar conditions are also found in the other conservation areas (Dewan & Yamaguchi, 2009). W. Sunderlin and Resosudarmo (1999) and W. D. Sunderlin et al. (2000) found that two-thirds of the people in forested areas were confronted with more conflicts during the economic crisis compared with their situation in the year before the crisis. The other finding is that small farmers are increasingly interested in clearing forests for perennial tree crops rather than raising food crops in shifting cultivation systems.

Overall, it appears that human transformations of land cover might likely to be a key driver of the loss of biodiversity and ecosystem services. Coupled with the effects of climate change, these pressures pose significant management and policy questions and thus, improved strategies to secure a more sustainable future is crucial (Anderson, 1990). It is vital, therefore, to know much more about how qualitative changes in land cover and land use impact upon biodiversity ecosystems services, as well as about quantitative changes in land cover and land use (Haines-Young, 2009).

6.2 Forest Fragmentation Analysis between 2001 and 2016

The results of the habitat patch analysis indicate that the forest landscape composition and configuration have been altered over the fifteen-year period as a result of land cover changes. The observed directional changes in the spatial metrics calculated for each time period has a correlation with the literature review and are discussed in section 2.4. These include a loss of total habitat area, increased number of patches, more edges, reduced core area, decreased mean patch size, and a wider, more discrete patch configuration over time. Patch size coefficient of variation and class area also showed major changes between 2001 and 2016 for forest classes as well as for each zone observed (Collin, submontane and montane zone). This pattern of change is consistent with fragmentation theory by Forman (1995), including phases of landscape evolution.

In a comparison of forest zones, that lie at different altitudes and were once connected to each other, the large decrease in MPS of forest classes in the Collin and submontane zone in GHSNP indicated that both areas were relatively more fragmented than similar forests in the montane zone. Despite of the fact that the number of patches in the montane zone was the highest among any zones in 2001 —which indicated the highest rate of fragmentation (McGarigal & Marks, 1995) in this zone—, for over a fifteen year, montane forest recorded slight decreases in MCA values representing a smaller number of forest interior declines compared with other forest classes. This could be attributed to a smaller edge effect on the shape of the patches of forest classes in the montane zone.

In addition, all three forest zones, namely Collin, montane and submontane forest in GHSNP acquired more irregular shapes and recorded MCA < 50% of MPS which suggested that the shape of the patches in these forests had been affected by edges created during clear cutting. This finding is also supported by Keenan and Kimmins (1993) that found patch area was reduced by 17% as a result of clear cutting. In relation to the mean core area, only two zones, namely the Collin and submontane zones, experienced a large decrease. This finding parallels that of Shinneman (1996) who stated that core, old-growth forest area in the Black Hills National Forest, USA had essentially been depleted by wood cutting. In other words, it was found to be more fragmented due to the decrease in the core area.

Reed et al. (1996) have pointed out that the changes in the shape, edge, density and diversity-related-measures reflect the impact of forest management in the landscape. The nature and amount of change detected by these different landscape statistics have significance for the ecology and management of forest landscapes. It could be assumed that the resulting landscape mosaics are strongly influenced by socioeconomic processes as well as by inherent environmental properties (Millington, Velez-Liendo, & Bradley, 2003). Reddy, Sreelekshmi, Jha, and Dadhwal (2013) also pointed out that the prime drivers of forest cover changes can be listed as shifting cultivation, along with increasing demand for agricultural land, mining, quarrying, forest fires, overgrazing, expansion of settlements, urbanisation, dam construction, illegal logging, and infrastructure development (Lele & Joshi, 2009; Reddy et al., 2013). Additionally, the magnitude of fragmentation was related to dominating forest classes, land use, and level of protection.

This is parallel to what has been occurring in GHSNP, in that ex-forest areas include plantations and agriculture, indicating that these areas have been widely planted. Forest conversion to plantation or other agricultural activity is supported by the fact that clear cutting occurred annually by about 1,473 ha or 1.3% of the GHSNP's total area (Ahadi et al., 2013).   As a result, about 22.000 ha (25%) of the natural forests in GHSNP were lost because of land use conversion and timber harvesting during 1998-2001 (Prasetyo, Setiawan, & Miura). These factors could be strong drivers for causing the fragmentation in GHSNP.

In summary, it is clear that the consequences of fragmentation are evident through changes in both composition, configuration and forest fragmentation in GHSNP. Such changes, however, are characterized by a reduction in the total amount of forest habitat and a change in the spatial characteristics and configuration of remaining patches.

6.3 Road Fragmentation and Disturbance in Gunung Halimun Salak National Park

In general, the results of sub chapter 5.3.2 could be seen in light of the empirical research, which found differences in the environmental characteristics between areas adjacent to and remote from the transport infrastructure (Benítez-López et al., 2010; Fahrig & Rytwinski, 2009; Forman & Deblinger, 2000).

Is it important to note that the road effects derived from Benítez-López et al. (2010) represent all possible causes, so that further separating of the causes and their severity are needed (recommended by e.g., Zetterberg et al. (2010)). Moreover, the spatial predictions of road effects were very coarse, and some of the assumptions can be questioned. For instance, the MSA predictions close to 1.0 imply conditions are equal to a control location with no road effects. This may not be very convincing, since other forms of impacts (e.g., from forest encroachment) were not taken into account. On the other hand, the adaptive capacity of mammals and birds should not be underestimated. In fact, areas exposed to moderate and low disturbance, such as areas in the vicinity of roads with lower daily traffic (down to 1000 and 100 vehicles per day for mammals and birds respectively) could still constitute suitable habitat for many species. In this study, even though home-range sizes were based on detailed empirical studies and were provided with a response range, there are still very high uncertainties in the expert judgements that supported the habitat suitability assumptions.

For all ecological profiles, the road effects imposed habitat loss that would increase extinction risks; and loss of connectivity, which would decrease the probability of colonisation. Both these processes are crucial since they determine the viability of populations in the landscape (Hanski & Gilpin, 1991; Holderegger & Di Giulio, 2010). Habitat loss would, in this study, particularly affect the forest-grassland mammal profile with high area demands. The probability of migration between patches is seen as negatively correlated to distance for many species (Hanski & Ovaskainen, 2003; Saura, Estreguil, Mouton, & Rodríguez-Freire, 2011).

Roads are an obvious threat to large-carnivore populations since many of those species are vulnerable to the effects of road-network expansion (Cardillo & McAlpine, 2004). Roads enable human access, thereby increasing disturbance, reducing available habitat, decreasing reproductive success, and increasing mortality rates (Maehr, 1997). The detrimental effect of roads can vary among large-carnivore species and among sex and age classes within species. For example, Florida panthers (*P.c.coryi*) tended to avoid crossing roads and contracted their home-ranges in the presence of roads, with females more road-averse than males (P. C. Cramer & Portier, 2001). Male panthers readily cross roads, however, resulting in relatively high mortality rates from vehicle collisions (Maehr, 1997). Gray wolves (*Canis lupus*) shift territorial boundaries to avoid heavily travelled roads (Thurber, Peterson, Drummer, & Thomasma, 1994), whereas female grizzly bears

(*Ursus arctos*) with cubs may be attracted to habitats adjacent to roads because males avoid those areas (Mattson, Knight, & Blanchard, 1987).

The degree of subdivision and total area of the eco-profile networks depends to a large degree on the radii used, with small radii producing more fragmented networks with small patches, and large radii producing more spatially aggregated networks with larger patches. Thus, a spatially aggregated network with large patches is likely to be relatively more fragmented and disturbed than an already highly fragmented network with small patches overlaid by one and the same fragmentation or disturbance data (Karlson, 2015). This is in line with these results that the constructed ecological profile with high area demand forest-grassland having larger radii is seen to be more spatially fragmented and disturbed compared to small radii with small patches. In this study, this relates to forest mammals for areas below 1500 m. Something to note is that, the need for the large activity ranges (radii) is due to the necessity of resources or resource scarcity. Thus, the response of the high demands forest profile was realistic, and could be interpreted as how the capacity of such habitats, which was reduced by transport infrastructure effects, supports forest mammals with high resource demands (Karlson & Mörtberg, 2015).

For the forest-grassland mammals, road effects had substantial impacts indeed, especially the disturbance effects, while the fragmentation effects did not massively impact the mammal profiles compared with the disturbance effect. For the mammal profiles with area demands below 1500 m and a 30–34 ha home-range, the effect of fragmentation was a small decrease in class area but an increase in number of patch, which was interpreted as loss of connectivity (Karlson, 2013).

For all profiles in this study, the decrease in CA was stronger in response to the disturbance effects compared with that of the fragmentation effects. This can be interpreted in different ways. Possibly these results, building on the empirical models for road effects on MSA, indicated that disturbance effects actually are more important than fragmentation effects for many ecological profiles, or the results depend on the current landscape pattern in the study area in combination with the habitat requirements of the species.

CA was reduced in all habitat networks of the ecological profiles in response to fragmentation and disturbance, whereas NP both increased and decreased. While a decrease in CA was interpreted as habitat loss, changes in NP have consequences for connectivity in the landscape. When NP decreases, it means that patches are either lost or that habitat is gained so that patches are merged. When NP increases, it means that patches are split into two or more smaller patches, or that new habitat patches are gained (Karlson & Mörtberg, 2015). Since habitat gain was not applicable in the current study, both changes in NP may imply a loss of connectivity, either as a loss of patches that could act as stepping stones, or as the splitting of previously homogenous habitat. These possibilities should be interpreted together with changes in CA.

It is clear that among all eco-profiles established, mammals having a high area demand are considered to be most impacted from fragmentation and disturbance effects. This is in line with other studies suggesting that species with high area demands, like large ungulates and carnivores, are especially sensitive to transport infrastructure effects like barriers to movement, disturbance and mortality (Davenport & Davenport, 2006; Fahrig, 2003; Jaeger et al., 2005; Rytwinski & Fahrig, 2012). In this study, mammals having a high area demand in GHSNP are represented by the Javan leopard (*Panthera pardus*). In Indonesia, this leopard can only be found on Java, particularly in conservation areas such as National Parks and Natural Reserves. Currently, it is still under threat from various human activities. Habitat loss and fragmentation, population decline of prey, hunting and trading activities are some of the threats for the Javan leopard. Such factors have also caused the extinction of the Javan tiger and another large cat species that once lived in Java (Wessing, 1995).

Moreover, it appears that species with home-ranges around 300 ha (medium area demand) were slightly more affected than those with the largest home-range, 600 ha. Furthermore, the low area demands profile with 30 ha home-range responded differently when the disturbance effect was applied, so that while CA decreased slightly, NP increased strongly. This resulted from a subdivision of this habitat network by the disturbance layer, with minor habitat loss but the splitting of many patches with subsequent loss of connectivity. The specific response of certain habitat networks can only be explained by the existing landscape pattern and habitat networks in combination with the location of the roads, since there is seldom linearity in the response of different metrics to landscape structure or change (Cushman et al., 2008).

In summary, this strengthens the motivation to further explore such potential effects when planning new road corridors, since sensitive habitat networks can be detected and the road effect on them could be avoided or mitigated, and this information could help prioritise mitigation and compensatory management activities and inform strategic planning, such as the development of national and regional transport strategies.

6.4 Implication for Management and Conservation

The process of landscape change as a result of fragmentation, which are caused primarily by human management activities (clear cutting, development of rail and road network, and plantations), has extensive implications for native plants, vertebrates and invertebrates, and particularly for the survival of threatened species. Several studies have reported that clear cutting or timber harvesting results in unbalanced removal of late succession forests (Johansson, Hjältén, Olsson, Dynesius, & Roberge, 2016; Tinker et al., 1998; Xu et al., 2015).

Those issues, however, have proven to be some of the most difficult and complex problems for Indonesia's conservation agencies since the downfall of President Suharto in 1996, and to date there is still no resolution to cope with such problems (Jepson, Jarvie, MacKinnon, & Monk, 2001; W. D. Sunderlin, 1999). Thus, understanding the effects of human disturbance is critical for effective management and conservation of endangered species.

The software Patch Analysis applied in the present study has been used in various regions of the world. The set of seven metrics quantified in this study are simple and proved to be useful for quantifying complex spatial processes and an effective means of monitoring in the GHSNP landscape. The approach of landscape level assessment and monitoring by select metrics has been recommended and adopted by many authors for different protected areas across the world (Asgarian, Amiri, & Sakieh, 2015; Jaafari, Sakieh, Shabani, Danehkar, & Nazarisamani, 2016; Leitão & Ahern, 2002; Riitters et al., 1995; Schindler, Poirazidis, & Wrbka, 2008). As the GHSNP area has experienced a massive conversion that lead to forest habitat reduction, changes in composition of forest patches and configuration must be monitored and the effects of land use and management interventions on landscape spatial pattern must be analysed. This knowledge, therefore, can be used to assess the progress in conservation efforts and to improve management decisions not only for national park landscapes, but also in other conservation area landscapes, particularly in Indonesia.

Much research has suggested that fragmentation and deforestation, such as has occurred in GHSNP, affects the persistence and abundance of wildlife resources. The persistence of many populations is linked to the number, size, and degree of isolation of forest patches, and a reduction in patch size that results in population declines for a number of species (Ambuel & Temple, 1983; Flockhart, Pichancourt, Norris, & Martin, 2015; Lynch & Whigham, 1984).

Van Dorp and Opdam (1987) reported a direct relationship between the number of bird species and forest area while Galli, Leck, and Forman (1976) stated that forest size is a primary determinant of the richness and size of bird assemblages. For bird species, those that are intolerant of fragmentation tend to be highly migratory, are forest interior specialists, build open nests, and nest on the ground (Whitcomb, 1981). The Javan hawk eagle, as a representative for the bird ecological profile, could be intolerant of fragmentation, since it is a highly migratory species and is probably more susceptible to forest destruction than all other Javan forest raptors (Thiollay & Meyburg, 1988). Thus, the status of this species is currently endangered according to IUCN due to its decreasing population (Jatna Supriatna, 2006).

Some large forest mammals such as tigers, elephants, and rhinoceroses tend to avoid forest boundaries. They prefer to occupy forest interiors, instead of forest edges, which allows them to avoid human activities that reduce cover and increase disturbance (including hunting at the forest edge and in the peripheral forest). Griffiths and Van Schaik (1993) found that large mammals in northern Sumatra, including elephants and tigers, moved away from areas of high human activity. B. V. Barnes, Zak, Denton, and Spurr (1997) and Theuerkauf, Ellenberg, Waitkuwait, and Mühlenberg (2001) found that elephant density in Gabon and elephant activity in the Ivory Coast decreased with proximity to roads and forestry operations. In relation to roads, it is important to note that it can have a major effect on large-carnivore mortality directly through overhunting, vehicle accidents, and poaching, and indirectly by providing greater hunting access that can result in reduced prey availability (Mattson et al., 1987; McLellan & Shackleton, 1988; Mech, Fritts, Radde, & Paul, 1988; Noss, 1993; Noss, Quigley, Hornocker, Merrill, & Paquet, 1996; Thiel, 1985). Finally, Woodroffe and Ginsberg (1998) and Revilla, Palomares, and Delibes (2001) also found a decrease in survival for carnivores and other mammals as a result of interactions with humans on park edges. For the past few years, the majority of the surrounding forests of the GHSNP have been converted for agricultural purposes. Thus, for the mammals of high area demand, these species could possibly be at greater risk when using these areas.

Leopards (*Panthera pardus*) are classified as endangered species according to the Convention on International Trade in Endangered Species (CITES) (Jayaprakash, PATIL, Kumar, Majumdar, & Shivaji, 2001). Three subspecies have become extinct since the 1950s due to human disturbances that include habitat loss, population fragmentation, and poaching (Mills, Jackson, & Gray, 1994; Nowell & Jackson, 1996; Seidensticker, 1986; Weber & Rabinowitz, 1996). Tigers, along with elephants and rhinoceroses, have large range requirements and require high-quality habitat composed of core forest. Large core areas are important habitat features for some mammals, especially forest carnivores (Tinker et al., 1998) and those living in an area with a low percentage of core forest will be at higher risk of mortality. As mean core area in GHSNP has experienced a relative decline over the 15 year study period (see Chapter 5.2.8) such animals will probably attempt to monopolize the core forest as much as possible while spending a higher percentage of time in an unfriendly matrix with increased risk of mortality (Kinnaird, Sanderson, O'Brien, Wibisono, & Woolmer, 2003).

Incorporation of an understanding of the ecological consequences of particular fragment spatial characteristics increases the environmental benefit of area management and planning. Some landscape architects and planners are increasingly using the principles of landscape ecology to preserve, restore and enhance biological diversity. For example, the restoration of three major wooded areas in New York's Central Park proposed by Andropogon Associates (L. A. Cramer, Kennedy, Krannich, & Quigley, 1993; Flather & Sauer, 1996) focused on maintaining large, intact forest patches within currently wooded areas of the park, reducing exotic plant invasion and sedimentation caused by disturbed forest edges and connecting these patches to enhance movement of birds and mammals. Such strong interconnetivity among patches is crucially important since weak interconnectivity among patches of similar forest classes, particularly in GHSNP could affect movement and dispersal of faunal species. In addition, clear cuts and roads block the movement of some species, resulting in population fragmentation and increased competition for resources in remaining forest resources (Lovejoy et al., 1986; Noss, 1993). To address these issues, Andropogon Associates devised a habitat corridor. In the context of ecological studies of habitat fragmentation, the term *corridor* generally refers to a linear landscape element composed of native vegetation that links patches of similar, native vegetation (Beier & Noss, 1998; Bennett, 1990; Brooks, 2003; Rosenberg, Noon, & Meslow, 1997a).

Landscape corridors play an important role in ecological dynamics within and between habitats (Bennett, 1990; Forman & Godron, 1986; Saunders et al., 1991; Taylor, Fahrig, Henein, & Merriam, 1993). The preservation of vegetated corridors among otherwise isolated habitat remnants is predicted to moderate the negative effects of habitat fragmentation by maintaining landscape connectivity (Diamond, 1975; Forman & Godron, 1981; Harris & Scheck, 1991; Lindenmayer & Nix, 1993; Noss, 1987). These suggest that the Mount Halimun-Salak corridor should be preserved by restoring the forest between Halimun-Salak to improve its function to the connect gene pools of the two populations, especially the populations of large mammals that require a large area in the Halimun and Salak areas.

Ideally, for conservation of species requiring large areas, it is important to prevent road construction wherever possible since roads decrease the survival and reproductive success of large mammals and in fact, protected areas seem to cease functioning as source populations when road access is introduced. Furthermore, closing unnecessary roads, and regulating access to roads through sensitive areas, particularly in areas supporting source populations, is also crucial to minimize the road effect on large mammals since they often use roads as travel corridors. Tigers displaced from their prey not only lose a valuable food resource, but they are at greater risk of being poached, or killed in traffic collisions because they may be on the road more often (Kerley et al., 2002).

Not only large mammals, but also small mammals with small area demands, such as the Javan gibbon, are also affected by road disturbances due to the increased number of patches that that results in the loss of connectivity; therefore, ex-situ management and intrusive management is vital for their conservation. Asquith (2001) stressed the dichotomy between strategies for Javan gibbon conservation suggested by geneticists, zoo biologists and captive-breeding specialist on the one hand and field biologists on the other. The first group focused largely on active management of small populations—including genetic supplementation, demographic management, and for small populations, rapid habitat expansion, translocation, and captive propagation (Jatna Supriatna & Manullang, 1999; J Supriatna et al., 1994)—whereas the second group has repeatedly argued that expansion of the protected area network, improved protection and further research and monitoring are the most urgent actions required (Asquith, 2001; Karen M Kool, 1992; Nijman & Van Balen, 1998; Sözer & Nijman, 1995).

In summary, to conserve mammal populations in tropical landscapes, such as those in the GHSNP, management must concentrate on conserving the remaining forest habitat within the park and reducing the threats to mammals in peripheral forest areas. Managing human activities inside and outside the park will also be crucial to mitigating threats (Revilla et al., 2001). Enforcement of existing laws prohibiting wildlife hunting and timber theft within the park would reduce harassment of mammals and reduce other forms of habitat deterioration. Managers may also need to consider restoration of lost or heavily disturbed forest and of the forest edge (Kinnaird et al., 2003). Laurance et al. (2000) stressed that it is not sufficient to just conserve isolated, fragmented reserves and that the intervening matrix must also be preserved, reforested, and probably reconfigured. In this way, the risks of mortality for wide-ranging mammals would decline and the amount of friendly habitat would more than double (Kinnaird et al., 2003).

Chapter 7
Conclusion and Recommendations

The conclusions of this thesis and the recommendations for future research based on the results will be drawn in this final chapter.

7.1 Conclusions

Based on the results obtained and their analyses, the following conclusions are drawn:

a) Based on the land cover analysis result, over the fifteen-year study period, class areas of agriculture and built-up increased, while forest and grassland decreased. Forest and Agriculture experienced the biggest decrease and increase by 35.36% and 463.74% respectively.

b) Among the three forest zones in GHSNP, the Collin zone experienced the most significant decrease of land cover, by 44.93%, which indicates that the majority of deforestation and forest encroachment occurred majorly in the submontane forest that lies in altitudes between 500–1000 m.

c) Based on forest landscape metrics analysis, over the fifteen-year study period, there was a significant decrease of approximately 30% for forest class area, followed by other metrics that also underwent a decrease, namely mean patch size (MPS), patch size coefficient of variation (PSCoV), mean shape index (MSI), mean core area (MCA) and mean proximity index (MPI). Conversely, number of patch (NP) and edge density (ED) increased substantially by 35.49% and 25.07% respectively.

d) Based on the landscape metrics analysis on the three forest zones within GHSNP, there was a significant decrease in mean patch size (MPS) in the Collin and submontane forest zones, of 70.47% and 65.51% respectively. On the contrary, the number of patches (NP) and edge density (ED) increased substantially in both these zones, with the highest increase occurring in the submontane zone: 175% for NP and 51.22% for ED. Thus, the two zones were found to be much more fragmented compared to the montane zone which was found to be least fragmented forests in GHSNP.

e) Among the three forest zones, a larger proportion of Collin zone forest (37.14% for mammals and 4.91% for birds) and submontane zone forest (20.17% for mammals and 2.91% for birds) were situated in zones with the highest road disturbance intensity with a predicted Mean Species Abundance (MSA) < 0.5,. Thus, it suggested that Collin zone forest is most affected by road disturbance effects. In addition, Collin and submontane zone were proportionally more exposed to road effects on mammals and birds, than montane forest.

f) Regarding the effects of fragmentation and disturbance on mammals and birds, the responses of the forest-grassland mammals (high area and medium area demands), forest mammals (area below 1500 m and above 1500 m) and bird profiles (forest area) were similar to each other, with class area (CA) and number of patch (NP) being reduced as a response to both effects, except for small mammals requiring forest area that is below 1500 m, which underwent an increase in the number of patches for the disturbed area.

g) GIS-based ecological assessment has proved effective in generating baseline environmental information and coarse predictions on the possible consequences of infrastructure development. Thus, the result from this approach could be a reference in formulating appropriate conservation measures, as well as for developing mitigation and management strategies in GHSNP.

7.2 Recommendations

In view of the conclusions drawn, the following suggestions for future research are made:

a) A follow-up study with finer resolution satellite imagery should be carried out in order to achieve mapping at a more detailed level, providing a long-term impression of land cover changes and enhancing the inventory of land resources in GHSNP for planning and monitoring.

b) Exploration of other classification approaches that might yield better results, taking into consideration the complexity of land cover types within GHSNP should be carried out.

c) In the present analysis, a selection of metrics was used that appeared to provide the best or optimal interpretation of the changing landscape structure. A complete and thorough test of the selected or potential landscape metrics (and others as they are developed or become available) should be conducted to reveal those metrics most sensitive to the type of change encountered in the landscapes under study.

d) More thorough habitat suitability data of each ecological profile should be gathered as well as field measurement of mean species abundance (MSA) of each ecological profile. Such data could then be compared with the MSAs of mammals and birds by Benítez-López et al. (2010) or other models of road effects.

e) There is also a general (but strong) need of verification and calibration of relevant spatial ecological models and methods, specifically of dispersal and movement models, species distribution models and habitat suitability models.

f) Modelling migration corridors for prioritised animal species could be a good research topic as the prediction of which group of animals is most likely to be impacted is now known.

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Appendix A

Coordinates Data

A.1 Coordinates used for accuracy assessment of land cover

Number	Coordinate System		Information
	x	y	
1	654,576.1032	9,273,382.9224	Plantations
2	666,163.1590	9,248,940.0000	Agriculture
3	651,380.3867	9,257,820.0000	Paddy field
4	647,321.5306	9,266,940.0000	Forest
5	638,166.4975	9,258,380.0000	Forest
6	668,445.7762	9,262,590.0000	Forest
7	678,224.7890	9,260,430.0000	Agriculture
8	655,287.4483	9,269,680.0000	Plantations
9	642,542.6727	9,269,590.0000	Agriculture
10	657,956.0134	9,272,610.0000	Paddy field
11	648,874.1356	9,266,140.0000	Forest
12	641,157.2297	9,250,960.0000	Shrubs and bushes
13	674,025.20	9,265,109.76	Waterbody
14	653,505.1315	9,255,800.0000	Forest
15	683,836.6835	9,256,517.2119	Open space
16	666,862.5881	9,244,730.0000	Forest
17	680,836.4099	9,252,000.7449	Shrubs and bushes
18	682,107.90	9,254,473.22	Residential Area
19	655,586.5380	9,252,620.0000	Forest
20	642,368.3911	9,261,030.0000	Paddy field
21	653,323.8796	9,257,130.0000	Forest
22	653,437.9143	9,262,560.0000	Shrubs and bushes
23	674,048.55	9,265,470.36	Water body
24	664,360.3082	9,249,430.0000	Paddy field
25	671,104.1395	9,251,990.0000	Forest
26	645,195.0282	9,252,810.0000	Forest
27	671,129.2768	9,259,540.0000	Forest
28	675,744.9108	9,260,230.0000	Shrubs and bushes
29	677,619.4859	9,260,390.0000	Agriculture
30	641,615.9514	9,253,730.0000	Shrubs and bushes
31	658,159.6049	9,274,690.0000	Plantations
32	648,444.0241	9,252,730.0000	Paddy field
33	653,832.2331	9,272,721.6723	Plantations
34	659,711.4790	9,249,240.0000	Forest
35	671,265.2008	9,255,680.0000	Shrubs and bushes
36	689,468.5024	9,257,153.3005	Open space

37	654,432.0223	9,265,640.0000	Agriculture
38	659,013.5516	9,267,590.0000	Agriculture
39	689,116.7382	9,257,144.0400	Open space
40	663,997.64	9,249,880.17	Residential Area
41	683,737.4976	9,254,782.3355	Open space
42	689,241.0926	9,257,199.6026	Open space
43	655,185.7715	9,272,850.0000	Plantations
44	674,080.30	9,265,419.56	Water body
45	651,475.6138	9,257,490.0000	Paddy field
46	655,392.6059	9,259,980.0000	Forest
47	693,089.82	9,261,215.85	Waterbody
48	642,699.4781	9,248,440.0000	Paddy field
49	654,588.4675	9,271,810.0394	Plantations
50	685,119.5514	9,255,576.8809	Open space
51	637,931.8584	9,257,510.0000	Agriculture
52	652,529.4979	9,269,720.0000	Forest
53	673,961.77	9,265,588.90	Water body
54	661,706.2739	9,267,410.0000	Agriculture
55	646,426.8544	9,253,450.0000	Forest
56	658,820.1103	9,250,600.0000	Paddy field
57	663,381.1644	9,265,150.0000	Agriculture
58	657,317.5264	9,250,280.0000	Shrubs and bushes
59	652,901.2658	9,268,260.0000	Agriculture
60	654,336.3902	9,273,513.0977	Plantations
61	681,979.84	9,254,571.65	Residential Area
62	643,785.9993	9,256,280.0000	Forest
63	645552.1526	9268057.363	Plantations
64	653,140.2331	9,273,116.1557	Plantations
65	670,120.4222	9,256,691.7175	Plantations
66	643,174.9305	9,253,270.0000	Shrubs and bushes
67	670,408.3098	9,255,750.0000	Shrubs and bushes
68	639,051.5649	9,249,620.0000	Agriculture
69	678,690.6136	9,252,520.4299	Shrubs and bushes
70	642,198.6058	9,252,860.0000	Agriculture
71	657,651.1533	9,262,170.0000	Forest
72	646,843.8495	9,253,780.0000	Forest
73	657,065.4537	9,274,190.0000	Paddy field
74	662151.6623	9242223.812	Shrubs and bushes
75	658,261.3554	9,254,450.0000	Forest
76	670,020.4461	9,256,908.0055	Plantations
77	649,210.7393	9,253,710.0000	Paddy field
78	664,837.4290	9,245,740.0000	Forest
79	651,203.4029	9,266,720.0000	Forest
80	667,330.6395	9,242,430.0000	Paddy field
81	654,923.0568	9,269,460.0000	Agriculture
82	651,691.1888	9,256,340.0000	Forest

83	685,178.8182	9,255,413.8972	Open space
84	653,668.4166	9,263,360.0000	Shrubs and bushes
85	641,811.0242	9,252,750.0000	Agriculture
86	655,471.93	9,272,544.87	Waterbody
87	641,970.9020	9,271,980.0000	Agriculture
88	646,151.9792	9,252,590.0000	Forest
89	653,745.3304	9,272,380.0000	Plantations
90	675,900.2307	9,261,650.0000	Agriculture
91	674,067.54	9,265,401.86	Waterbody
92	675547.5228	9262720.322	Shrubs and bushes
93	667,771.8982	9,243,030.0000	Shrubs and bushes
94	683,834.3023	9,255,813.0220	Open space
95	660200.1541	9240585.419	Shrubs and bushes
96	679,344.4109	9,252,218.6773	Shrubs and bushes
97	654,359.1934	9,273,780.0000	Plantations
98	657,981.3150	9,253,310.0000	Forest
99	645,076.9382	9,255,050.0000	Forest
100	682,167.69	9,254,574.29	Residential Area
101	654,949.3860	9,265,430.0000	Forest
102	643,624.1366	9,250,450.0000	Forest
103	654,229.3502	9,268,450.0000	Agriculture
104	656,432.3622	9,272,200.0000	Paddy field
105	677845.2099	9255178.239	Shrubs and bushes
106	689,260.5395	9,257,300.9383	Open space
107	673,953.30	9,265,389.93	water body
108	641,731.4335	9,258,000.0000	Forest
109	647,027.6246	9,261,550.0000	Shrubs and bushes
110	682,973.4787	9,255,320.6314	Open space
111	681,871.44	9,254,482.20	Residential Area
112	676,750.6663	9,258,190.0000	Agriculture
113	654,900.6953	9,265,200.0000	Agriculture
114	639,104.0170	9,258,320.0000	Shrubs and bushes
115	681,971.39	9,254,537.63	Residential Area
116	668,587.1435	9,251,150.0000	Forest
117	641,586.6848	9,252,900.0000	Agriculture
118	664,063.25	9,249,887.58	Residential Area
119	648,500.4710	9,251,610.0000	Forest
120	668,248.7822	9,243,160.0000	Shrubs and bushes
121	684,109.5025	9,254,136.4863	Open space
122	674,725.3013	9,257,750.0000	Forest
123	642,062.8868	9,271,440.0000	Agriculture
124	649,671.8537	9,250,490.0000	Paddy field
125	636,744.8726	9,250,420.0000	Shrubs and bushes
126	673,201.3157	9,240,970.0000	Forest
127	643,046.3410	9,272,150.0000	Plantations
128	661903.7101	9268528.429	Shrubs and bushes

129	663,651.0060	9,266,550.0000	Agriculture
130	673,995.63	9,265,662.98	Water body
131	642,911.7878	9,255,090.0000	Forest
132	649,084.9276	9,269,880.0000	Paddy field
133	644591.8161	9266212.425	Plantations
134	682,158.72	9,254,540.80	Residential Area
135	642,596.2604	9,255,050.0000	Forest
136	676,910.3023	9,243,510.0000	Agriculture
137	642,139.7044	9,257,380.0000	Paddy field
138	674,042.20	9,265,243.88	Water body
139	664,975.3556	9,251,990.0000	Paddy field
140	675,213.5987	9,261,370.0000	Agriculture
141	677308.6308	9256590.885	Residential Area
142	639,985.8970	9,251,170.0000	Agriculture
143	671,504.4850	9,248,230.0000	Forest
144	673,919.37	9,265,342.59	Waterbody
145	681,979.84	9,254,602.34	Residential Area
146	650,198.0244	9,258,250.0000	Paddy field
147	638,212.5932	9,252,030.0000	Agriculture
148	657,928.7637	9,252,080.0000	Shrubs and bushes
149	651,638.9922	9,270,730.0000	Paddy field
150	675,400.7917	9,261,080.0000	Agriculture

Appendix B

Data of Landscape Indices of Each Ecological Profiles

B.1 Class area (CA) and number of patch (NP) of each ecological profile

Number	Habitat/Ecological Profile	Habitat Network	Landscape indices			
			Class Area (ha)	Class Area (%)	Number of Patch	Number of Patch (%)
1	Forest-Grassland/mammals High area demands	**Roadless Area**				
		Habitat Network 1	7639.40	100	30	100
		Habitat Network 2	1054.71	100	21	100
		Fragmented Area				
		Habitat Network 1	5,638.25	73.80	13	43.33
		Habitat Network 2	292.95	27.78	2	9.52
		Disturbed Area				
		Habitat Network 1	3,933.97	51.50	2	6.67
		Habitat Network 2	110.86	10.51	2	9.52
2	Forest-grassland/mammals - medium area demands	**Roadless Area**				
		Habitat Network 1	16281.58	100	61	100
		Habitat Network 2	12772.75	100	40	100
		Fragmented Area				
		Habitat Network 1	13720.49	84.27	41	67.21
		Habitat Network 2	10639.19	83.30	31	77.50
		Disturbed Area				
		Habitat Network 1	11583.40	71.14	6	9.84
		Habitat Network 2	8071.19	63.19	3	7.50
3	Forest mammals/below 1500 m area demands	**Roadless Area**				
		Habitat Network 1	16966.42	100	162	100
		Habitat Network 2	17564.46	100	224	100
		Fragmented Area				
		Habitat Network 1	14844.46	87.49	213	131.48
		Habitat Network 2	15199.29	86.53	171	105.62
		Disturbed Area				
		Habitat Network 1	13527.34	79.73	33	20.37
		Habitat Network 2	6014.17	34.24	12	5.36

Number	Habitat/Ecological Profile	Habitat Network	Landscape indices			
			Class Area (ha)	Class Area (%)	Number of Patch	Number of Patch (%)
4	Forest mammals/above 1500 m area demands	**Roadless Area**				
		Habitat Network 1	3351.44	100	6	100
		Habitat Network 2	2439.11	100	6	100
		Fragmented Area				
		Habitat Network 1	3327.01	99.27	6	100.00
		Habitat Network 2	2269.15	93.03	1	16.67
		Disturbed Area				
		Habitat Network 1	3030.26	90.42	4	66.67
		Habitat Network 2	1997.39	81.89	1	16.67
5	Forest/bird	**Roadless Area**				
		Habitat Network 1	16198.51	100	63	100
		Habitat Network 2	14246.77	100	43	100
		Fragmented Area				
		Habitat Network 1	14775.97	91.22	35	12.69
		Habitat Network 2	11371.13	79.82	46	9.30
		Disturbed Area				
		Habitat Network 1	12563.30	77.56	8	55.56
		Habitat Network 2	10721.18	75.25	6	44.44

Appendix C

Keystone Species in Gunung Halimun Salak National Park

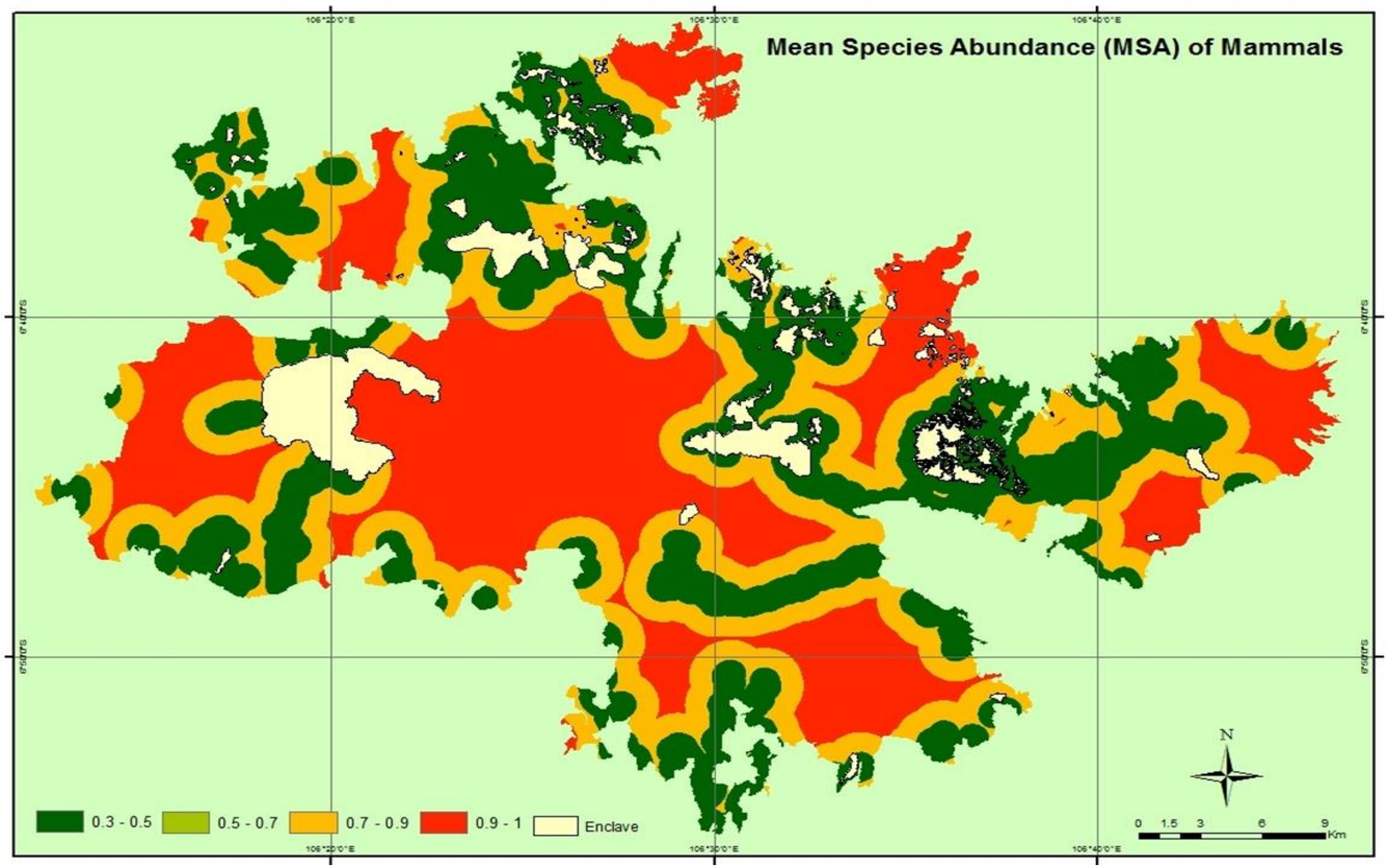
C.1 Some of keystone species in GHSNP that represent each ecological profile

(a): *Panthera pardus*, Javan leopard; (b) *Prionailurus bengalensis*, Leopard cat; (c) *Nisaetus bartelsi*, Javan hawk-eagle; (d) *Trachypithecus auratus*, Javan langur (B. T. N. G. H. Salak, 2016)

Appendix D

MAPS

D.1 Mean Species Abundance (MSA) of mammals in Gunung Halimun Salak National Park



D.2 Mean Species Abundance (MSA) of Birds in Gunung Halimun Salak National Park