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Ian Joseph Berdie
University of Miami, berdiei@gmail.com

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UNIVERSITY OF MIAMI

ASSESSING CANOPY COVER REQUIREMENTS OF STORM'S STORK (*CICONIA STORMI*) AT MULTIPLE SCALES

By

Ian J. Berdie

A THESIS

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Ian J. Berdie

Approved:

Dr. Doug Fuller Professor of Geography and Regional Studies	Dr. Terri A. Scandura Dean of the Graduate School
Dr. Rinku Roy Chowdhury Professor of Geography and Regional Studies	Dr. Mathew Potts Professor of Biology

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Much conservation work focuses on individual species, partly because of the perception that wildlife species are effective symbols for raising funds and drawing awareness to environmental causes. However, for species-based studies to aid conservation efforts, the biological and ecological needs of species need to be addressed in a way that informs decisions and provides concrete recommendations for land managers. This thesis addresses the forest cover needs of *Ciconia stormi*, a rare and understudied bird species that inhabits the islands of Borneo and Sumatra and parts of peninsular Malaysia. Levels of forest canopy cover associated with areas inhabited by *Ciconia stormi* are identified at multiple spatial resolutions using a 500m MODIS soft classification product, 30m Landsat data, and hemispherical photographs. Important threshold values of 75 percent tree cover was identified at the regional scale, and 85 percent at foraging sites. There has been severe forest disturbance in regions inhabited by *Ciconia stormi* between 1993 and 2004, indicating the species may be somewhat tolerant to disturbance. Areas having been logged at least 20 years before present average over 85 percent canopy cover and have few large gaps, indicating that these forests may be suitable habitat for the species.

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Chapter I

Introduction

This thesis addresses the habitat needs of Storm's stork (*Ciconia stormi*), a rare and understudied bird species that inhabits the islands of Borneo, Sumatra and peninsular Malaysia. The goal is to identify the distribution of *Ciconia stormi* on Borneo and examine the amount of forest canopy cover associated with areas inhabited by the species. Also examined are the forest canopy conditions at various times after logging in order to better understand the impacts of logging activities on the species' habitat. Finally levels of deforestation and habitat fragmentation that have occurred within areas of potential *Ciconia stormi* presence are analyzed to better assess the stability of the habitat within areas utilized by the species.

The approach is to demarcate regions of species presence based on recorded sightings and the various levels of imprecision associated with those records. Areas of *Ciconia stormi* presence have been identified with sighting records and assigned a range of values indicating the level of confidence that an individual was present at any given location within that area, based on the point-radius method for geo-referencing species occurrences (Wieczorek et al., 2004). The map developed in this thesis improves upon commonly binary polygons of presence/non-presence in that the areas identified as presence are characterized by a continuous range of values and can be examined at various levels of confidence. Environmental variables within the potential distribution are then analyzed using a variety of remote sensing and GIS techniques to identify thresholds of percent canopy cover and forest patch size where *Ciconia stormi* is seen.

1.1 Conservation Biology Paradigms

In its early manifestations, conservation biology evolved from a preservationist mindset, to a more economically driven resource utilization premise (Meffe and Carroll, 1997). Those ideas gave way to an ecologically based approach that viewed nature as a complicated network of interdependent processes and components that naturally approach a stable equilibrium state. Recently, this equilibrium paradigm has shifted towards a dynamic non-equilibrium ecological perspective that accepts disturbance and change as mechanisms that have an entrenched role in ecosystem function (Botkin, 1990). Today, applied conservation is a mixture of these different paradigms, with various organizations adopting philosophical positions based on their own goals and beliefs. The most effective strategy for biodiversity conservation is most likely a balanced approach that acknowledges each of the different philosophies (Meffe and Carroll, 1997; Sodhi and Brook, 2006). While the ecological approach is the most suitable for managing environmental processes, socio-economic and cultural needs of people need to be accounted for when making land-management decisions (Meffe and Carroll, 1997). This research approaches conservation of *Ciconia stormi* and Borneo's lowland forests from the perspective that some level of resource use is inevitable in the region, and that the careful management of forest resources should allow for the persistence of the species.

1.2 Driving Forces of Land Cover Change in the Region

The island of Borneo is experiencing some of the highest rates of deforestation in the world (Laurance, 1999; Achard et al., 2002). Three pieces of legislature implemented by the Indonesian government, The Basic Agrarian Law (BAL), transmigration and decentralization, have contributed to the high rates of deforestation on Kalimantan (Indonesian Borneo) by enhancing the effects of two of the major proximate causes of deforestation Borneo: fire and logging.

The constitution of the Republic of Indonesia states that all natural resources in Indonesia are subject to state control as of 1960 (BAL; State Gazette 160, No. 104, Law No 5) (MacAndrews, 1986). Land may be privately owned, although individually owned parcels may not exceed 20ha in sparsely populated areas or 5ha in densely populated regions (MacAndrews, 1986).

Transmigration in Indonesia was a major policy begun in 1949 to relocate people from the more densely populated islands, mainly Java, Madura and Bali, to the more sparsely populated islands such as Borneo and Sumatra, Sulawesi and Papau (Lumbanraja et al., 1998). Families are subsidized for up to five years, after which time they are considered independent and are turned over to the local governments to be integrated into local villages (Fasbender and Erbe, 1990). Because BAL makes it difficult to secure land on the populous islands, the chance to obtain a plot through the transmigration program is appealing (Fasbender and Erbe, 1990).

Transmigration increases the population on outlying islands, especially Borneo and Sumatra. Increased population places added strain on the environment (Erlich and

Holdren, 1971; Meyer and Turner, 1992) partially through increasing the amount of agricultural land necessary to support the growing population (Lumbanraja et al. 1998).

Beginning in 2001, decentralization transferred responsibility of forest management from the Indonesian federal government to the local, or Kabupaten, administrative level (Fuller et al., 2004; Suwondo, 2002; Matsui, 2005). This has led to inconsistencies in land use designations with vast tracts of land often designated as both production forest and protected forest by the separate levels of government (Fuller and Murphy, 2006; O'Conner, 2004; Curran et al., 2004). Each level of government will independently assign land use boundaries, leading to areas of forest with often conflicting dual, or even multiple designations.

1.2.1 Logging

Several years after the BAL was implemented, nearly 80% of the forested land on the outer islands of Sumatra, Borneo and Irain Jaya was partitioned out to industrial logging companies in the form of 35 year leases (O'Conner, 2004). These concessions have been industrially efficient, as official sources estimate that more than 180 million cubic meters of large logs were harvested from the forests of East Kalimantan between 1969 and 2001 (Siegert et al. 2001).

Decentralization policies have helped to increase the rate of deforestation by creating a chaotic situation regarding land use rights. After decentralization, the 35 year leases originally awarded to timber concessions by the federal government fell within areas of land now controlled by regional governments. Local governments then began awarding their own leases, resulting in a dramatic increase in the total number of timber extraction leases (Scott Stanley, personal communication). This has resulted in overlapping boundaries of timber concessions, and stakeholders are now under pressure to extract the timber on their leased land before competitors.

Transmigration has also led to increased logging because transmigrants are generally resettled in forested lands controlled by the government (Siegert et al. 2001; Fearnside, 1997). This land is then deforested for building materials and agriculture.

1.2.2 Fire

The massive fires that occurred in 1982-83 and 1997-98, resulted in immense forest loss. In 1982-83, an estimated 3.5 million hectares of mainly forested land burned in Kalimantan (Malingreau et al., 1985). During the 1997–98 fires, at least 11.7 million hectares of forest, farmland and other vegetated land in Indonesia were destroyed or badly damaged (Tacconi, 2003), 4.7 million hectares of which were lowland evergreen forest (Fuller and Fulk, 2001).

Undisturbed tropical rainforest is normally highly resistant to fire because of low loads of available fuel, low fuel-energy content and high humidity even during drought (Cochrane and Schulze, 1999; Goldammer, 1999). However, a combination of anthropogenic and natural factors has recently increased tropical forests’ susceptibility to fire in Kalimantan.

Logging enhances the effects of fire. Siegert et al. (2001) showed that after the 1997-98 fires, only 5.7% of undisturbed forests were affected by fire compared to 59% of logged forest and 70.7% of the non-forest area. Additionally, 48% of logged forests showed severe or total damage, in contrast to only 4% of the undisturbed forests. Thus, logging activities were associated with both the location and severity of fires. This establishes a positive feedback loop, where forests not destroyed in the fires were severely damaged, significantly increasing the risk of recurrent fire disasters by leaving huge amounts of dead flammable wood (Siegert et al. 2001).

A linear relationship was found between El Niño-Southern Oscillation cycle (ENSO), and the occurrence of fires in Borneo (Fuller and Murphy, 2006). Specifically, La Niña years generate drought conditions in certain areas on the western side of the Pacific Basin, leading to an increased risk of wildfires in these areas (Kitzberger et al., 2001). Thus, natural climactic variation has increased the propensity for fire. Sources of ignition are somewhat more controversial, although it is fairly well understood that "few fires are accidental" (Stolle et al., 2003), in the sense that they are sparked by intentional human acts. Specifically, fire is used to convert forest into plantations and agricultural lands, and these are the source of the large scale fires events in 1982-83 and 1997-98 (Ketterings et al. 1999; Barber and Schweithelm, 2000).

Given these pressures on Borneo's forests, what contributions have been made by groups that adhere to alternate land management philosophies? Meijaard and Sheil (2007) reviewed the wildlife and conservation research conducted on Borneo and found that there has been a sizeable amount of work based on ecological approaches that typically investigates species-habitat interactions. The ecological approach employed by these and other scholars provides some balance to the resource-maximization strategies embraced by the Indonesian government. However, Meijaard and Sheil (2007) also identified a shortcoming in the literature; a failure to address threats to species or provide recommendations or guidelines for management.

1.3 The role of remote sensing and GIS

In addition to biome-centered and ecosystem-centered studies on Borneo, a significant effort has been placed on monitoring the land cover and land cover changes described above (Jepson et al., 2001; Curran et al., 2004). This is where the field of remote sensing may have made its greatest contribution. A large number of studies have investigated the relationship between fire and forest loss (Siegert and Hoffmann, 2000; Page et al., 2002; Dennis et al., 2005; Langner et al., 2007; Miettinen, 2007; Miettinen et al., 2007), which is not surprising given the significant impact of fire on Borneo's landscape. Many of these papers are methodological in nature and stress refining analytical procedures associated with remotely sensing fire (Fuller and Fulk, 2000; Siegert and Hoffmann, 2000; Boyd et al., 2004; Huang and Siegert, 2004; Siegert et al., 2004; Miettinen, 2007). Often the environmental relevance of these works is focused on the climactic impacts associated with effects of fire on global carbon budgets (Lu, 2006; Shimada et al., 2001) and addresses the impacts on biodiversity only broadly in the sense that forest loss negatively impacts biodiversity. While this approach may raise a "call to action" regarding the state of Borneo's forests, it falls short in addressing management implications or specific ways of fire threatens biodiversity as identified by Meijaard and Shiel (2007).

Outside of the fire literature, other remote sensing and geographic information science (GIS) work has attempted to make more direct links between land cover change and biodiversity, and develop methodology that can be applied directly towards conservation-based land management strategies (Foody and Cutler, 2003; Marjokorpi and Otsamo, 2006; Rodriguez et al., 2007). Specifically, GIS has been shown to be a valuable tool for assessing risk and designating conservation targets (Phua and Minowa, 2005; Marjokorpi and Otsamo, 2006). This work is valuable in that the methodologies employed (multi-criteria evaluation, simulation models) are able to incorporate different land use priorities and can be tailored to a preservationist, resource-maximization or ecological philosophy, or some combination thereof (Marjokorpi and Otsamo, 2006). In fact, perhaps the strongest contribution of the remote sensing and GIS literature to conservation in Indonesia is that the body of work is based on multiple land use paradigms, including resource maximization (Glauner et al., 2003) and eco-system centered approaches (Foody and Cutler, 2003; Marjokorpi and Otsamo, 2006). Furthermore, perhaps as a result of Geography's cross-disciplinary tradition, the geographic literature produces a holistic body of conservation research that addresses the role of and impacts on local peoples (Fox and Atok, 1997; Dennis et al., 2005; Dewi et al., 2005) government policy (Jepson et al., 2001; Dennis and Colfer, 2006) and economic markets (Glauner et al., 2003; Wright et al., 2007).

1.4 Species-Based Approach

Almost 80 percent of conservation research conducted on Borneo is restricted to describing species characteristics and species interactions with their environment (Meijaard and Sheil, 2007). Typically, these species-based approaches focus on endangered or charismatic species such as large carnivores. Utilizing charismatic and endangered species to promote conservation efforts is a strategy that is often adopted by various government and non-government organizations because of the perception that these species attract funding and public support as well as raise awareness of conservation efforts and environmental issues (Sergio et al. 2006; Kontoleon and Swanson, 2003). However, this approach to conservation raises broad concerns regarding the sometimes unpredictable socio-economic impacts, and the ecological effectiveness of the approach (Sample, 1994). With limited funding available for environmental conservation work, there is a concern that directing resources towards any one species leaves ecosystems and other biological processes understudied, despite the indispensable role of these systems in the life cycle of the species being researched (Entwistle and Stephenson, 2000).

The charismatic species approach (CSS) is based on principles taken from the umbrella species and focal species concepts. Both CSS and umbrella species approaches assume that the conservation of a selected umbrella species will confer protection to a large number of naturally co-occurring species. A challenge to this approach is that it depends on land managers having a thorough understanding of the habitat needs and threat tolerances of all aspects of the system they are managing, including correctly identifying the most species whose habitat requirements will best meet the habitat requirements of co-occurring species.

Unfortunately, the effectiveness of CSS has rarely been compared to alternative conservation strategies, and the collection of studies that evaluated the approach reveals mixed results (Simberloff, 1999; Roberge and Angestam, 2004; Lindenmayer et al., 2002). Andelman and Fagen (2000) examined several classes of potential umbrella species, including among others, large carnivores, charismatic species and highly endangered species. They found that very few species intentionally selected as umbrella species were more efficient at protecting habitat or other species than subject species chosen at random. Still, other studies have found umbrella species approaches do have the ability to convey protection to other species (Sergio et al. 2006; Caro, 2003), although the surrogate measures of biodiversity used in such studies has been questioned (Lindenmayer et al. 2002, Roberge and Angstrom, 2004).

Additionally, CSS rely on patterns of nestedness amongst conservation targets (Lindenmayer, et al. 2002). However, it has been shown that some communities are not nested (Malmqvist and Eriksson, 1995) and “it is inappropriate to assume without evidence that communities are nested” (Lindenmayer et al. 2002). The impact of scale is typically not fully accounted for in studies testing these strategies. Even if CSS provide sufficient habitat to background species (species to which it is assumed protection will be conferred if the target species is protected) on a landscape level, intensive fragmentation on a local scale may cause declines in species such as invertebrates that utilize micro-habitats (Murphy and Wilcox, 1986).

CSS is valuable to conservation efforts because charismatic fauna garner much broader public support than more abstract issues such as habitat destruction or climate change (Tobias and Mendelshon, 1991; Brown and Henry, 1993; Kollmuss and Agyeman, 2002). This appeal makes governments particularly quick to support CSS (Metrick and Wietzman, 1996). However, not all charismatic fauna are highly threatened, nor do they universally function as an umbrella species in their particular system, leading to conservation projects being prioritized on the bases that the species spear-heading the conservation effort is appealing or charismatic, rather than that it serves as an ecological umbrella species (Sergio et al. 2006; Entwistle and Dunstone, 2000).

Of particular importance for conservation is that an individual's willingness to pay (WTP) for a protected species extends beyond the existence of the species itself to protection of the species habitat as well. Kontoleon and Swanson (2003) showed there was three times the WTP for the natural habitat of a Panda population than there was for conservation of the population itself (Kontoleon and Swanson, 2003). This demonstrates that charismatic fauna can act as a magnet for raising funds that can be extended to the protection of ecosystems.

1.4.1 Implementation of CSS

The ability to gather public and support and funding makes CSS appealing for governments, and conservationists. Because the ecological usefulness of species-based conservation projects is debated, CSS is best applied as one of a suite of strategies, as part of an adaptive management approach for example (Sample, 1994; Meffe and Carrol, 1997). Placing too much emphasis on the ability of any one species' protection to protect other co-occurring species may prove disastrous if those background species do not respond as anticipated. Therefore, charismatic species may best be employed as a “social hook” to obtain public and government support (Lindenmayer and Fischer, 2002), but complimentary strategies, such as ecosystem-based approaches, should be applied simultaneously when delineating protected areas.

Even though species-based approaches are not sufficient conservation strategies on their own, they still have an *important role* in conservation efforts, in that species are often more able to attract public attention and raise funds than less tangible causes such as habitat loss and climate change (Tobias and Mendelshon, 1991; Brown and Henry, 1993; Metrick and Wietzman, 1996; Lindenmayer and Fischer, 2002; Kollmuss and Agyeman, 2002). Addressing the species-based approach, Meffe and Carroll (1997) surmise “…species should *always* be part of the conservation scene, [although] a species-by-species approach will not *by itself*, make much headway in the big picture.” (original authors’ emphasis).

1.4.2 The role of Storm’s Stork

Charismatic species tend to be physically attractive, mysterious or possess anthropomorphic traits. Avifaunas are perhaps an exception, as large conservation movements have been directed at species such as the California Condor (*Gymnogyps californianus*). This is in part due to the interest of ‘birders’, individuals who have devoted substantial leisure time for seeking out birds in the wild (particularly rare and little known species). Birders have become a significant voice for the protection of avifauna and their habitats by way of the tourist dollars spent seeking specific species and their role in the research community as primary data collectors for many current range and status assessments (Birdlife International, 2001). Bird watchers have been called “ideal eco-tourists” because they tend be affluent, are likely to have a high awareness of nature, and spend significant amounts of money to see birds (Cordell and Herbert, 2002; Sekercioglu, 2002). In 1996 alone 17.7 million birdwatchers spent approximately $7.6 *billion* on expenses related to bird watching trips, excluding equipment (US Department of the Interior, Fish and Wildlife Service and US Department of Commerce, Bureau of the Census, 1996). Even an individual bird can have a large economic value. It was estimated that a single Macaw visiting a clay lick in south-eastern Peru can generate $750 to $4700 in tourist receipts in a year and $22,500-$165,000 over the course of its life (Munn, 1992).

One species in Borneo that has the potential to generate interest and revenue is Storm’s stork (*Ciconia stormi*). *Ciconia stormi* has been described as "probably one of the least numerous of all the storks in the world" (Hancock et al., 1992), and an assessment in 1997 placed its numbers below 1,000 individuals (Rose and Scott, 1997). Birdlife International (2001) categorized the species as endangered because “it has a very small, very rapidly declining population, owing to destruction of lowland forest through logging, dam construction and conversion to plantations.”

A variety of eco-tour companies in Borneo have tapped into the bird watchers market, and specifically state the potential for seeing *Ciconia stormi*:

> “See as many as nine primate species, crocodiles, butterflies and rare birds such as Storms stork.” – Rimba eco-lodge (http://www.rimbalodge.com)

> "Highlights included stunning views of … the endangered Storm's Stork." - Rockjumper birding tours (http://www.rockjumper.co.za)

With the average bird watching trip to a developing country organized by one of the six major international bird watching companies (see birding.com for examples) containing 12 participants each paying $4,000, this market has become very lucrative (Weaver, 1998). Indeed, in addition to those quoted above, several other tour companies and lodges operating in Borneo also appeal to bird watchers specifically interested in seeing *Cicnia stormi* including the Borneo Eco-lodge (http://www.responsibletravel.com) and Victor Emanual Nature Tours (http://www.ventbird.com).

CSS have shown to be effective in Borneo where the Orangutan has become an emblem for generating funding, as demonstrated by USAID's $8 million contribution targeted at "…reducing the level of threat to select orangutan populations. This project will not support orangutan rehabilitation, re-introductions, or translocation efforts. This approach [to orangutan conservation] is consistent with the scientific consensus that the first priority in orangutan conservation must be the protection of habitat and populations in the wild" (Rose, 2007). This last statement again reflects the ability of protecting charismatic species to extend beyond the species itself to the species habitat.

Strategic plans utilized by NGOs are easily expanded to include other charismatic species. One such group, The Nature Conservancy (TNC) which receives a portion of the USAID Orangutan funding, has also embraced *Ciconia stormi's* ability to "effectively promote our conservation work in this region and get public support for our work"

(Meijaard, 2005). Areas of small rivers in undisturbed forests are considered important habitat for *Ciconia stormi*, making it a potential indicator species for forest ecological health and a “perfect umbrella species for the protection of large, healthy forest ecosystems” (Meijaard, 2006a; Meijaard, 2006b).

1.5 *Ciconia stormi* Ecology

In addition to being rare, *Ciconia stormi* might also be the least studied stork. Only a handful of field studies have been conducted, most of them of short duration and focused on breeding behavior (Danielsen et al., 1997; Nakhasathien 1987; Yaacob 1992). Birdlife International (2001) compiled all available information on the ecology of *Ciconia stormi* and its distribution and threats. Following Silvius and Verheugt (1989), they concluded that more work is urgently needed to assess the species’ habitat requirements and exact distribution and to identify critical areas for nature reserves.

Ciconia stormi habitat is primary dipterocarp, freshwater swamp, lowland evergreen and semi-evergreen forests within the Greater Sunda region (Birdlife International, 2001), although it information may be too scant to determine if optimal habitat is riverine or swamp forest (Danielsen et al., 1997). Records indicate that it may be tolerant of logging to some degree, so long as it has access to water margins (oxbow lakes appear to be particularly favored). However records from disturbed, and logged habitats give no indication of the long-term value of such areas to the species, and until research proves otherwise it is prudent to assume that such areas are sub-optimal, or at least offer only temporary benefits (Sebastian in litt. 1999). The bird has also been observed in small agricultural plots that were recently burnt, presumably picking up invertebrates killed by the blaze (Birdlife International, 2001).

The core of this species' habitat appears to be floodplains of large rivers, since it requires creeks, rivers, pools, trackside puddles, lakes, freshwater swamps and other damp clearings, where it forages along the shaded margins, sometimes venturing into paddyfields and other cultivated areas flooded in the wet season (Birdlife International, 2001). *Ciconia stormi* feeds primarily on fish, frogs and other aquatic organisms (Danielsen et al., 1997). The crop and stomach of one bird were exclusively packed with one species of large black earthworm, found commonly in riverbanks (Smythies 1981). Danielsen et al. (1997) also found that the species tended to feed within 2 to 3 km from its nest site, which was located overhanging a river.

The species is known to be shy and avoid contact with humans, and a historical database of stork sightings compiled in conjunction with The Nature Conservancy (TNC) showed that only five of 190 sightings were made from a road, suggesting the species avoids this feature of the landscape. Observations by Danielsen et al. (1997) revealed, "When there was noise from chain saws or motorized boats, the adult bird present would press its head and body into the nest, forming a ball of black feathers with only the eyes exposed. If humans came within sight of the nest, the adult bird would not return until 2-3 hours after the intruder had departed." Noise from roads was shown to penetrate approximately 350m and 380m into forests (Pocock and Lawrence, 2005), so species may need this buffer distance to avoid anthropogenic noise pollution.

The species typically utilizes lowland forests under 240m above sea level (a.s.l.), with most observations under 30m a.s.l. (Birdlife International, 2001; Lambert, 1992). There have been sightings of the species over 600 m a.s.l., although these records are sparse.

1.6 Methods and Limitations for Estimating Species Distribution from Point Observations

Data availability is often a major constraint when creating large-scale models of species distribution (Osborne et al., 2001; Kaschner et al., 2006). "The vast majority of data that is available today consist of presence-only data sets (i.e. where there is no information on the absence of species)" (Zaniewski et al., 2002). Time and budget constraints mean that conventional ground-based survey methods are often not a viable option for obtaining detailed data regarding species distributions. In locations such as Borneo, the undeveloped and remote landscape make conducting systematic surveys that result in presence/absence data extremely challenging. Even in less challenging circumstances, true absence data is often very difficult or near impossible to obtain (MacKenzie et al., 2002; Mao et al., 2005), and working with presence-only data may be a more accurate portrayal of what is known about species distributions considering false absences can bias analyses (Hirzel et al., 2002). Common ecological modeling tools (e.g. logistic regression or classification and regression trees) are difficult to apply to systems with limited information on areas where species are absent (Guisan and Zimmermann, 2000).

Several models have been developed for assessing species-habitat relationships when only presence data is available. Most presence-only models work by generating predictions based on the frequency distribution of 'presence-cells' in relation to environmental variables. The genetic algorithm for rule set production (GARP), ecological-niche factor analysis (ENFA) and relative environmental suitability model (RES) have all been developed to work with presence-only data.

The genetic algorithm for rule set production (GARP) is a predictive ecological niche model capable of working with presence only data (Anderson et al., 2002; Kaschner et al., 2006). GARP is a genetic algorithm (GA) approach that uses localities of a species' occurrence and environmental data from GIS layers to produce a niche-based model of the species' environmental requirements (Stockwell and Noble, 1992; Stockwell and Peters, 1999).

One limitation of the GARP method is that it requires 50-100 representative occurrence records to generate accurate predictions (Stockwell and Peters, 1999), and so may not be appropriate to apply to extremely rare or elusive species such as *Ciconia stormi*.

The ecological-niche factor analysis (ENFA) is a multivariate approach that compares the distributions of the eco-geographical variables between the presence data set and the whole study area (Hirzel, 2002). ENFA has been useful in identifying habitats associated with various species, as well as their tolerance to living in sub-optimal or average habitats (Santos, 2006).

ENFA is purely a descriptive method and cannot identify causal relationships (Hirzel et al. 2002). Another major limitation to the model is that it assumes data represent an unbiased sample of the available habitat (Hirzel et al. 2002). This is often not the case as sampling methods are regularly subject to constraints such as logistical restrictions, or financial limitations.

The relative environmental suitability (RES) model is useful when species presence records are extremely rare or not geographically specific. RES works by generating an index of the relative environmental suitability of each cell for a given species by relating known habitat usage to local environmental conditions. RES predictions are based on clearly defined assumptions and parameter settings and are thus reproducible and testable. “Using available expert knowledge, RES modeling allows the prediction of potential distribution and habitat usage on very large-scales across a wide range of species in a standardized, quantitative manner.” (Kaschner et al. 2006).

GARP, RES and ENFA depend on large sample sizes, unbiased sampling methods and available expert knowledge respectively. These restrictions make these models inappropriate for work on *Ciconia stormi*, because available data are characterized by a small number of sightings, limited expert knowledge and ad hoc sampling methods.

1.6.4 Issues of Scale

The effects of scale on analysis of species-habitat relationships should be integrated into every study (Weiers et al., 2004). For work that deals with classification of habitat, the optimum number of habitat features and the most appropriate scales need to be addressed. This has to be specified according to the ecology of the target species and the local landscape configuration (Gottschalk et al., 2005).

Pearlstine et al. (2002) argued that many of the basic ecological characteristics of species' habitat are not incorporated into land cover classes. For example, analysis at a broad scale will not identify hedgerows, fences and small patches of habitat that may be important to wildlife. By applying ground-based habitat data, Mack et al. (1997) found that species–area relationships described the bird species numbers better than those from larger-scale data. A hierarchical approach viewing habitat analysis first from the broadest scale, and then working down to the finest level of scale necessary to answer the question of interest may be useful in deciding which scale to use (Morrison et al. 1992).

1.6.5 Precision of presence data

A second challenge to investigating species-habitat relationships is that presence data is often obtained from opportunistic sightings, museum records and other ad hoc methods, resulting in records that are rarely geo-referenced with precision, and heavily influenced by observer bias (Zaniewski et al., 2002; Burgmand and Fox, 2003; Ottaviani et al., 2004). Again, a variety of methodologies have been developed to deal with the challenge of translating imprecise presence data into species distributions, although there has been little conformity on the issue (Flamm et al., 2001; Burgman and Fox, 2003; Szabo et al., 2007; Franzreb, 2006). Typically, methodologies that adjust for the imprecision associated with presence data due so by representing species presence “inside a polygon of variable size such as a cell of a sampling grid, a protected area, a geographic area associated to a toponym, a cell of an atlas, etc. Any of these polygons describes the uncertainty associated with the presence data, as the exact location can no longer be recovered within the polygon.” (Ottaviani et al., 2004). The limitation of this process is that invariably, areas of non-presence are included in the constructed polygons, and this area is likely to bias subsequent analysis of the environmental conditions inside of the presence polygon.

The most widely employed methods are kernel density estimators and minimum convex polygons or convex hulls (IUCN, 1994; Worton, 1995; Burgman and fox, 2003). Each of these methods results in polygons of presence, rather than point data. Minimum convex polygons and variations thereof are more widely adopted, in part because they can be constructed using data from a variety of sampling strategies (IUCN, 1994; IUCN 2001; Guichon et al., 2005; Farnsworth and Ogurcak, 2006; Franzreb, 2006; Matson et al., 2006). Minimum convex polygons “are the smallest polygon in which no internal angle exceeds 180 degrees and which contains all sites” (Burgman and Fox, 2003). These polygons are constructed around the most extreme points associated with species presence records and therefore have a strong tendency to over-estimate species occurrences (Burgman and Fox, 2003; Franzreb, 2006). Overall, “There is little to recommend convex hulls for range estimation…biases may be very substantial, even when errors in the location of observations are small” (Burgman and Fox, 2003). Kernel estimators are less likely to be biased by small sample sizes than minimum convex polygons, and result in more accurate distributions in some instances (Ostro et al., 1999; Franzreb, 2006). However they are most often applied in situations where radio-tracking data is available (Worton, 1995), and so are limited in their potential application.

1.7 Thesis Overview

This thesis builds upon the idea that species-based approaches have a role in conservation-biology, particularly when targeted species coincide with hot-spots of biodiversity (Meffe and Carroll, 1997). Conservation efforts have found certain avian species effective symbols for raising funds and drawing awareness to environmental causes. In species-based studies, there is a need to address the biological and ecological needs of species in a way that informs decisions and provides concrete recommendations for land managers (Meijaard and Sheil, 2007).

This thesis identifies potential areas of *Ciconia stormi* presence, with these areas being assigned a range of values describing the level of confidence that an individual was present at any given location. The areas of presence are created by combining sighting records and the various levels of imprecision associated with those records with the point-radius method for geo-referencing species occurrences (Wieczorek et al., 2004). The resulting map improves upon polygons of binary presence/non-presence data in that the areas identified as presence are represented as continuous values of potential presence, and can be examined at various levels of confidence. Forest canopy conditions within those areas of potential distribution are then analyzed at multiple scales using a variety of remote sensing and GIS techniques to identify levels of percent canopy cover associated with locations where *Ciconia stormi* is seen. Finally hemispherical photographs obtained in forest locations with different logging histories are used to link those land-use regimes to the canopy requirements of *Ciconia stormi*.

Chapter II

Probabilistic historical distribution of Storm's Stork (*Ciconia stormi*) on Borneo: A MCE approach to mapping rare species

2.0 Background

A problem common to biological and ecological research is that records of species presence are often not geo-referenced. In fact, there are over one billion biological specimen records worldwide, 99% of which lack geographic coordinates (Duckworth et al., 1993; Beaman and Conn, 2003; Guralnick et al., 2006). This severely limits the extent to which species distributions can be mapped and analyzed in conjunction with other spatial data (Graham et al., 2004). In spite of the need to make these types of biological specimen data compatible with Geographic Information Systems (GIS) (Krishtalka and Humphrey, 2000), there has been little effort to developing appropriate methodologies to geo-locate records of this nature, (Wieczorek et al., 2004). Some researchers have argued for the necessity of categorizing the various levels of geographical uncertainty associated with geo-locating species records (McLaren et al. 1996; Knyazhnistskiy et al. 2000). However, researchers often ignore the imprecision associated with geo-referencing species presence locations and assign location based on educated guesses (Wieczorek et al. 2004; Duckworth et al. 2006), despite the fact such imprecision may render subsequent analysis of little utility (Fisher, 1999).

The most common method used to deal with imprecise presence data is to assign presence the data to a polygon, in which the exact location of the presence record is then lost within the area of the polygon (Flamm et al., 2001; Ottaviani et al., 2004; Chauvel et al. 2006; Szabo et al., 2007). However, by scaling up the level of analysis in this way, information on relationships and processes that manifest themselves at more local scales are potentially lost (Li et al., 2002). The most widely applied methodology, as discussed in chapter one, is the construction of minimum convex polygons (IUCN, 1994). This technique however, is prone to over-estimating areas of species presence (Burgman and Fox, 2003; Franzreb, 2006).

One alternate methodology that has been developed to assign geographic coordinates to presence data is the point-radius method for geo-referencing species locations from descriptive records (Wieczorek et al. 2004). The point-radius method assigns each record a point to mark the position most closely associated with the description of the record. A radius is then derived to “describe the maximum distance from the point within which the locality is expected to occur” (Wieczorek et al. 2004). This method has been applied by the Global Biodiversity Information Facility (GBIF; http://www.gbif.net) to provide over 100 million species-occurrence records. Other variations of this method have been adopted by the Tulane University’s GEOLocate program (http://www.museum.tulane.edu/geolocate/) and BioGeomancer (http://www.biogeomancer.org/) projects for geo-referencing biological species occurrences.

A thorough examination of the literature reveals no other studies that have developed a probabilistic distribution of an individual species based solely on known occurrences. MacKenzie and colleagues (2002) constructed a methodology intended to do so, but their work relied on estimating probabilities of detection based on mark-recapture data, or known sampling efforts. These types of data are often unavailable for rare or difficult to capture species or in situations where presence records were collected *ad hoc* without standard or known sampling effort.

2.1 Chapter Overview

The work presented in this thesis differs from MacKenzie et al. (2002) and others in that it not only addresses the imprecision associated with the location of presence records for a historical database of *Ciconia stormi* sightings, but it develops a methodology to account for this imprecision. Here, a rule-based system is used to identify potential sources of imprecision in the dataset, and this imprecision is then expressed spatially via the development of a confidence map of species locations. The map was created by combining the point-radius method of geographical uncertainty with a fuzzy-logic multi-criteria evaluation (MCE) approach. The final product contains values that express levels of confidence that a sighting occurred at that location. This is expressed as likelihood, with values between 0 and 1. These values are chosen because they are easily interpretable, although they do not represent true statistical likelihoods or probabilities. Taken as a whole, these areas represent locations where *Ciconia stormi* was potentially present and are hereafter referred to as areas of potential presence.

2.1.1 Presence Only Data

As discussed in chapter one, research methodology is often constrained by the types of data available. This process explored in this chapter is based on presence-only data, as opposed to presence-absence data. Because of the shy and secretive nature of *Ciconia stormi* (Birdlife International, 2001), along with the remoteness of much of its habitat, systematic surveys that result in absence-data are not feasible. Many studies investigating species-habitat relationships have developed methods to deal with presence-only data (for a review see Gottschalk et al., 2004). Typically, these models use deductive approaches to examine species-environment relationships by generating predictions based on the frequency distribution of 'presence-cells' in relation to environmental variables. Several such models have been developed including the Generic Algorithm for Rule-set Prediction (GARP) (Anderson et al. 2002), ecological niche factor analysis (ENFA) (Hirzel, 2002) and relative environmental suitability model (RES) (Kaschner et al. 2006) as discussed earlier in chapter one. These deductive approaches derive knowledge of species-habitat relationships from expert knowledge such as information drawn from literature or personal experience. Alternatively, inductive approaches draw conclusions from species-habitat relationships from correlations between observations of species' presence and the environmental variables at those locations. In the case of *Ciconia stormi*, there is very little verified expert knowledge available, meaning an inductive approach was with only available option.

2.2 Methods

2.2.1 Data Set Description:

A list of known sightings of *Ciconia stormi* on Borneo was compiled from various sources, including first-hand sightings, primary literature, records previously compiled by Birdlife International (2001), natural history museum collections and eco-tour logs. Unpublished field records were also acquired through correspondence with wildlife biologists and other researchers working in Borneo. The dataset may omit some information, as many natural history museums around the world were not contacted. Regardless of some omissions, this dataset is the most complete compilation of *Ciconia stormi* sightings known to the author. The dataset and consists of 197 sightings for Borneo, ranging from 1954 to 2007. 152 of the sightings (77%) occurred during or after 1990, and 78 of the sightings (40%) were recorded during or after the year 2000. There are 30 records in the dataset with relatively precise GPS coordinates; however, the majority of the records are descriptive in nature and precise geographic locations are not available.

2.2.2 Precision Rankings

Based on the information describing the location of each sighting, online gazetteers were used to assign a point location to each record of *Ciconia stormi* that had not been previously geo-located. Several records could not be associated with any location using these sources, or the location was too general (e.g. “Borneo”) and were excluded from the analysis. A set of rules were developed based on the point-radius method to categorize each record based on the various sources of uncertainty associated with the descriptive localities and the resources used to geo-reference them such as Global Positioning Systems (GPS), on-line gazetteers and available maps (table 2.1) (Wieczorek et al. 2004). Based on those rules, a confidence ranking ranging from one to five was assigned to each record with one being the most precise location and five being the least precise.

Sightings that had been recorded with GPS coordinates were regarded as having the least amount of uncertainty associated with them and assigned a confidence ranking of one. It was assumed that the only uncertainty associated with these records resulted from the error of the GPS location, which may be as low as sub-meter for some commercial receivers that allow 3-D differential correction. Because of *Ciconia stormi*, is known to utilize the 2-3 km of habitat surrounding a nest-site (Danielsen et al., 1997), the surrounding 3 km were considered areas of presence as well. No other information on ranging distances for this species is available in the literature.

Records of species sightings at a named locality such as a town or a feature of the landscape (e.g. a lake or bridge) were assigned a confidence ranking of two. For these records, uncertainty is associated with the areal spread the location. For example, if a sighting occurred at the town of Kuching, it is unknown if that was at the northern or southern edge of the town, at the town square, or in the geographic center of the town. The distance from one edge of the location to another generates uncertainty about the precise location of the sighting.

Records of a sighting recorded as an offset along a path were assigned a confidence ranking of 3. Records such as: ‘five kilometers north from the city of Samarinda along the Mahakam River’, have three sources of uncertainty. The first source of uncertainty results from the span of the starting location as previously described. Distance imprecision is also a source of uncertainty, and this source of uncertainty was calculated “based on the fractional part of the distance, using 1 divided by the denominator” (Wieczorek et al. 2004). For example, 5 km has a fraction of 1/1 resulting in an uncertainty of 1 km because it is unknown if the sighting occurred at 5.01 km or 5.99 km. For a record with a distance of 5.5 km, the fraction is ½ resulting in an uncertainty of .5 km and for a record stated as 5.25 km, the fraction is ¼ resulting in an uncertainty of 0.25 km (Wieczorek et al. 2004). The third source of uncertainty is from measurement technique, i.e. whether the sighting was recorded as distance traveled on the ground, or distance “as a bird flies” (recorded as a straight line drawn on a map). This uncertainty was calculated based on the average distance between the starting location and the sighting, combined with a qualitative estimate that accounted for the intensity of the curvature of the path or route.

Sighting records listed as an offset at a heading, for example, 8 km north of Samarinda, were assigned a confidence ranking of four. This category of records has four sources of uncertainty. Uncertainty associated with the areal spread of the starting location, distance uncertainty, measurement technique uncertainty and uncertainty associated with directional precision. Uncertainty here arises from the fact that we cannot know what the observer meant by ‘north’, other than that it is not ‘east’ or ‘west’. In this case, uncertainty is 45 degrees in either direction from the given heading. If the sighting was recorded more precisely than cardinal directions, for example NW, then the observer demonstrated a sensitivity to inter-cardinal directions. Therefore, NW could be any direction between WNW and NNW. Thus the directional uncertainty for these records is 22.5 degrees in either direction from the given heading. If the location was recorded with more precise directions, for example WNW, the directional uncertainty is correspondingly lessoned to 11.25 degrees in either direction (Wieczorek et al. 2004).

Sightings records that were associated with a region or within a boundary were assigned a precision ranking of 5. These included sightings within a national park or a large geographic area such as a peninsula or simply along a river without providing any other directional information.

Table 2.1. The rules used to assign confidence rankings to sighting records of *Ciconia stormi* were based on the point-radius method for geo-referencing locality descriptions. The rankings range from one to five in decreasing order of precision, with one representing the most precise records, and five assigned to the least precise records. The rankings take into account the different ways imprecision is introduced into a record.

Type	Description	Example	Ranking
Coordinates	Geographic coordinates provided. Datum may still be unknown.	N1.6145; E117.1553	1
Named place	Sighting assigned to a specific location	Kuching, at the airport	2
Offset along a path	The sighting is described as a route from a named place	Kinabatangan River, 5 km from Kinabatangan Jungle Camp North along river	3
Offset at a heading	The locality contains a distance in a given direction from a named point	Upper catchment of the Sungai Sebangau, 20 km south-west of Palangkaray	4
Named area or region	The sighting is described as being within a known geographic region	Seen in the Sepilok Forest Reserve or Seen on the Mahakam River	5

2.2.3 Multi-Criteria Evaluation

Multi-criteria evaluation methods provide the technical tools for relating (standardizing, weighting, and combining) different factors into a single spatial output within a GIS framework (Eastman, 2006). This method provides a way to integrate expert knowledge into empirical models using fuzzy logic techniques (techniques that assign an observation partial membership to multiple classes). Fuzzy logic techniques aside from MCE are widely used to produce predictive maps of potential species distribution based on suitable habitats (Urban and Szymelfenig, 2003; Store and Jokimaki, 2003; Real et al. 2006). However, a search of ISI Web of Science found no studies utilizing MCE approaches to create probabilistic maps of species presence based strictly on species occurrence, despite the previously noted lack of precise locations for much of the world's species presence data.

2.2.4 Preprocessing techniques

The complete data set of *Ciconia stormi* sightings was converted to a vector point shapefile using ArcView 3.3 and then projected from Lat Long to the UTM 50 North coordinate system. The data set was also separated into two temporal groupings, sightings that occurred between 1988 and 1998 and sightings that occurred from 1999 to 2007. These delineations were chosen because they represented a nearly equal allotment of sightings from the dataset (74 and 76 sightings respectively), and because they are fairly evenly distributed temporally from the acquisition dates of the land cover data used in an analysis of forest change addressed in chapter 4, 1993 and 2004 (see chapter 4). Each of the temporal groupings was then further divided according to their precision rankings. There were no sightings between 1988 and 1998 in category 3, resulting in 9 separate files. The shapefiles were then imported into IDRISI (Eastman, 2006) for subsequent processing and analysis.

The data sets were rasterized and assigned a 500 m resolution for the extent of Borneo (108° 30' E to 119° 17' E; 4° 15'S to7° 35'N) using IDRISI (Eastman, 2006) to match the resolution of forest cover data used in chapter 3. A distance file was then created for each of the files, where the distance of each pixel from a sighting was calculated. It should be acknowledged that this distance layer does not retain information where multiple sightings occurred within a single pixel, i.e. distance from a single sighting is the same as distance from a location of multiple sightings. This may lead to underestimation of confidence of presence in these instances.

2.2.5 MCE Implementation and Factor Weightings

Fuzzy functions were used to standardize the nine spatial layers of individual confidence rankings and assign all grid cells a “proxy” value representing the overall confidence of *Ciconia stormi* presence within a pixel. These fuzzy functions can be specified to be monotonically increasing or decreasing functions, and are controlled by breakpoints (Eastman, 2006). For a monotonically increasing function, the first of these control points establishes where the function begins to rise above the minimum confidence score of zero, while the second point indicates where it begins to level off and approach the maximum confidence score of 255. A monotonically decreasing function adheres to these same principles but the function begins near the maximum confidence score and decreases to a zero score (Eastman, 2006).

The weightings and control points for the factors were assigned based on the precision rankings described in section 2.2, where sightings with a precision ranking of one were assigned a high weight within the area where the sighting could have potentially occurred, and precision rankings 2-5 were assigned progressively lower weightings within the areas those sightings could have potentially occurred. The goal was to map areas where a sighting could have potentially occurred, based exclusively on evidence from the data. The process of assigning weights and fuzzy functions to point sightings involves a set of assumptions about the likelihood of presence as a function of distance. These assumptions were as follows:

1. Areas surrounding high-precision sightings (e.g. area with r =20 km) were assigned low values of confidence in presence. There is strong evidence that a sighting occurred at the location marked by the GPS, but there is no evidence that a sighting occurred 20km away, therefore assigning a high confidence of presence to areas far from that point may be spurious and to do so may lead to errors in the analysis by over-valuing the environmental variables found at those distances if *Ciconia stormi* is not present there. 2. Areas surrounding the point associated with a low-precision sighting receive moderate levels of potential presence, because imprecision in the data implies that a sighting may have occurred anywhere within a polygon. For instance, there was a sighting recorded as "seen in Sepolak Forest Reserve". The center of Sepolak Forest Reserve was identified via visual estimation, and the observation point was placed at that center. All areas within the radius specified by the fuzzy functions (see page 39) were assigned an equal weight. The process of assigning low precision sightings larger areas of potential presence than precise sighting records may seem to reduce the importance of high-confidence sightings, but the relative weightings of the sightings helps compensate for this. Overall, high precision sightings are weighted much higher than low precision sightings, so the total area of potential presence surrounding a high precision sighting is greater than the total area associated with a low-precision sighting.

Expert knowledge derived from existing published literature and personal communication with other researchers active in the region was used to establish the type of function utilized and the location of control points.

Confidence ranking 1: Danielsen and colleagues (1997) found that the species tended to feed within two to three km from its nest site, meaning if a sighting occurred in the middle of an individual's territory, that individual's range would encompass all the area within a three km radius of the sighting location. However, if the sighting occurred at the outermost edge of an individual's territory, the opposing outermost edge of that territory could potentially be six km away in any direction, resulting in an circle with a radius of six km (see figure 2.1). This results in a central area of six km, within which the species was likely to have been sighted and sharp decrease in the likelihood of presence outside of that radius (i.e. there was no evidence in the data suggesting a sighting occurred outside of that area). Therefore, sightings with an error ranking of one were assigned a monotonically decreasing J-shaped curve with breaks created at 3,000 and 6,000 meters. The J-shaped curve assigns equal probability of occurrence to all areas within 6,000 meters, with sharply declining likelihood of occurrence as distance increases.

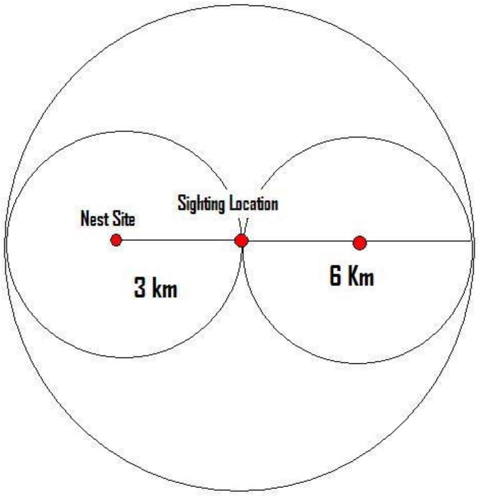

Figure 2.1. Figure depicting potential distance from Stork sighting to outer edge of its range.

Confidence Ranking 2: Sightings with an error ranking of two were assigned a monotonically decreasing J-shaped curve with breaks at five km and eight km. These represent the three and six km used to bound the uncertainty associated with a potential home range, and the additional two km of uncertainty associated with the areal spread of the named place. Two kilometers was assumed as an average extent of named places for this dataset, which tended to be small-to- medium sized towns and villages oriented along rivers (Meijaard, Boorsma, personal communication). The J-shaped curve assigns equal likelihood of occurrence within the areal spread of the named place, with a sharply declining likelihood of occurrence as distance from the named place decreases.

Confidence Ranking 3: Sightings with an error ranking of three were assigned a monotonically decreasing sigmoidal function with breaks at seven km and 10 km. This accounts for the uncertainty associated with home range (three and six km), plus the size of named place (approximately two km), plus distance uncertainty (one km for all of the records in the dataset) and uncertainty of measurement technique (estimated to be one km for the records in the dataset). This uncertainty was calculated based on the average distance between the starting location and the sighting, combined with a qualitative estimate that accounted for the intensity of the curvature of the path or route. A sigmoidal function was chosen for all sightings not associated with a named place or X,Y coordinates. The sigmoidal function places the greatest amount of confidence at the location of the centroid (location where the sighting was thought to have occurred based on the description in the sighting record), followed by a slow decay of confidence as distance from the designated centroid increased. Confidence then decreases exponentially when distance from the centroid (i.e. the radius length) surpasses the distance that can be accounted for by the total imprecision embedded in the sighting record. Placement of the centroid derived from descriptive records are more subjective than those associated with a named place or X,Y coordinates, making it less appropriate to distinguish a sharp break between the areas of highest and lowest likelihood associated with the sighting as results from a J-shaped curve.

<u>Confidence Ranking 4:</u> Sightings with an error ranking of four were assigned a monotonically decreasing sigmoidal function with break points at 16.5 km and 19 km. The sigmoidal function was chosen for the same reasons described for sightings in category 3. Wieczorek et al. (2004) have developed a set of complex algorithms to derive error associated with orthogonal directions and distances that depend on the location of the point being geo-referenced on the globe, and made a geo-referencing

calculator available to compute those values (http://manisnet.org/gc.html). Their calculator was used to establish the minimum and maximum errors associated with the records for the dataset. Uncertainty associated with the areal spread of the named place and home range extent were also added which resulted in a minimum combined uncertainty of 16.5 km and a maximum of 19 km. Because of the complexities associated with calculating combined distance and directional error, the calculator was deemed appropriate to use for records in category four. However, more control was desired for calculating the potential location error of the other categories so it was not applied to records in categories one, two, three or five.

Confidence Ranking 5: Sightings with an error ranking of five were assigned a monotonically decreasing sigmoidal function with break points at 5,650 m and 36,000 m. These values were based on the size of the radii corresponding to the smallest and largest areas described in the dataset. There were several sightings from within small national parks with areas around 10,000 ha, an area covered by a radius of approximately 5,650 meters. There were also some sightings from within Tanjung Puting national park, which at approximately 400,000 ha was represented by a radius of 35,700 meters.

After establishing the monotonic function and control points for the factors in each sighting category, the factors were comparatively weighted using the Analytical Hierarchy Process (AHP), which "utilizes a pair-wise comparison approach to derive factor weights" (Eastman, 2006). These values were then aggregated using a Weighted Linear Combination (WLC), which results in an averaging of factor scores by allowing a low suitability score in one factor to be compensated for by a high suitability score in another factor (Eastman, 2006). Sightings with a lower confidence rank were weighted more heavily than those with a larger error ranking (table 2.2). The consistency ratio between factor weights was .05, which is considered acceptable (Eastman, 2006).

Table 2.2: the eigenvector of weights for the potential confidence rank categories. The consistency ratio between factor weights was .05, which is considered acceptable.

Confidence Rank	Weighting
1	0.5455
2	0.2396
3	0.1220
4	0.0641
5	0.0289

2.3 Outputs and Results

The final model was run three times, once for the complete dataset (figure 2.2), once for the records from 1988-1998 (figure 2.3) and again for the records from 1999-2007 (figure 2.4). The MCE utilizes values from 0-255, which were rescaled to values between 0-1, where zero represents no likelihood of *Ciconia stormi* presence in that location, and 1 represents a 100% likelihood of stork presence in a pixel. It is important to note that these are not true probability or likelihood values, only numeric representations of high to low confidence of presence. For example, regions of 20% confidence do not really have a 20/100 chance of stork presence having occurred, nor are those regions half as likely to have had a stork sighting as areas labeled 40% likelihood. Instead, the numeric values provide a continuous range of data values that reflect a systematic synthesis of disparate streams of sighting evidence. The continuous data can then be manipulated or analyzed in accordance with user needs. This may be useful in instances where geographically targeted conservation efforts are more feasible than those based on species-environment relationships (Sherry and Holmes, 1996) (e.g. efforts to protect a site where a species is known to occur may be more practical than attempts to restructure national timber extraction guidelines).

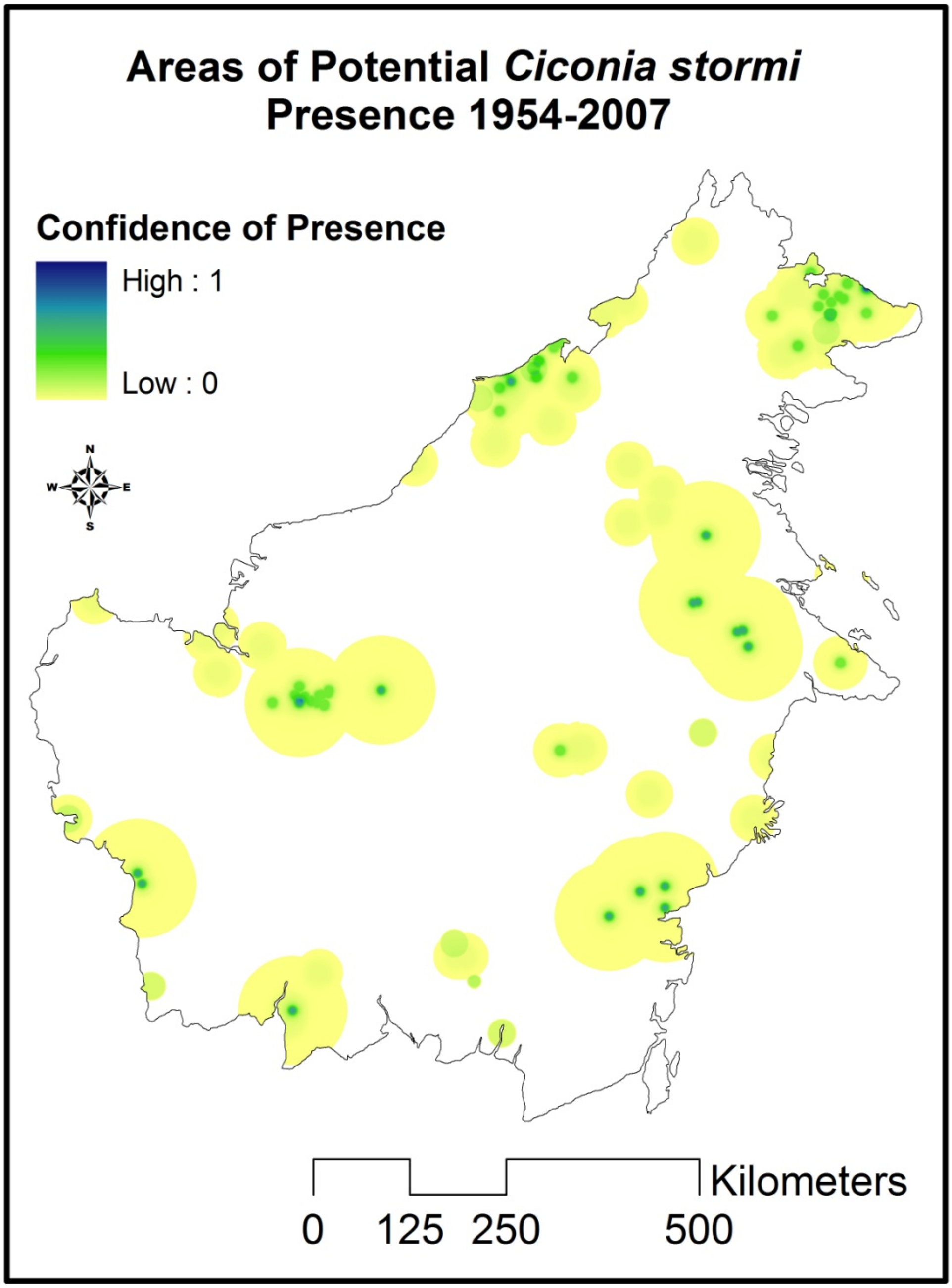

Figure 2.2. MCE results depicting areas of potential stork presence from 1954-2007.

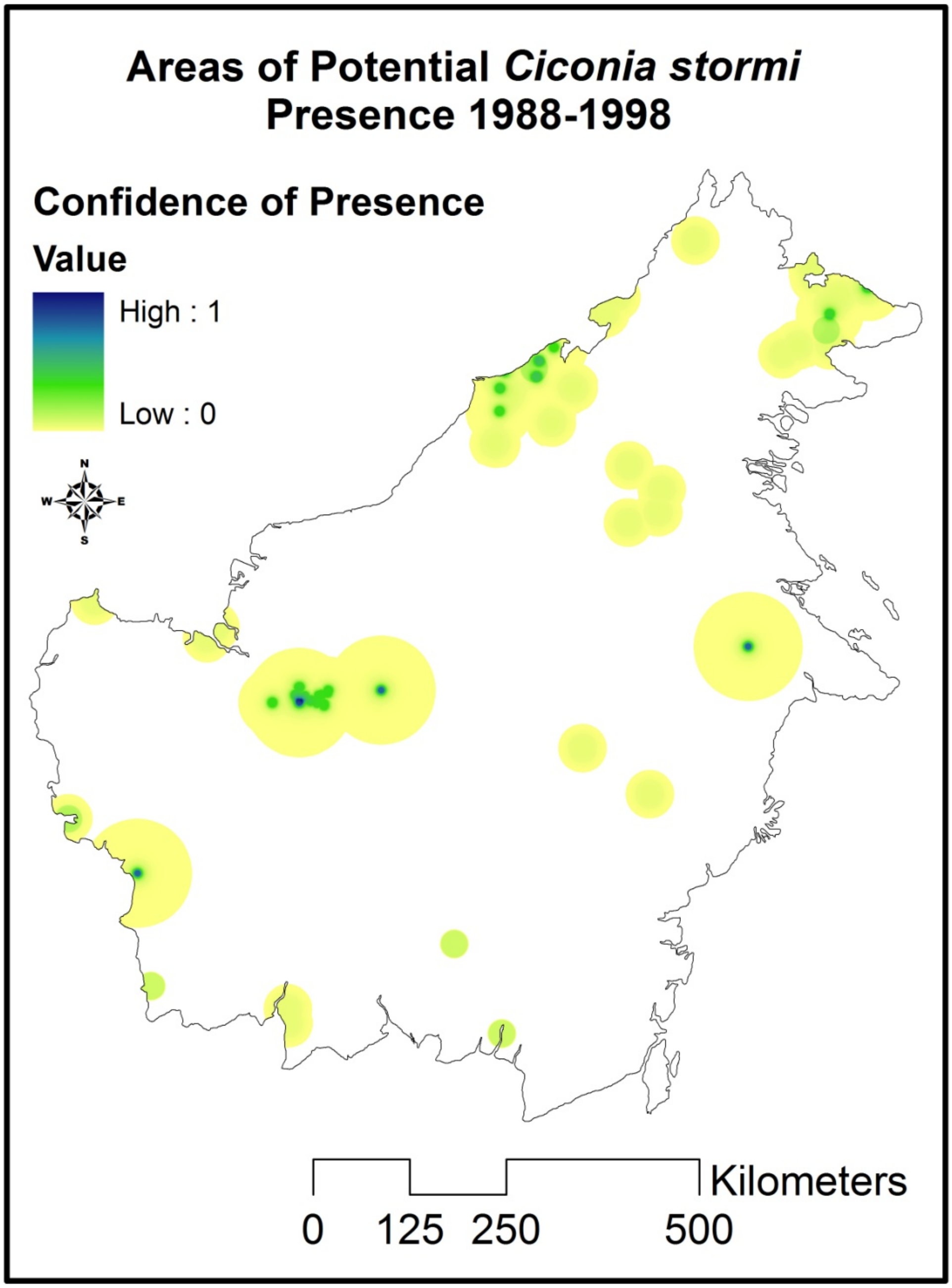

Figure 2.3. MCE results depicting areas of potential stork presence from 1988-1998.

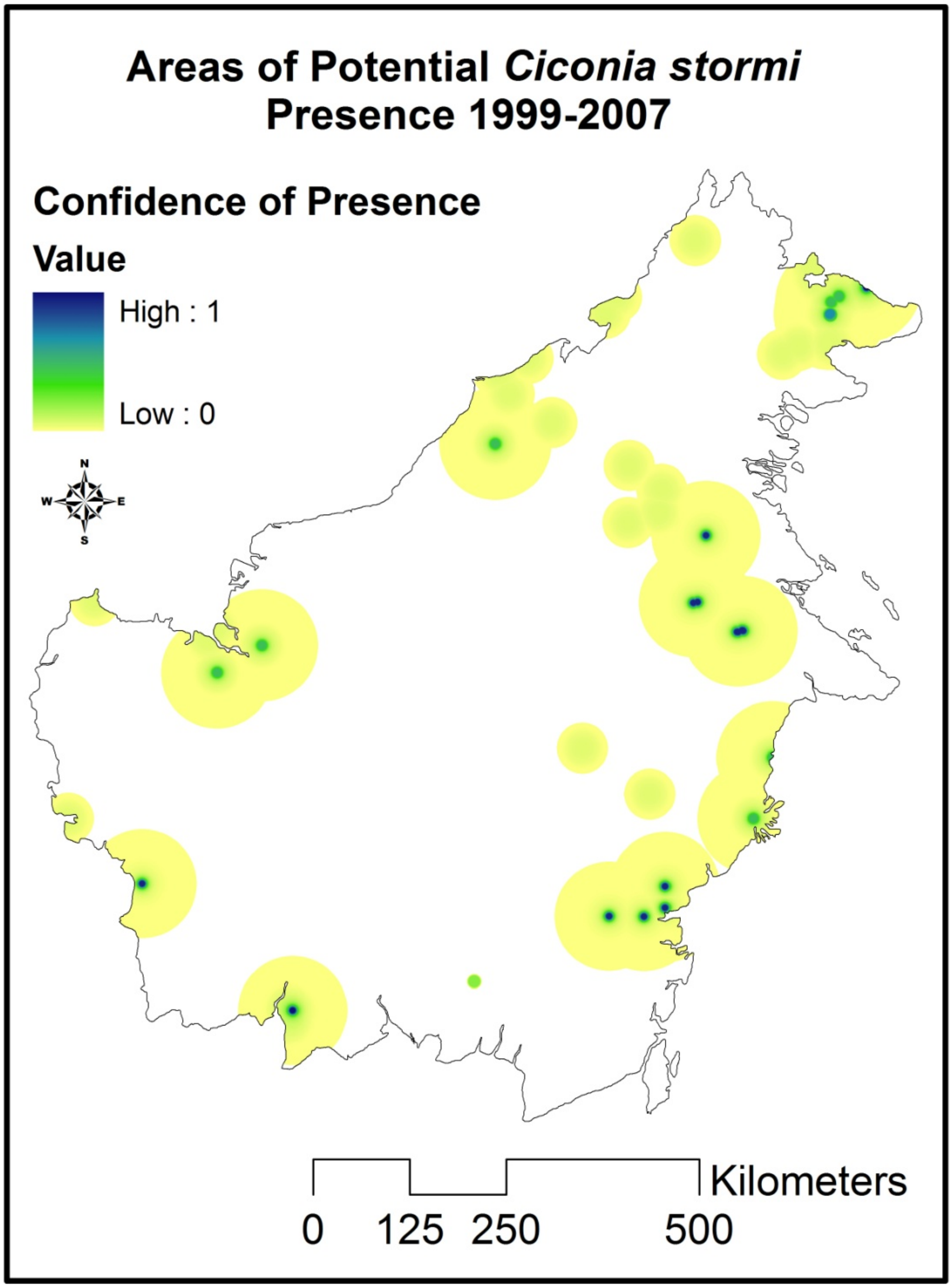

Figure 2.4. MCE results depicting areas of potential stork presence from 1999-2007.

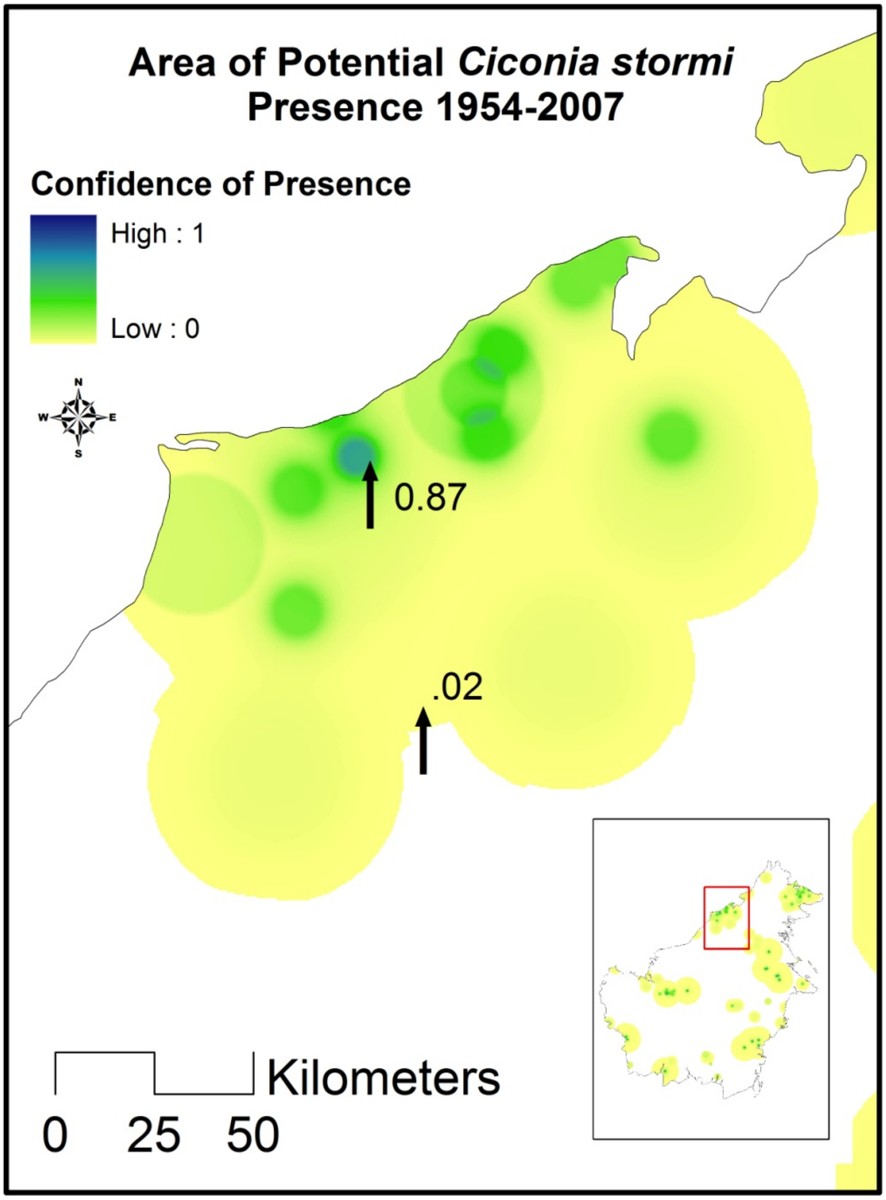

Figure 2.5. Numbers show the confidence in presence at the point indicated by the arrow. Sightings recorded with high precision are assigned a small area of high confidence and are surrounded by a large area of low confidence of presence. Areas where multiple independent sightings overlap result in higher confidence in presence, as these multiple sightings provide stronger evidence that a *Ciconia stormi* was present at that location.

The data of confidence in stork presence was then divided into different strata, with breaks created at the 20%, 40%, 60%, 80% and 90% confidence levels. Again, these are not true statistical probabilities, but are instead examples of levels at which one may choose to accept the data. If one accepts all locations at which there has been any likelihood of *Ciconia stormi* presence, one would accept all values of the data. If one chooses to accept only the locations where one can be highly confident of *Ciconia stormi* presence based on the adopted methodological approach, one might accept only the data greater than 90% confidence. This particular categorization scheme easily interpretable, however it is user-defined and other classifications could be constructed based on user needs or the natural distribution of the data.

Maps were then derived isolating the locations corresponding to the various levels of confidence. Maps of areas of *Ciconia stormi* presence at the 20 percent confidence level and 90 percent confidence level are shown below (figures 2.6a and 2.6b).

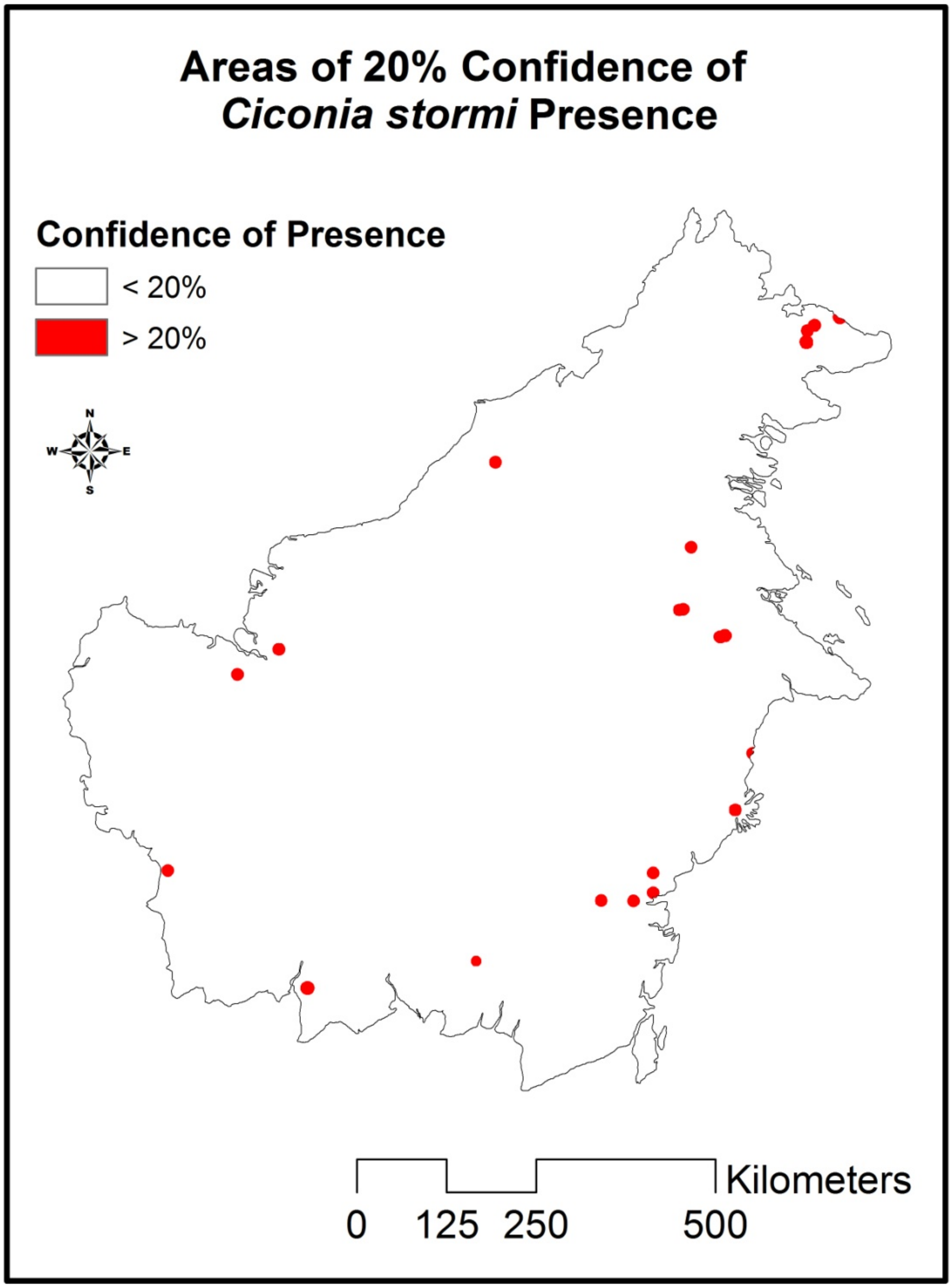

Figure 2.6a. Areas of potential *Ciconia stormi* presence at the 20 percent confidence.

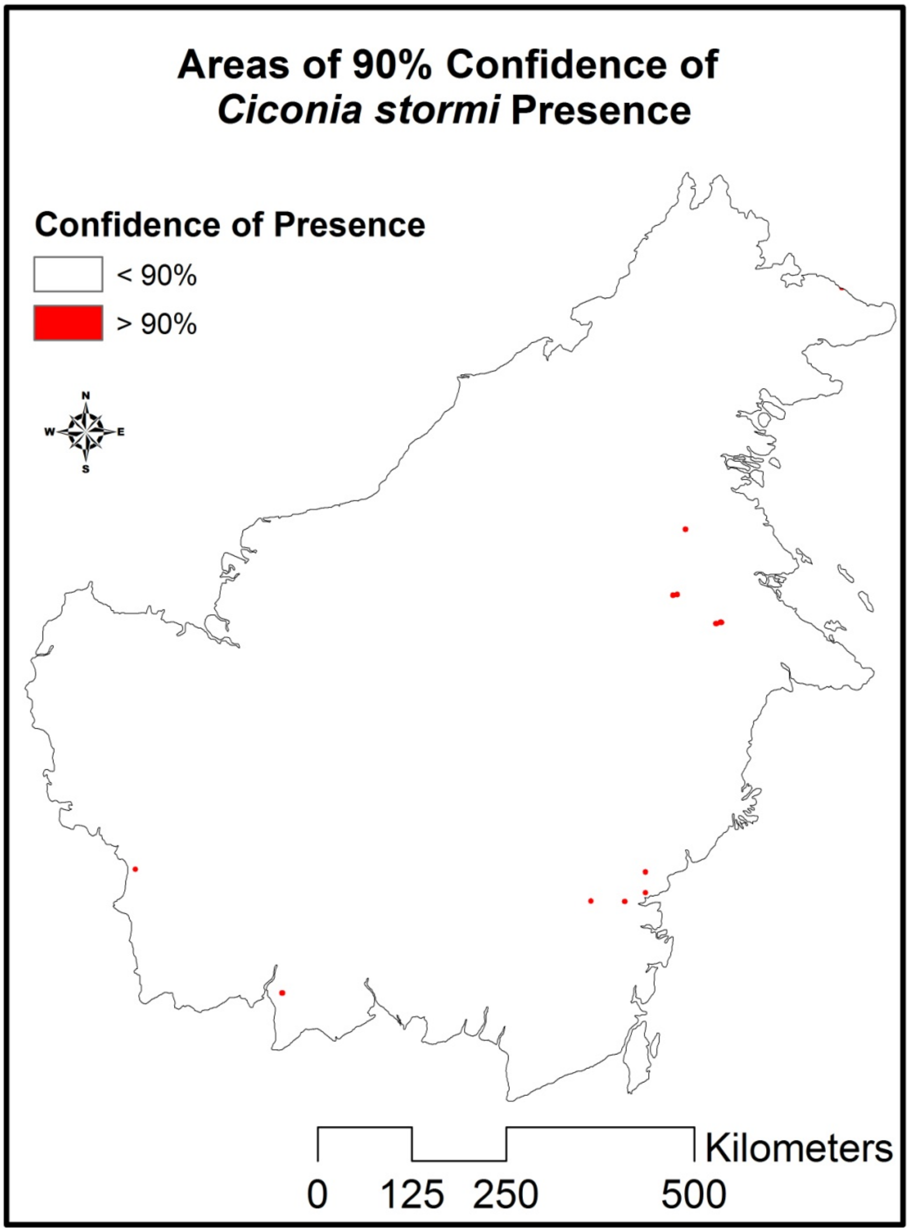

Figure 2.6b. Areas of *Ciconia stormi* presence at the 90 percent confidence.

2.4 Discussion

This chapter demonstrates a methodology that combined MCE analysis with the point-radius method for geo-referencing spatial locations to create a map depicting confidence of presence for a rare and elusive species. This method is an appropriate way to deal with the imprecision associated with geo-referencing species occurrence records. The resulting confidence map may be a more realistic depiction of known species presence than the standard methods of designating a specific point to represent the location of a presence record, or "representing the presence data by a polygon of size larger than the model cell" (Ottaviani et al., 2004). Representing presence with a single point or pixel fails to acknowledge the inaccuracies and imprecision associated with the often subjective process of selecting these points. The dominant approach of constructing presence polygons is prone to over-estimate species presence, particularly when the number of presence records is small. The methodology demonstrated here not only acknowledges these complications, but attempts to represent these spatial uncertainties using user-defined criteria.

It should be noted that the objective of the map created with the point-radius method is slightly different from the common application of the minimum convex polygon method. Minimum polygons generally attempt to identify the distribution of a species over space by interpolating between presence records. The point-radius method, as applied here, identifies spaces of potential presence, i.e., an area within which the target species may have been present. Nevertheless, both techniques identify an area in which the target species is likely to have occurred or occur. The point-radius method applied in this thesis improves upon the standard creation of binary polygons by explicitly basing the size of the area representing species presence on the various sources of imprecision embedded in records of species occurrence.

Additionally, the resulting areas of presence produced contain a continuous range of values rather than a binary output of presence non-presence. This allows for land managers to accept the data at different levels of confidence according to their specific needs. This flexibility is important because the level of confidence needed to make specific land management or population-management decisions may vary in different scenarios. The trade-off for accepting the data at higher levels of confidence is a reduction of the total amount of data available. For instance, if one is interested in analyzing factors related to areas of stork presence, there are 12,546,844 ha available for analysis if areas where one considers likelihood of stork presence > 0. However, if one chooses to restrict the analysis to areas where they can have a 95% confidence in stork presence, the analysis is limited to the 161,974 ha. By limiting the data in this manner, one is essentially reducing the sample size. At the same time, with limited funding and resources available for conservation activities, land managers may wish to ensure that their conservation efforts are targeted in areas with the highest likelihood of success, in which case accepting the data at higher levels of confidence may be the best strategy.

2.4.1 Assumptions and Limitations

This model deals with the lack of absence data by expressing the data in a fuzzy-format likelihood of presence, rather than creating hard-classifications of presence and absence. It is acknowledged that this methodology may not fully account for the lack of true absence data. Several other spatial models have been developed to account for instances were absence data is unavailable, but those models typically sacrifice large amounts of data, result in overly simplified models, or rely on large amounts of presence data which may not be available in cases of rare or elusive species (Anderson et al. 2002; Hirzel, 2002; Ottaviani et al. 2004; Kaschner et al. 2006). Furthermore, it is acknowledged, that presence only datasets like the one used in this study, are often the only data available for predictive modeling (Arau´jo and Williams, 2000). Others agree that true absence data is often very difficult or near impossible to obtain (MacKenzie et al., 2002; Mao et al. 2005), and working with presence-only data may be a more accurate portrayal of what is known about species distributions considering false absences can bias analyses (Hirzel et al., 2002).

Given the limitations in the type of data available, it is important to note the distinction between presence-only and presence-absence data when examining species-habitat relationships. One can make inferences about habitat suitability of *Ciconia stormi* based on where it tends to be seen although caution should be exercised when making assumptions about the unsuitability of habitat were the species has not been detected, because the lack of presence may simply be a result of insufficient sampling efforts (Solow, 1993). This model produces circular distance-based representations of potential presence within which confidence in potential presence varies isotropically (i.e. there is equal decay in all directions). However, *Ciconia stormi* is not limited to utilizing territory in a circular pattern, but likely adheres to topographical and land cover boundaries. Specifically, many sightings are associated with linear landscape features such as rivers. Ideally the areas of presence would have been created as functions of distance from a river. Unfortunately, detailed river data was not available at the time the analysis was performed, although future iterations of this model should focus on making the areas of potential presence conform more closely with the landscape features associated with those sighting records.

Additionally, the fuzzy functions and weightings used in this iteration of the model could be refined or adjusted, which may result in a more accurate final product. One way to improve upon the parameters specified in this thesis would be to incorporate expert knowledge concerning known habitat preferences of the species into the model to exclude areas of unsuitable habitat or weight more heavily potential presence in areas of preferred habitat. Due to the lack of substantiated knowledge regarding the habitat preferences of *Ciconia stormi*, this was not deemed appropriate at this time. However, this would be the next logical step of any future work on the species, or application of the point-radius method.

Chapter III

Percent Canopy Cover in Areas of *Ciconia stormi* Presence on Borneo

3.0 Background

Forest canopy cover affects a multitude of processes that impact wildlife behavior and distribution (Geiger, 1965; Burger, 1972; Weiss et al., 1999; Belisle et al., 2001; Lee et al., 2005; Pasher et al., 2007). Deforestation is an imminent threat to biodiversity, yet the precise amount of forest cover needed or utilized by many species remains largely unknown, despite the fact that forest canopy coverage is one of the few aspects of forest structure that land managers can both quantifiably measure and actively manage (Meffe and Carol, 1997; Weiss et al. 1991). Canopy cover affects wildlife habitat choices across multiple scales. Johnson (1980) distinguished four levels of habitat selection incorporated into scale dependent habitat use of avian species: first-order selection of the geographical distribution of a species; second-order selection of individual home ranges within the geographical distribution; third-order selection of habitat components within home ranges; and fourth-order selection of a specific sites, such as a locations used for foraging or nesting. Conservation efforts should address species habitat needs at each of these scales. Even if habitat is protected at one scale, failure to protect sufficient habitat at other scales may still result in species extinction.

Amongst forest birds, understory specialists tend to be those most affected by forest loss (Thiollay, 1992). Some forest interior birds rely on a mixture of micro-habitats within forests, and may actively seek out or avoid large gaps in the canopy within areas dominated by heavy canopy cover (Pasher et al. 2007). The composition and abundance of these microhabitats may be affected by disturbance from logging activities and fire.

Storm's stork (*Ciconia stormi*) is an endangered species known to utilize forest interior habitats in Indonesia (Birdlife International, 2001), a country that is experiencing some of the highest rates of deforestation in the world (Curran et al., 2004; Fuller et al., 2004). Canopy cover loss associated with logging is of particular importance to *Ciconia stormi* because of the corresponding impacts on stream biodiversity and function (Danielsen et al., 1997). Increased sedimentation, elevated water temperatures, enhanced nutrient loads, and changes in the relative availability of aqutic food resources have all been linked with loss of canopy cover following logging (Likens et al. 1970; Webster et al. 1983; Waters 1995, Harding et al. 1998). These impacts result in a loss of macro-invertebrate species richness (Benstead et al. 2003; Gibon et al. 1996), a critical trophic level for stream food-web dynamics. Assessing canopy cover provides an efficient way to indirectly measure stream biodiversity and function at multiple scales and can be applied in situations where biological assessments (macro-invertebrate surveys) are not feasible.

Hemispherical photography is one method that has been widely used to measure canopy structure (Rich, 1990; Chen et al. 1991; Leblanc et al. 2005), although it has seldom been used to quantify canopy cover and canopy gaps in relation to bird habitat (Pasher et al., 2007).

3.1 Chapter overview

Percent forest cover is one of the more easily quantified aspects of forest structure, yet information is needed to identify the amount of canopy cover necessary to ensure various ecological processes and relationships persist. This chapter employs a multi-scale analysis of the percent forest cover utilized by *Ciconia stormi* across Borneo. Percent forest cover in areas of potential *Ciconia stormi* presence across Borneo was extracted from a MODIS 500m soft classification product. Canopy cover was also extracted from individual MODIS pixels corresponding to precisely geo-located sightings (those marked with a GPS unit), and in 6 kilometer buffers from those sightings. A higher resolution soft-classification product from 30 meter Landsat imagery is used to find the percent canopy cover associated with a subset of the precise presence records. Finally, analytical detail is further increased by identifying the percent forest cover from hemispherical photographs at locations where *Ciconia stormi* was seen feeding or roosting. These data provide information regarding the habitat needs of *Ciconia stormi* across multiples scales. These findings can assist land managers regarding the extent to which forests may be disturbed before they are no longer viable habitat for the species.

3.2 Data and Methods

3.2.1 Study area

Ground data on tree cover was collected in the Berau district of East Kalimantan on Indonesian Borneo (Figure 3.1) during June-August 2007. The Berau district is characterized by large blocks of disturbed and undisturbed lowland dipterocarp forests, derived grasslands, oil palm plantations and small amounts of urban area. The topography is undulating and steep in places and is dissected by numerous rivers and streams. There is little large scale agriculture other than oil palm plantations, although there has been a long history of timber extraction (selective logging) and mining in the region. The area was selected for study owing to the frequent sightings of *Ciconia stormi* (Danielsen et al. 1997), as well as the active forest conservation program of The Nature Conservancy, which has been operating in the region since 2001 and provided logistical support for data collection.

3.2.2 Hemispherical Photographs

Hemispherical photography has been widely used to measure percent tree cover (Souza and Martins 2005; Kobe and Hogarth, 2007; Schedlbauer et al., 2007), although the application of this technique for verifying remotely sensed data is less extensive (Lovell et al., 2003). The method involves use of an upward-pointing lens with 180 degree field of view (FOV), which provides a complete view of the forest canopy, stems, trunks and lianas and other plant parts that may be visible within the lens FOV. The technique has been used widely to measure various canopy parameters such as leaf area index (Jonckheere et al., 2004), as well as habitat quality for various species of insects and vertebrates (Weiss et al., 1991; Rich et al. 2007).

Hemispherical Photographs were collected from June to August 2007 using a Sigma f4 fisheye lens and Nikon D70 digital SLR camera. Systematic sampling was done in the following forest succession strata: recently heavily logged forests (logged less than 5 years before present (ybp), older heavily logged (greater than 10 ybp), selectively logged stands of assorted ages, and undisturbed forest (table 3.1). Chapter four further addresses the tree cover conditions in each of these strata in relation to habitat needs of *Ciconia stormi*. Photographs were also obtained at or near (< 25m) several locations where *Ciconia stormi* had been recently sighted foraging on the ground. Care was taken to reduce spatial autocorrelation by ensuring that sample sites were separated by more than the size of tree crowns. Thus, photographs were collected every fifty meters along transects, a length greater than the average distance between Borneo tree crowns (Cannon et al., 1994). Attempts were made to adhere to 1km straight line transects, although in many instances the topography, rivers and dense vegetation made this difficult. The Nikon D70 was mounted on a tripod with its major axis pointed north and leveled at a height of 1.5 M. A bi-plane level was used to ensure the camera sensor was level to the horizon. Because improper exposure can result in measurement error (Macfarlane et al. 2000; Zhang et al. 2005) a series of bracketed exposures was taken to ensure a range of exposures and the focus was set on infinity. All locations for which a satellite signal was obtainable were marked with a GPS. If no signal was available, azimuth and distance from the last known location were recorded. Distance was measured with a tape measure. Because this measurement is influenced by topography, while those taken with the GPS were not, an estimate was made in the field to take topography into account so that samples were taken as close to 50 meters apart in a straight-line measurement as possible.

Table 3.1. Land use and forest type at sites were hemispherical photographs were obtained. Land use history was provided by researchers at The Nature Conservancy active in the study area, and forest type was determined from field observations. The locations of transect sites are shown in figure 3.1.

Site Name	Land Use History	Forest Type
Lesan Conservation Area	Selectively logged throughout the 1980's.	Lowland Dipterocarp forest
Long Gie	Mixed use of heavy commercial logging (1996-1997), and selective community extraction.	Lowland Dipterocarp forest and community agricultural
Long Duhung	Mixed use of heavy commercial logging (2003), and selective community extraction.	Lowland mixed Dipterocarp forest
Long Lamcin	Unlogged Forest with logging road network present	Lowland mixed Dipterocarp forest
Long Sului	Small-scale community use	Primary mixed Dipterocarp forest
Long Oking	Clear cut and heavily logged (1980-1984), community agriculture and extraction	Lowland mixed Dipterocarp forest
Long Laai	Clear cut and heavily logged (1980-1984), community agriculture and extraction	Lowland mixed Dipterocarp forest
Labanan	Current agricultural use and heavy timber extraction	Disturbed lowland forest
Wahea	Commercial Logging activities	Mixed Dipterocarp forest

Digital hemispherical photographs were processed using Gap Light Analyzer (GLA) Version 2.0 (Frazier et al. 1999). The grey-scale pixels within each photograph were converted to a binary image such that each pixel was classified as either sky or canopy according to a user-defined threshold value. Determination of the threshold value can cause biases in analyzing hemispherical photographs due to operator subjectivity (de Freitas and Enright 1995; Roxburgh and Kelly 1995). However, this bias can be minimized if the user is properly trained (Robinson and McCarthy 1999). In addition, most standard digital SLR cameras such the one used in this study produce a rectangular image that truncates the top and bottom of the photographs. Therefore, the circle drawn within the canopy field of view for each photograph corresponded to a zenith angle range of 0-60 degrees. To avoid large errors in cover estimation caused by lens distortion, the Sigma lens was also calibrated using a third-order polynomial equation, which corrects for changes in projected area with zenith angle (Jonckheere et al. 2005). However, cover estimates between calibrated and uncalibrated photographs did not differ by more than 1-2 percent.

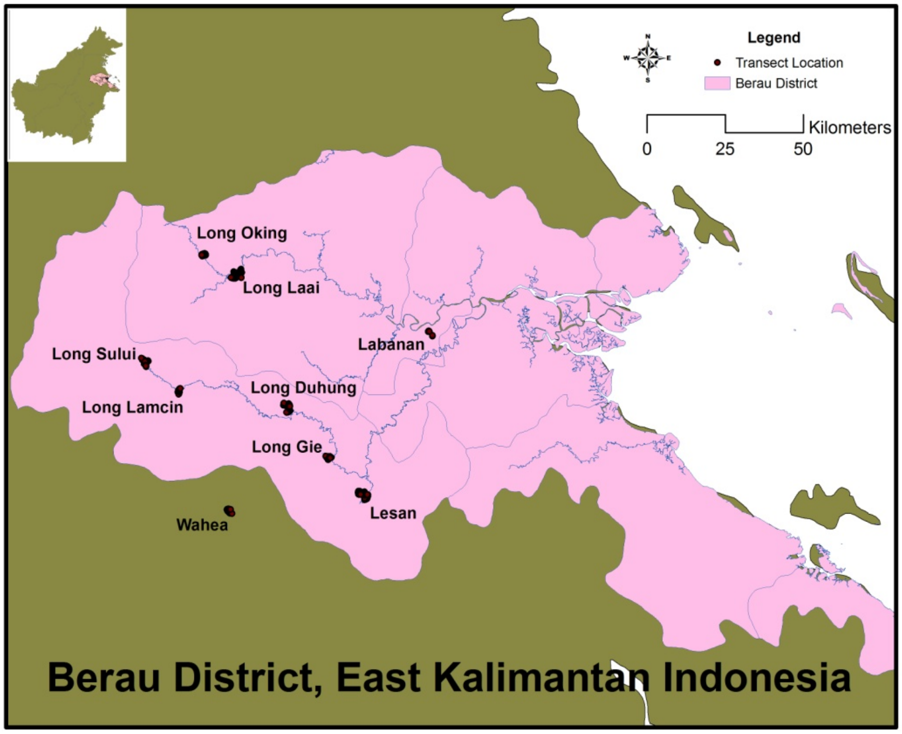

Figure 3.1. Locations where hemispherical photographs were collected.

3.2.3 Percent Tree Cover Maps

Accurate tree cover data are essential to analyze *Ciconia stormi's* relationship to forest cover. One of the most widely used tree cover products is the MODIS derived VCF (MOD 44B) (Hansen et al., 2002) available from the Global Land Cover Facility (http://glcf.umiacs.umd.edu/data/vcf/) (Zhang and Kondragunta, 2006; Morton et al. 2005; Miles et al. 2006; Schwarz and Zimmermann, 2005; Dymond et al.,2004; Carabajal and Harding, 2006; Schmullius et al. 2006). However, this product has been shown to saturate at 80 percent tree cover for values starting around 60 percent ground-measured tree cover (figure 3.3) (Fuller and Berdie, submitted). Because *Ciconia stormi* utilizes areas of dense forest-cover (Birdlife international, 2001), the MOD 44B product was not appropriate for this study. Therefore, a second tree-cover product for Borneo was generated that better represents ground-measured tree cover over 80 percent. This alternate product (hereafter referred to as MODNN3) was created using a series of 2004 MODIS composite images also obtained from the Global Land Cover Facility (table 3.2). These 32-day reflectance composites from MODIS bands 1-7 were further composited using a minimum value rule to create cloud-reduced 96-day composites for each band (Fuller and Murphy, 2005). All seven bands were used in an artificial neural network soft-classification module in Idrisi software (Eastman, 2006) implemented as a multi-layer perceptron (MLP). Training sites were selected using SPOT quicklook images of the study area produced at 100m resolution by the Centre for Remote Imaging, Sensing, and Processing at the National University of Singapore (http://www.crisp.nus.edu.sg/) and the MLP inputs such as learning rate, hidden nodes, accuracy rate, and iterations were adjusted experimentally to produce a set of soft-classification outputs for comparison with the canopy hemispherical photographs. A total of 185 hemispherical photographs were processed for percent canopy cover, falling into 81 separate MODIS pixels. Thirty six of the 81 VCF pixels had only one hemispherical photograph contained within in it. The remaining pixels had between 2 and 7 photos assigned to them, with one pixel having 16 hemispherical photos fall within its bounds. Where multiple photos fell within a single MODIS pixel, the percent canopy cover from those photos was averaged.

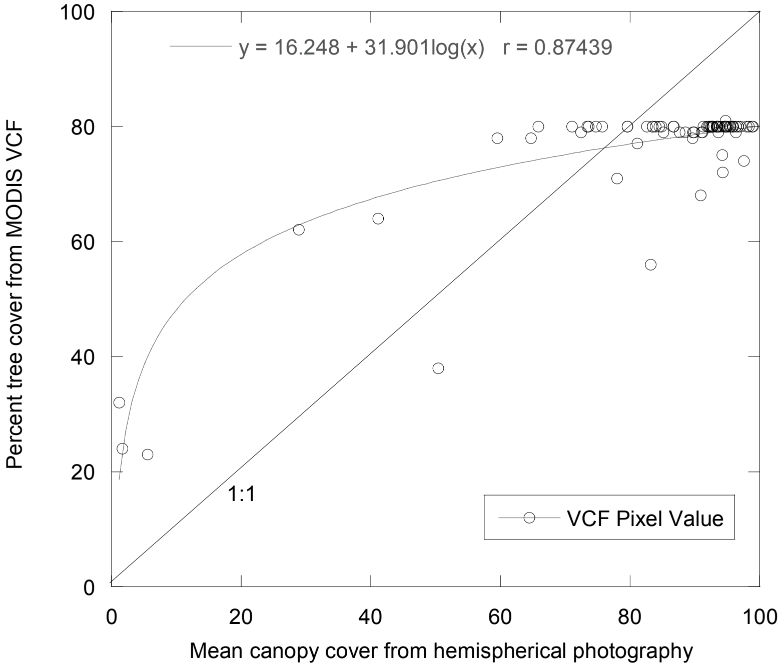

Figure 3.2. Percent tree cover from 2005 MODIS VCF plotted against hemispherical photographs taken in 2007 for the full range (0-100 percent) of cover values.

An orthorectified Landsat image (P117 R059) from the year 2000 was obtained for an area containing a relatively large number (19) of precise *Ciconia stormi* sightings marked with a GPS (table 3.2). Obtaining tree cover values from the Landsat product provides more detailed information regarding the relationship between *Ciconia stormi* and tree cover than analysis of MODIS data alone. The image was processed following the same MLP procedures as the MODIS image as described above, although Digital Globe Quickbird 2.4m images were used as training sites in lieu of SPOT quicklook data. A total of 139 hemispherical photographs were used to verify the final product (hereafter referred to as LandNN4). No Landsat cell had more than a single hemispherical photograph fall within it. Unfortunately, because of the high cloud-cover in the area, and the failure of the scan-line corrector in 2003, the most recent cloud-free image available was from 2000. This resulted in a seven year temporal gap between the date of image acquisition, and the acquisition dates of hemispherical photographs used to verify the data.

Finally, a digital elevation model was downloaded from the USGS (http://edc.usgs.gov/products/elevation.html) (table 3.2). The data was obtained as part of the Shuttle Radar Topography Mission (SRTM) and is provided at a resolution of 3 arc second (approximately 90 meters). This information was used to examine the relationship between elevation and confidence in *Ciconia stormi* presence.

Table 3.2. Data used to create soft-classification forest-cover products and examine relationship between elevation and confidence in presence.

Data Set	**Source**	**Year Acquired**	**Spatial Resolution**	**Spatial Extent**
MODIS	Global Land Cover Facility	2004	500 m	Windowed to Borneo (108° 30’ E to 119° 17’ E; 4° 15’S to7° 35’N)
Landsat (P117 R059)	Global Land Cover Facility	2000	28.5 m	115° 28’ E to 117° 26’ E; 0° 32’N to 2° 23’N
DEM	USGS Shuttle Radar Topography Mission	2000	3 Arc Second (Approx. 90m)	Windowed to Borneo (108° 30’ E to 119° 17’ E; 4° 15’S to7° 35’N)

3.2.4 Data Processing

The areas of stork presence from 1999-2007 were divided at different levels of confidence: any confidence in presence, 20%, 40%, 60%, 80% and 90%. As discussed in chapter two, these delineations are not true statistical likelihoods and are somewhat arbitrary, but they represent a wide range of levels at which the data could be accepted. These layers were then overlayed onto the MODIS soft classification product, and the average canopy cover values were extracted. This methodology was also used by Grubb et al. (1997), who measured percent canopy cover within concentric analysis zones of various radii from spotted owl nests (Grubb et al., 1997). Because of *Ciconia stormi*'s affinity for water, larger water-bodies such as lakes or wide rivers threatened to distort forest cover results, i.e. sightings over water may bias results by underestimating tree cover, even if those sightings occurred within a forested area. Therefore, water-pixels identified from a separate 2004 MODIS hard classification product were masked out of the image prior to analysis.

The percent tree cover from the MODNN3 and LanNN4 pixels that corresponded to locations of precise sightings of *Ciconia stormi* was recorded. A six kilometer buffer was then created for each of these sightings to approximate a potential home-range as identified by Danielsen et al. (2007). The average percent tree cover in these buffers was then extracted from the Landsat 2000 and MODIS 2004 soft classifications. Finally, locations of *Ciconia stormi* marked with GPS coordinates for which hemispherical photographs were available within 25m were identified and the mean tree cover was calculated from the corresponding photographs.

A random sample of 5,000 points was selected from the areas of potential *Ciconia stormi* presence. Scatter plots and a regression model were developed to examine the relationship between percent tree cover from the MODNN3 product and confidence in stork presence, as well as elevation and confidence in stork presence. It is important to note that it is not feasible to conduct formal hypothesis testing because of the highly derived nature of the confidence in presence data. Finding the best fit for the data is instead helpful to better explore relationships between the two variables, rather than deduce the amount of one variable explained by the other. A qualitative assessment was then used to assess the level of canopy cover and elevation at which confidence in presence showed the greatest increase.

3.3 Results

3.3.1 Tree cover products

Figure 3.3a displays the results of four-different MODIS-based tree cover products (three based on the MLP neural network, and the VCF or MOD44B) over the range of 60-100 percent tree cover, which is considered one criterion of the definition of “forest” by the International Geosphere-Biosphere Program (Giri et al. 2005; Fuller 2006). This figure shows significant scatter around the 1:1 line, although there is a clear cluster of points on the line for the MODIS neural network products (particularly NN3) in the upper end of the range from 90-95 percent.

Further analysis to address one possible cause of the outliers in the NN MLP products is shown in Figure 3.3b, in which only mean values from hemispherical photographs are plotted for sites with relatively low (<10 percent) standard deviations. Such sites had relatively low local variation in tree cover (i.e., relative spatial homogeneity) and therefore may be more easily related to 500m pixels from MODIS. In this figure the scatter around the 1:1 line is clearly reduced, which suggests that local spatial variation may have an influence on the relationship between averaged canopy cover from hemispherical photography and percent tree cover from MODIS. In this figure, too, saturation may be discerned from 75-100 percent cover, although it is difficult to know if this pattern is due to the paucity of ground observations in the ranges of 60-75 percent and 75-90 percent. Nonetheless, as a representation of tree cover, the products from the MLP algorithm appear superior to the MOD 44B VCF for analyzing areas of dense, intact forest that have high conservation value.

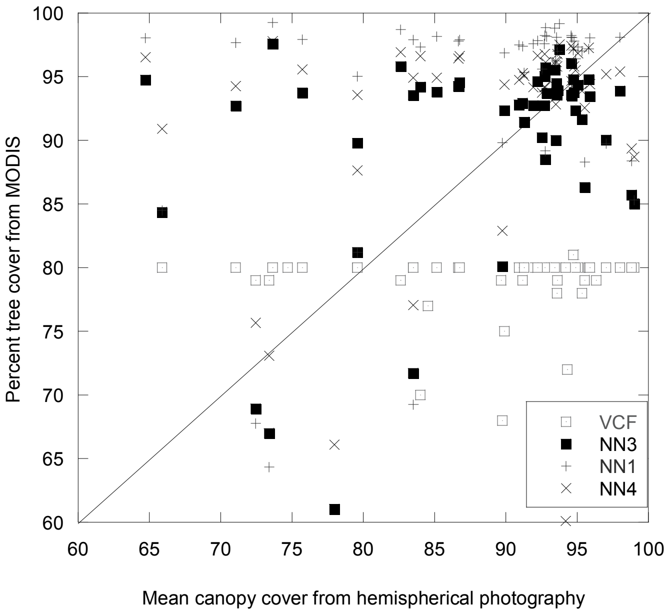

Figure 3.3a. Four MODIS-based products (NN = those from the neural network algorithm; VCF = MOD44B) plotted against mean canopy cover from hemispherical photographs.

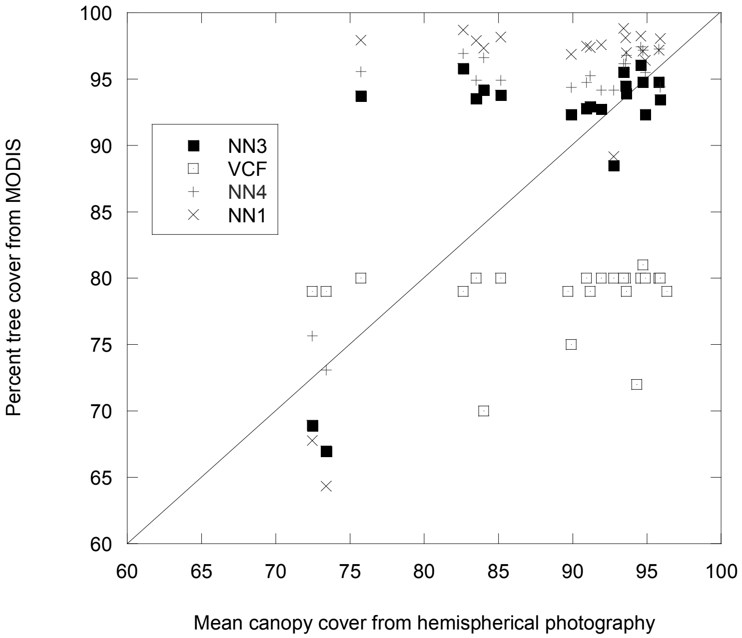

Figure 3.3b. The subset of the data plotted for field sites where the standard deviation (and hence local spatial variation) was less than 10 percent.

Percent canopy cover from hemispherical photographs was plotted against the percent tree cover values for each of the 15 MLP iterations run on the Landsat data. Visual assessments of the plots and the final map products showed the fourth trial to have the best correspondence to the hemispherical photographs. Similar to the MODNN3 product, the LandNN4 product had significant scatter around the 1:1 line (Figure 3.4). The outliers may in part be attributed to the seven year temporal lag between the Landsat image and the verification data. In areas that had experienced disturbance shortly before 2000, forest regeneration will result in higher values from the hemispherical photographs than the LandNN4 as vegetation would have been present during the time of hemispherical-photo acquisition, but not when the remotely sensed data was obtained. During the acquisition of hemispherical photographs, the camera is only 1.5 meters off the ground. This means that fast-growing pioneer species that take advantages of gaps in the forest created by disturbance events are apt to influence values obtained from the photographs by obscuring the sky above the camera.

Conversely, forest disturbance that occurred shortly before 2007 (as was the case for several areas where hemispherical photographs were collected) may result in lower values in hemispherical photographs than the LandNN4, because the vegetation present when the remotely sensed data was obtained may not be present when the photos were taken on the ground. Both types of outliers may be present and influence the relationship between hemispherical photographs and the soft-classification product in this analysis.

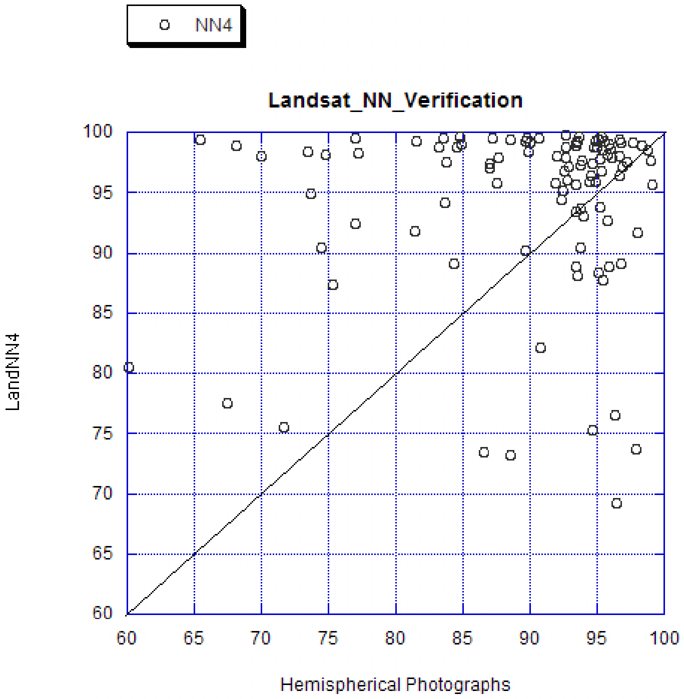

Figure 3.4. Percent canopy cover from hemispherical photographs plotted against the LandNN4 soft classification.

3.3.4 Tree cover in areas of *Ciconia stormi* presence

MODIS Data

The average amount of tree cover extracted from the MODNN3 product in areas of potential *Ciconia stormi* presence ranged from 70.1 percent to 73.0 percent depending on the level of confidence examined (table 3.3). The percent tree cover in areas of potential stork presence was negatively skewed (Skewness = -1.18), indicating the majority of this area is covered by higher values of percent tree cover than that 70-73 percent range (figure 3.5). Closer examination reveals that 67.1% of the area of potential stork presence has over 70% tree cover and 57.8% of the area has greater than

80% tree cover. A second smaller peak of pixels with less than 5% tree cover decreases the overall average percent tree cover in those areas.

Table 3.3. Average percent tree cover in areas of potential *Ciconia stormi* presence at various levels of confidence from MODNN3.

Confidence Level of Presence	Mean Forest Cover	Standard Deviation	Range
Some Potential of Presence (0-100)	70.1	30.3%	0 – 98.6
20	71.3%	30.0%	<1 – 98.1
40	70.8%	30.3%	<1 – 98.1
60	73.0%	28.6%	<1 – 98.1
80	71.9%	29.0%	<1 – 98.1
90	72.3%	28.8%	<1 – 98.1

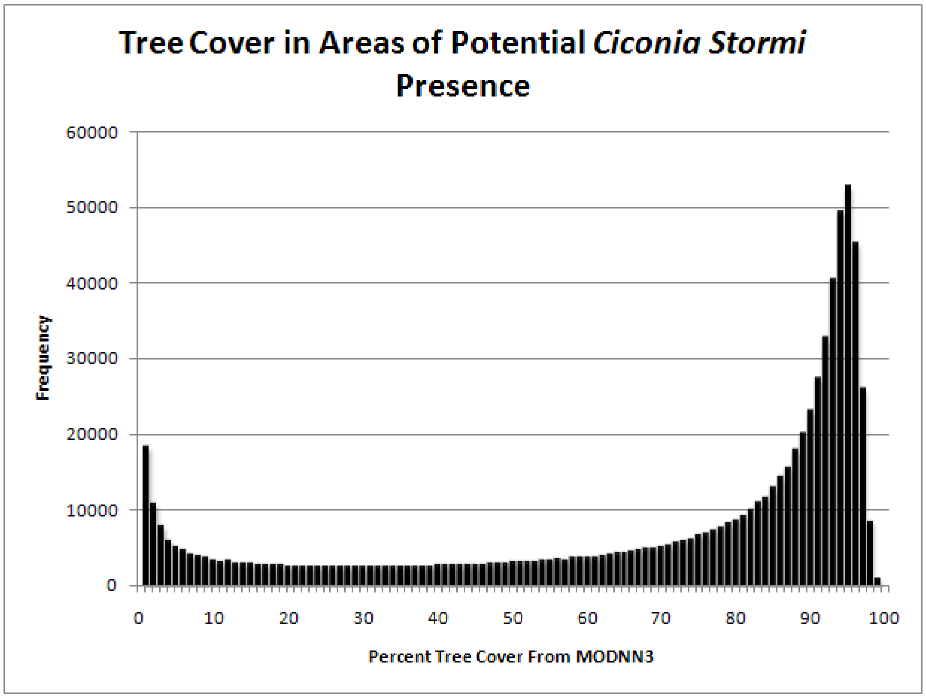

Figure 3.5. Histogram of percent tree cover in areas with of potential *Ciconia stormi* presence from MODNN3.

Within six km of *Ciconia stormi* sightings recorded with a GPS, the average percent tree cover from the MODNN3 product was 68.9 percent (table 3.3). The distribution was also skewed (-1.12) indicating that over half of the distribution is higher than the mean (figure 3.5). Areas within six km of *Ciconia stormi* sightings contained a large amount of area with less than five percent tree cover. These results showed little difference from a random selection of six km buffers. These randomly selected areas had an average of 70.2 percent and ranged from 0-98.2%.

The average percent tree cover extracted from individual MODIS pixels at locations of precise sightings increased to 73.8 percent (table 3.3). Examination of the histogram also reveals a trend of these locations to be covered by over 70% tree cover (22 out of 30 instances). Additionally, there are four sightings that occurred in areas with less than 50 percent tree cover, and two sightings that occurred at locations with less than 10 percent tree cover.

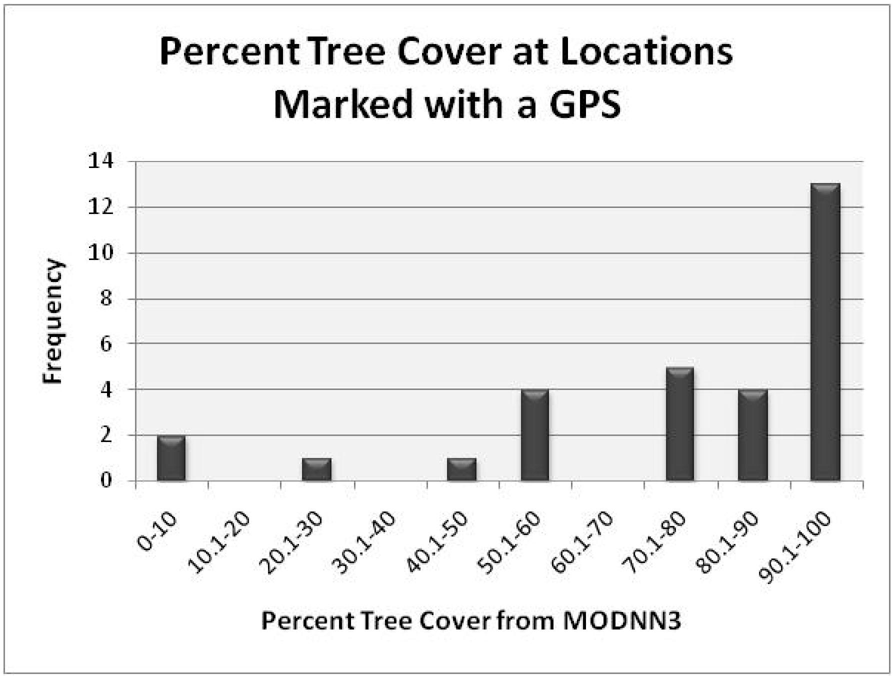

Figure 3.6. Tree cover values extracted from the MODNN3 product for individual pixels that correspond to locations where a *Ciconia stormi* was seen and the location marked with a GPS unit.

Landsat Data

Canopy cover within 6 km of precise *Ciconia stormi* sightings extracted from the LandNN4 product had a mean value of 86.1%. Similar to the MODIS product, there was tree cover conditions varied widely within this buffer, ranging from 0-99 percent.

Individual pixel values from the LandNN4 image averaged 90.5% at locations of precise *Ciconia stormi* sightings (n = 19). These values also ranged from 0-99 percent, with a standard deviation of 23.5 percent (Table 3.4, Figure 3.7).

Within those data, there were 12 sightings of *Ciconia stormi* either feeding along the river or roosting in a tree. The percent tree cover at these locations increased to 95.7 percent, and ranged from 90.7 - 99.2 percent with a standard deviation of 3.1 percent (Table 3.4).

Hemispherical Photographs

The percent tree cover from hemispherical photographs at locations where *Ciconia stormi* was seen on the ground (n = 10) ranged from 74 percent to 92.6 percent (table 3.4). The average percent forest cover at these locations was 85.6% with a standard deviation of 6.2% (figure 3.7).

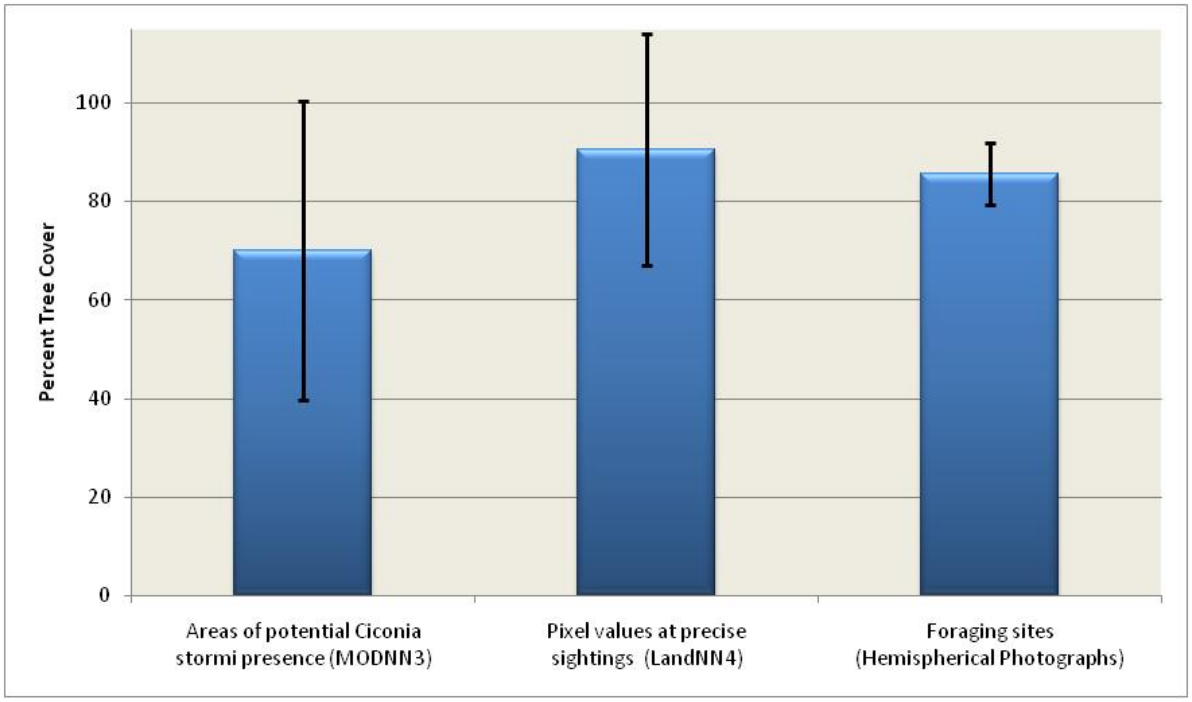

Figure 3.7. The percent canopy cover at different spatial resolutions. The standard deviation of canopy cover (black bars) decreased as the resolution of analysis increased.

Table 3.4. Average percent canopy cover at *Ciconia stormi* sighting locations (those marked with a GPS) from MODIS imagery, Landsat imagery, and hemispherical photographs. Also shown are canopy cover of the 6km surrounding those locations from the MODNN3 and LandNN4 products.

Source	Mean Forest Cover	Range	Standard Deviation	Range
6 km buffer from precise sightings (MODNN3)	**68.9%**	98.1%	31.2%	0-98.1%
6km buffer from precise sightings (LandNN4)	**86.1%**	99.9%	24.1%	0-99.9%
Pixel values at locations of precise sightings (MODNN3)	**73.1%**	97.6%	30.6%	0-97.6%
Pixel values at locations of precise sightings (LandNN4)	**90.5%**	0-99.2%	23.5%	0-99.2%
Pixel values at locations where *Ciconia stormi* was seen feeding/roosting (LandNN4, N = 12)	**95.7%**	8.5%	3.1%	90.7-99.2%
Canopy cover where *Ciconia stormi* was seen feeding/roosting (Hemispherical Photographs, N=10)	**85.6%**	18.2%	6.2%	74-92.6%

There was no strong fit between the overall level of confidence in *Ciconia stormi* presence and the MODNN3 tree cover product (figure 3.8). The best fit was an exponential relationship, although the r value was still low, 0.025, only slightly higher than a linear relationship. However, a qualitative assessment shows an increase in the number of high confidence presence values beginning around 75 percent tree cover. This suggests that areas where there is a high confidence of stork presence are more likely to have over 75 percent tree cover than lesser amounts of canopy closure.

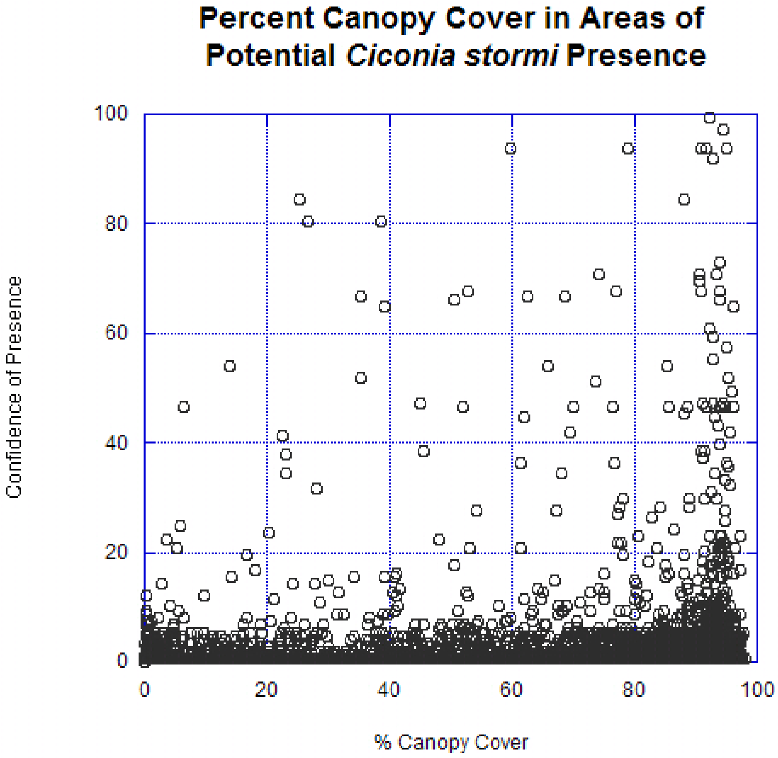

Figure 3.8. Scatter plot showing the relationship between tree cover values extracted from the NN3 product and the corresponding confidence in *Ciconia stormi* presence values.

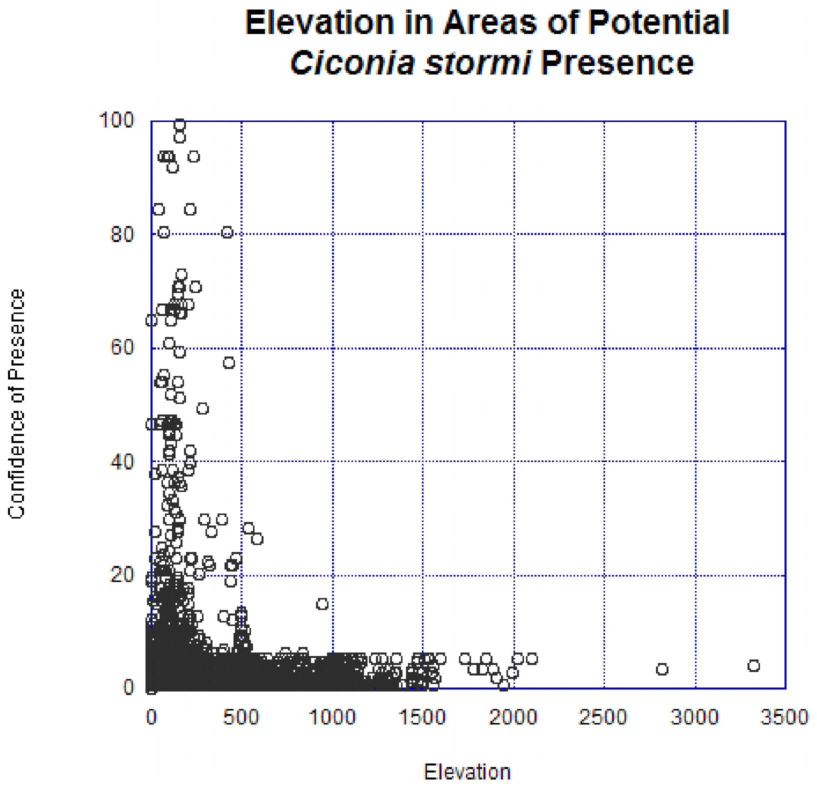

Figure 3.9. Scatterplot depicting the relationship between confidence in stork presence and elevation.

There was no linear relationship between confidence in stork sighting and elevation (r = -.036), although only low-confidence presence areas (<20 percent) were found higher than 550 meters (figure 3.9). This suggests that 550 meters may approximate the upper end of *Ciconia stormi's* topographic range.

3.4 Discussion

Forest canopy cover is known to influence avian behavioral strategies such as home range behavior (Belisle et al., 2001), nest placement (Pasher, 2007) and foraging success (Walther, 2002). Areas of potential *Ciconia stormi* presence are characterized by at least 70 percent canopy cover as determined from the MODNN3 product, with the majority of this area having over 80 percent tree cover (figure 3.5). Specifically, 75 percent canopy cover was identified as an apparent threshold, above which high confidence in presence areas occurred more frequently. In other words, confidence of *Ciconia stormi* presence is higher in areas with more than 75 percent canopy cover than areas with less than 75 percent canopy cover, indicating this level of canopy cover may be important for maintaining the species' population. However, at increased spatial resolution, the average percent tree cover from 30m Landsat data at locations of precise sightings is over 90 percent. As the resolution is further increased to examine canopy conditions *Ciconia stormi* was seen foraging or roosting, the average percent tree cover extracted from hemispherical photographs was 84 percent.

3.4.1 Landscape Scale Habitat

3.4.1a Elevation

Despite the lack of a linear relationship between elevation and confidence in stork presence, the data supports the conclusion that the species is restricted to lowland forests (below 600 meters). A threshold value of 550 meters was identified, below which the confidence in stork presence was substantially higher than in areas over 550 meters. There is a high-precision record in the dataset of a group of four individuals sighted at 600 m altitude. There have been other instances of the species being recorded at high altitudes, including one sighting made at 1600 m, although it was noted that this was most likely a displaced individual as the lowlands were on fire at the time (Birdlife International, 2001), and overall these high altitude records are scarce. Again, the circular isotropic patterns of the confidence of presence data may have included areas of high altitude not actually utilized by *Ciconia stormi*. However, no areas display over a 25 percent confidence in presence occurring higher than 600 meters (figure 3.8), indicating the species is a lowland specialist.

3.4.1b Foraging Strategies

Forest loss is also known to impede landscape-scale movements of forest birds (Belisle et al., 2001) as well as increase the risk of predation (Gotmark and Post, 1996). In tropical rainforests, birds have three main predators: bird raptors, mammals, and snakes (Trail 1987; Gomez and Thery, 2007). In the case of *Ciconia stormi*, a relatively large bird known to soar at high altitudes when searching for feeding areas (Birdlife international, 2001), the risk of predation seems less likely to limit movement.

Instead, *Ciconia stormi* movements across the landscape may be tied to seeking out high-quality habitat. For forest birds in general, the inability to locate areas of high-quality foraging sights can lead to increased mortality, particularly after the natal stage (Beauchamp et al., 1997; Anders et al., 1998). Birds are also known to have higher breeding success if they are able to evaluate larger areas of quality habitat in which to settle and forage (Reed et al., 1999; Schjørring et al., 1999). Reduction in canopy cover likely reduces the amount of high-quality habitat available to *Ciconia stormi* directly, as well as through the negative impacts on stream biodiversity and function associated with forest cover loss (Benstead et al., 2003).

A wide body of research on the feeding ecology of various organisms is based on optimal foraging theory (MacArthur and Pianka 1966; Emlen 1966; Kamil et al. 1987; Stephens and Krebs 1986). Studies have investigated stork foraging strategies and selection of foraging sites both at a micro-habitat scale (Bao et al., 2006), and on a landscape scale (Powell, 1987; Johst, 2001; Gawlik, 2002).

The “prey availability hypothesis” states that prey availability limits wading bird population size (Halfner, 1997; Gawlik, 2002). Prey availability depends on both the density of prey, and the accessibility of prey (Kahl, 1964; Gawlik, 2002). Storks are tactile-feeders (Kahl, 1964) and are extremely dependent on high prey densities for foraging success (Kahl, 1964; Gawlik, 2002). This leads storks to abandon feeding sites more readily than other wading birds when prey densities begin to decrease (Gawlik, 2002). They exhibit what Gawlick (2002) labeled a “high-grading” foraging strategy, in which they select the highest quality patches and abandon them quickly as prey density decreases (Gonzalez, 1997; Gawlick, 2002). Like other storks, *Ciconia stormi* is known to exploit thermals and utilize non-flapping flight when searching for foraging habitat, (Birdlife International, 2001) and thus have a relatively low flight cost which makes the “high-grading” foraging strategy energetically beneficial.

Water depth also impacts the foraging success of wading birds (Powell, 1987). Wading birds are likely to have increased foraging success in shallower waters because of increased strike efficiency (less drag from the bill’s contact with water), greater prey detection rate (prey are less able to conceal themselves in the water column) and limited escape options of prey (an inability escape vertically within the water-column)

(Gawlik, 2002). Additionally, a bird's ability to access a water body is restricted by morphological features such as leg and bill length (Powell, 1987).

In Borneo, the varied micro-topography and intense precipitation events lead to a patchwork of flooded riparian networks and isolated pools. These shallow floodplains, small creeks, oxbow lakes and small pools hold high-densities of fish and aquatic invertebrates. At the same time, deep fast-flowing rivers and flooded waterways are likely to be inhospitable to *Ciconia stormi*, as the prey availability is reduced and *Ciconia stormi's* ability to access deep waters is limited by their leg length (Powell, 1987). This results in only a relatively small percentage of the landscape being suitable to forging at any one time. Therefore, *Ciconia stormi* likely requires a large spatial area to search for suitable foraging habitat, as only a small fraction of the landscape has accessible high-quality sites at any given time.

3.4.2 Site Specific Habitat

Areas of potential *Ciconia stormi* presence captured a wide range of canopy cover, including substantial areas with less than 5 percent tree cover. The dataset includes several sightings marked with a GPS, of *Ciconia stormi* flying over areas of low canopy cover. This demonstrates that fragmented forests do not limit *Ciconia stormi's* movements, although there is little evidence that these non-forested areas provide any benefits to the species. At this time, no data is available regarding the distances *Ciconia stormi* will to travel in search of suitable habitat, and this should be a focus of future research.

Interestingly, the lowest amounts of canopy cover from the high resolution data (LandNN4 and hemispherical photographs) was substantially less at locations where the bird was seen foraging or roosting, with lowest amount of canopy closure found at those locations to be 74% (table 3.4). Understory birds are somewhat confined to the general light conditions that dominate their habitat on the forest floor. However, some species are known to actively choose favorable microhabitats within their forest strata, either actively avoiding or seeking out patches of filtered and direct sunlight (Bell, 1982; Walther, 2002). The low variation in canopy cover found at locations were *Ciconia stormi* was seen foraging, relative to the variation of the surrounding area may indicate that the species actively selects areas of the forest where little direct sunlight penetrates (figure 3.7).

Variation in light intensity may influence the foraging success of *Ciconia stormi*, as has been shown with other gleaning species (those that take stationary prey from a substrate while perched) (Bell, 1982). *Ciconia stormi* is a tactile feeder, therefore forest light habitat most likely influences *Ciconia stormi's* foraging success by affecting prey availability. Large canopy gaps can result in a drier climate on the forest floor than in areas under denser canopy cover. This is likely to lead to a reduction in the availability of amphibians and riparian invertebrates that inhabit wet substrates and on which *Ciconia stormi* is known to feed (Birdlife International, 2001).

A second reason for the importance of dense canopy cover may be the ability to attract mates. Understory birds exist in habitats with low light flux, and light habitats of different spectral qualities than canopy birds (Walther, 2002). Specifically, forest shade is associated with yellow-green light due to vegetation surfaces reflecting most incident light (Endler, 1993). Small gaps in the canopy are associated with reddish light and small yellow-orange patches where direct light from the sun penetrates to the forest floor (Endler, 1993). Because of these properties, bird coloration patterns that have evolved to maximize conspicuousness in the understory are suggested to be orange-red (Endler, 1992). Furthermore, colors such as yellow and orange are very conspicuous in the understory because of the high contrast to background colors (Gomez and Thery, 2007). *Ciconia stormi* is characterized by a large reddish-orange bill and a large yellow eye-patch (Silvius and Verheugt, 1989). For a species presumed to exist in extremely low densities (Silvius and Verheugt, 1989), the ability to detect and attract mates is essential from a reproductive standpoint. A reduction of canopy cover may hinder the species ability to attract mates, which most likely depends on the exploitation of longer red and orange wavelengths of light that penetrate small gaps in the canopy (Endler, 1992; Gomez and Thery, 2007).

Finally, microclimatic conditions may restrict *Ciconia stormi* to areas of high canopy cover. Forest canopy cover is a principal determinant of microclimatic conditions within forest stands (Weiss et al. 1991). Foliage intercepts solar radiation and wind to create an environment that is cooler, calmer, and more humid than that outside groves (Geiger, 1965). Tropical birds that utilize forest mid-story and canopy habitats tend to shift their activity vertically downwards as diurnal sunlight increases, presumably in response to the increased temperature (Bell, 1982; Walther, 2002). *Ciconia stormi* utilizes understory habitats, and thus is incapable of relocating downward to cooler parts of the forest as diurnal temperature increases, therefore closed canopy conditions may be necessary for maintaining body temperatures.

3.4.3 Areas of Low Canopy Cover

Despite the overall high percentages of tree cover found in regions inhabited by *Ciconia stormi*, there are conspicuous extents of low-canopy area within this territory. Large gaps in the canopy have been shown to be correlated to Black stork nest locations (*Ciconia nigra*), presumably because they facilitate access to the nest (Jiguet and Villarubias, 2004). However, the patterns identified in the coarser resolution MODNN3 product are more akin to large areas of deforestation rather than what would be associated with natural gaps in the forest or even those created by selective logging. Higher resolution data from Landsat imagery and hemispherical photographs showed no sightings of the species roosting or foraging under large canopy gaps, although there was a small number of sightings with sufficient geographical precision to be included in that analysis.

These areas of low canopy cover most likely signify that the species may be somewhat tolerant of deforested areas within its wider territory. However, the percentage of area having low canopy cover constitutes a minority of the area likely to be occupied by *Ciconia stormi*, and the existence of these areas probably do not represent ideal habitat given the arguments for the importance of dense canopy cover.

3.4.4 Assumptions and Limitations

There are several limitations of this study that may lead to low canopy cover in areas of potential *Ciconia stormi* presence, and/or play a part in the poor relationship between confidence in stork presence and percent canopy cover. First there were temporal lags between the dates of remote sensing data acquisition and many of the sightings in the dataset. There was a temporal lag of 5 years between the earliest sightings of *Ciconia stormi* (1999) and the date of data acquisition used to create the MODIS forest cover product (2004). The temporal lag was as high as seven years between the date of the Landsat image (2000) and the most recent sightings of *Ciconia stormi* (2007). Individuals may have occupied forested areas at the time of sighting, but subsequent deforestation could result in low percent canopy cover in this analysis. Unfortunately, the small sample size precluded eliminating sighting records that stretched the temporal reliability of the data.

Additionally, the spatial resolution of MODIS or Landsat images may not be ideal for use with hemispherical photographs. Local variation of canopy conditions within 500 meter MODIS pixels appears to influence the utility of hemispherical photographs for validating MODIS derived products. Conversely, the imprecision inherent in the GPS locations where hemispherical photographs were taken (differential correction was not available in the study area) may cause errors when these photographs are correlated to individual Landsat cells. These issues of scale complicated the use of hemispherical photographs with both the MODIS and Landsat products. More work should be done to evaluate the ideal spatial resolution at which to apply hemispherical photographs.

Another potential source of error stems from the MCE-derived areas of potential of *Ciconia stormi* presence. Sightings assigned the lowest confidence ranking had a large amount of imprecision associated with them, and were assigned a large area in which the sighting could have potentially occurred. The large extent of the area designated to these sightings most likely encompasses territory not utilized by *Ciconia stormi*. The circular isotropic nature of the areas created by the point-radius method may also compound this type of error, as it does not account for topographic or land cover determinants of *Ciconia stormi's* distribution. Future work should incorporate what is known about habitat preferences into the model to further refine the areas designating confidence of presence.

The species' affinity for water may also result in the inclusion of lakes and rivers that lead to lower percent tree cover in the soft classification. Masking out water pixels identified in an alternate MODIS product resulted in only slightly higher values of percent tree cover (1-3%). Still, several sightings in the Mului River delta area appear to be associated with a large riparian system which may have influenced percent tree cover results (figure 3.10).

Finally, observer bias in the dataset may have led to more sightings occurring near towns and other developed or disturbed areas, which in turn would lead to disproportionate amounts of low canopy cover area included in the analysis.

One limitation of using hemispherical photography and the resulting tree cover map to assess forest disturbance is that recently disturbed forests comprised of many fast-growing pioneer species may result in high values of canopy closure. This may over estimate the importance of undisturbed forest for the persistence of *Ciconia stormi* in this analysis, as areas of high tree cover associated with species presence may in fact have been heavily disturbed. Future work should explicitly include analysis of land use history (e.g. forest succession stages) in addition to canopy cover.

A second limitation in the analysis is the small number of sightings for which hemispherical photographs could be obtained at locations where a *Ciconia stormi* was seen on the ground (n = 10). While the least amount of canopy cover at those locations was 74 percent, this may be a limitation of the small sample size.

3.4.5 Conclusions and Recommendations

Forest canopy cover is one of the few aspects of forest structure that land managers can both quantify and manipulate (Weiss et al. 1991). From a management perspective, understanding how much tree cover can be removed while continuing to ensure the existence of viable habitat for species is an important part of natural resource and wildlife population management (Pasher, 2007). The physiological and behavioral adaptations of understory bird species to understory light environments makes them particularly vulnerable to forest disturbance (Thiollay, 1992).

Ciconia stormi may utilize a “high-grading” foraging strategy. Because of the climatic patterns and topographic features of Borneo, only a small percentage of the landscape is high-quality habitat at any given time. This suggests that *Ciconia stormi* requires large areas from which to select locations of high-quality habitat. Because deforestation may decrease the biodiversity and abundance of aquatic organisms found in streams, these landscapes should be predominantly forested, specifically having over

75 percent canopy coverage. Additionally, *Ciconia stormi* displays morphological features that are consistent with species that have evolved to attract mates in low-light conditions.

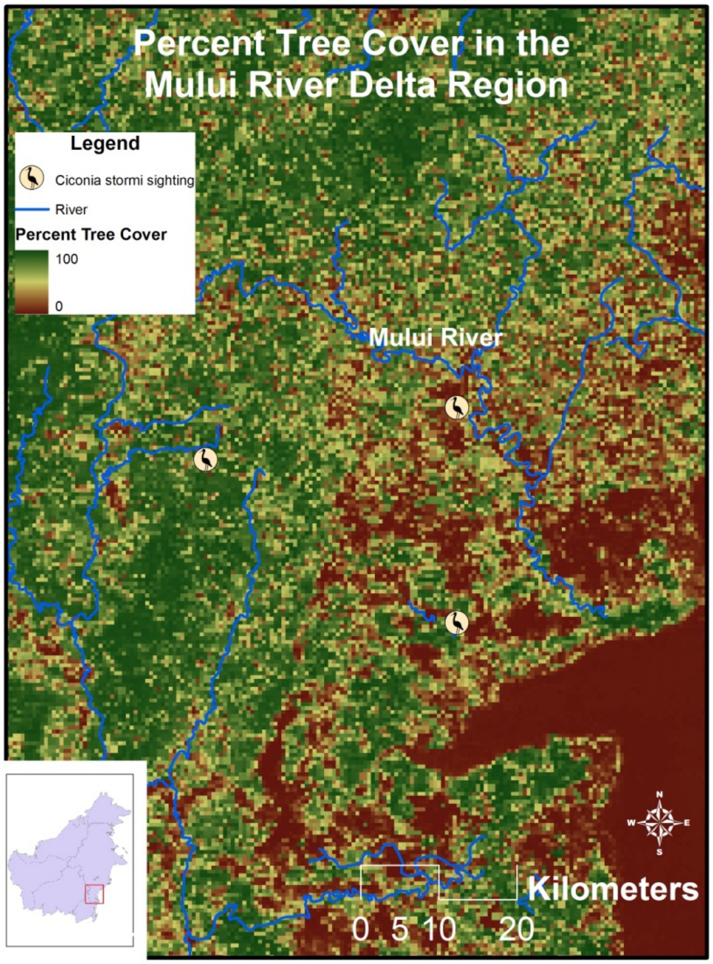

Figure 3.10. Some *Ciconia stormi* sightings occurred in riparian systems and are associated with low levels of tree cover (tree cover is derived from the MODNN3 product).

Chapter IV

Assessing Changes in *Ciconia stormi* Habitat Between 1993 and 2004

4.0 Background

Because of high biodiversity and endemicity in Borneo, protected areas alone cannot safeguard all of the biodiversity found in Borneo's tropical forests (Jepson et al., 2002). Therefore, conservation of many tropical species may depend on the ability of disturbed forests to provide suitable habitat (Frumhoff, 1995; Fimbel et al., 2001). While production forests cannot substitute for protected areas, many species can persist within silviculture projects that are carefully managed for ecological sustainability (Frumhoff, 1995). However, for logged forests to have high conservation value, the rate of forest recovery must surpass the rate of disturbance events, namely subsequent logging activities or fire events that are prevalent in degraded forests (Chapman and Chapman, 2003). Meijaard and Sheil (2007) illuminate the necessity for a collaborative relationship between logging interests and conservation efforts: "sustaining biological diversity in production forests requires environmentally sound management practices that incorporate available scientific knowledge about species vulnerability into interventions".

Many studies have investigated bird community composition in tropical forests after logging, nearly all of which conclude species that utilize understory habitats are the most vulnerable to logging activities (Johns, 1996; Sekercioglu et al., 2002; Thiollay, 1992; Yap et al., 2007). To supplement this work, more information is needed regarding specific levels of disturbance individual species are able to tolerate so that land managers are able to ensure sufficient wildlife habitat is sustained (Frumhoff, 1995; Meijaard and Sheil, 2007).

4.0.1 Forest Fragmentation

While net changes in forest cover are important to ecological processes (e.g. carbon budgets; micro-climate), forest structure also influences these processes (Turner, 1989). Forest fragmentation in particular is a principal threat to biodiversity (Harris, 1984; Pichancourt et al. 2006). To minimize the negative impacts of forest fragmentation on biodiversity, land managers need to quantify landscape fragmentation. Landscape ecology has emerged as one of the dominant paradigms in ecology beginning in the 1980s, bringing spatial analysis to the forefront of ecological research (Li and Wu, 2004). Landscape ecology focuses on the mutualistic interactions between spatial pattern and ecological processes, scale and hierarchy (Turner, 1989; Brandt, 1998; Wu and Hobbs, 2002).

Some of the most powerful tools developed by landscape ecologists have been the array of landscape metrics used to quantify and describe changes in land-cover. Despite the widespread adoption and application of these various landscape indices, there is the potential for misapplication. First, landscape indices may have variable or unpredictable responses to changes in spatial patterns, or be difficult to interpret (Li and Wu, 2004). Additionally, many indices are highly correlated and thus redundant when used in conjunction (Ritters et al., 1995; Hargis et al., 1998). This stems from the fact that the metrics used are based on the same limited number of basic measurements: patch size, shape, perimeter-to-area ratio, and inter-patch distance (Li et al. 1993). It has also been shown that many landscape metrics are sensitive to methodological choices such as spatial resolution, categorical precision and map accuracy (Li and Wu, 2004; Neel et al., 2004; Langford et al., 2006). Furthermore, certain landscape indices behave non-linearly and so are not appropriate for correlative work (Li et al., 2005; Li and Wu, 2004). Finally, caution must be used to separate form from function, as some ecological processes, such as fire, both affect and are affected by spatial patterns of landscapes (Turner, 1989; Pickett and Cadenasso, 1995). Regardless of these challenges, landscape metrics can be a powerful measure of land cover change, and have been widely utilized in wildlife and ecological studies (Clayton et al. 2003; De Orio et al. 2005; Clerici et al. 2007; Cakir, 2008).

Because of the potential difficulties associated with using landscape indices, selecting the metrics most applicable to the study question can be challenging. Ritters et al. (1995) demonstrated that because many indices are highly correlated, most indices can be grouped into six basic categories according to the type of information they represent. These categories are average patch compaction, overall image texture, average patch shape, patch perimeter-area scaling, number of attribute classes and large-patch density-area scaling (although this last family was associated with only a single outlier in their data and may not be as strongly correlated to other metrics as the five preceding groups). Subsequent work by Hargis et al. (1998) showed that measures of inter-patch distance such as mean nearest neighbor distance and mean proximity distance, had low correlations with other metrics and one of these measures should be added to the set of landscape metrics identified by Ritters and colleagues (1995). Ritters and colleagues (1995) went on to suggest a suite of representative metrics that correspond to each of the families of pattern and structure. However, later discussion has illuminated the importance of selecting metrics that are pertinent to the study question, rather than adopting a standard list of metrics (McGarigal et al. 2002).

4.1 Chapter Overview

This chapter explores the forest canopy conditions at different times after logging activities of varying intensities in East Kalimantan, Indonesia. Those canopy conditions are then related to the forest canopy requirements of *Ciconia stormi* identified in the previous chapter.

Levels of forest fragmentation in areas potentially occupied by *Ciconia stormi* are also examined with a two-fold objective in mind. Firstly, this portion of the research addresses the state of *Ciconia stormi* habitat by quantifying the changes in forest pattern that have occurred in areas potentially occupied by the species. Secondly, this analysis may provide some insight into the level of habitat fragmentation the species is able to tolerate. This assumes that if the species is present in an area following forest loss and fragmentation, the species may tolerate some level of disturbance. However, this assumption does not account for all delayed responses; i.e. the species may persist in the area but not be able to breed, thereby facing local or global extinction.

4.2 Methods

4.2.1 Percent Canopy Cover

Hemispherical photographs were collected at forest sites that experienced various levels of disturbance at different points in time as described in chapter three (see table 3.1). These areas were then grouped into four strata: primary forest, selectively logged forest, older heavily logged forest [greater than ten years before present (ybp)] and recently heavily logged forests (less than five ybp). Anecdotal information on land use history was obtained from researchers at The Nature Conservancy East Kalimantan program who work closely with logging concessionaires and are familiar with the study area. A total of 200 hemispherical photographs were then processed using GLA as described in chapter three in order to relate the canopy conditions found within each of the forest disturbance strata to the 75 percent regional canopy requirements and the over 85 percent site-specific canopy requirements identified in chapter three.

4.2.2 Forest Loss and Forest Fragmentation

Land Cover Maps

Land cover data was obtained for two years, 1993 and 2004. The 1993 product was downloaded from the Global Land Cover Characterization (GLCC) website (http://edcsns17.cr.usgs.gov/glcc/). This dataset was created from 1-km AVHRR (Advanced Very High Resolution Radiometer) 10-day NDVI (Normalized Difference Vegetation Index) composites using an unsupervised classification, specifically the CLUSTER algorithm (Kelly and White, 1993). The AVHRR source imagery derived from April 1992 through March 1993 and was supplemented by additional data sources including digital elevation data, ecoregions interpretation, and country or regional-level vegetation and land cover maps (Loveland et al. 1999). A series of independent investigators found the map's accuracy to be 73.5 percent overall or 78.7 percent for an area weighted accuracy assessment (Scepan, 1999). While this level of accuracy is less than ideal, it is within the 60-80 percent accuracy recommended for inventorying resources for management (Green et al., 2000). The IGBP land cover classification (Belward, 1996) was selected from a variety of available classification schemes. This map was then reclassified into water, forest, other vegetation, and bare/urban.

The second land cover map from 2004 was provided by Dr. Doug Fuller of the University of Miami. This product was created from a series of Moderate Resolution Imaging Spectrometer (MODIS) images captured between June and August 2004. The 500 meter resolution images were classified using a neural network classifier into five land cover types: dense forest, other vegetation, bare/urban, water, and cloud (Fuller and Murphy, 2005). The authors assessed the map's accuracy using 15 SPOT-5 false color composites. An error matrix revealed an 83 percent overall accuracy, which increased to 91 percent if only validation polygons larger than 5,000 ha were used.

Both of the maps were then imported into Idrisi (Eastman, 2006) and projected into the UTM 50 North geographic reference system. The 1993 AVHRR product was assigned a 500m resolution. While this increased resolution may compound certain classification errors, and will affect some fragmentation metrics, matching the image resolutions is necessary for comparison and change detection. Following Nagendra et al. (2006), an overlay function and careful visual observation was used to ensure that the images overlapped exactly across both image dates. Additional geometric correction was not performed on the images beyond that performed by the image providers. Because the 2004 map was originally 500 meter resolution, it provided more detail in the coastline areas. Additionally, the 2004 map contained more pixels classified as water on the island's interior, again due to resolution differences and potentially because of differences in the algorithms used to classify the maps. Therefore it was necessary to exclude water pixels from the analysis, as theoretically, changes of other land cover types to or from water would be a function of resolution and classification differences between the two maps, rather than actual land cover changes. In order to exclude water pixels, the land change modeler was used to map transitions of various land cover types to or from water (e.g. forest to and from water, other vegetation to and from water, bare/urban to and from water). These areas were then overlaid onto the land cover maps from 1993 and 2004, assigning water to any pixel that was ever classified as water in either of the two images and eliminating the possibility of any transformations between alternate land-cover types and water. This resulted in the final land cover maps used in the land cover change analysis (figure 4.2a and 4.2b).

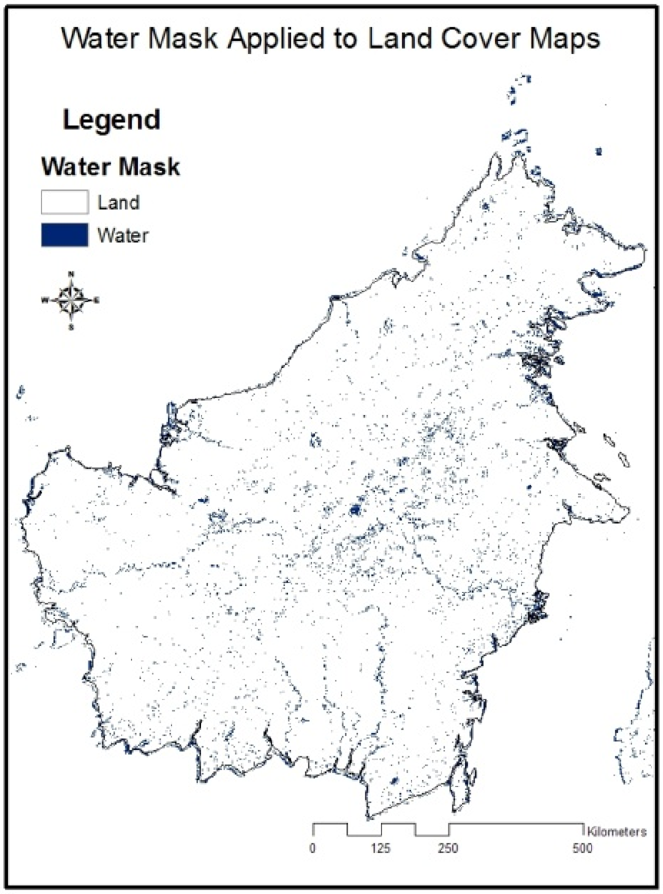

Figure 4.1. Map identifying pixels classified as water in either the 1993 AVHRR or 2004 MODIS land cover maps.

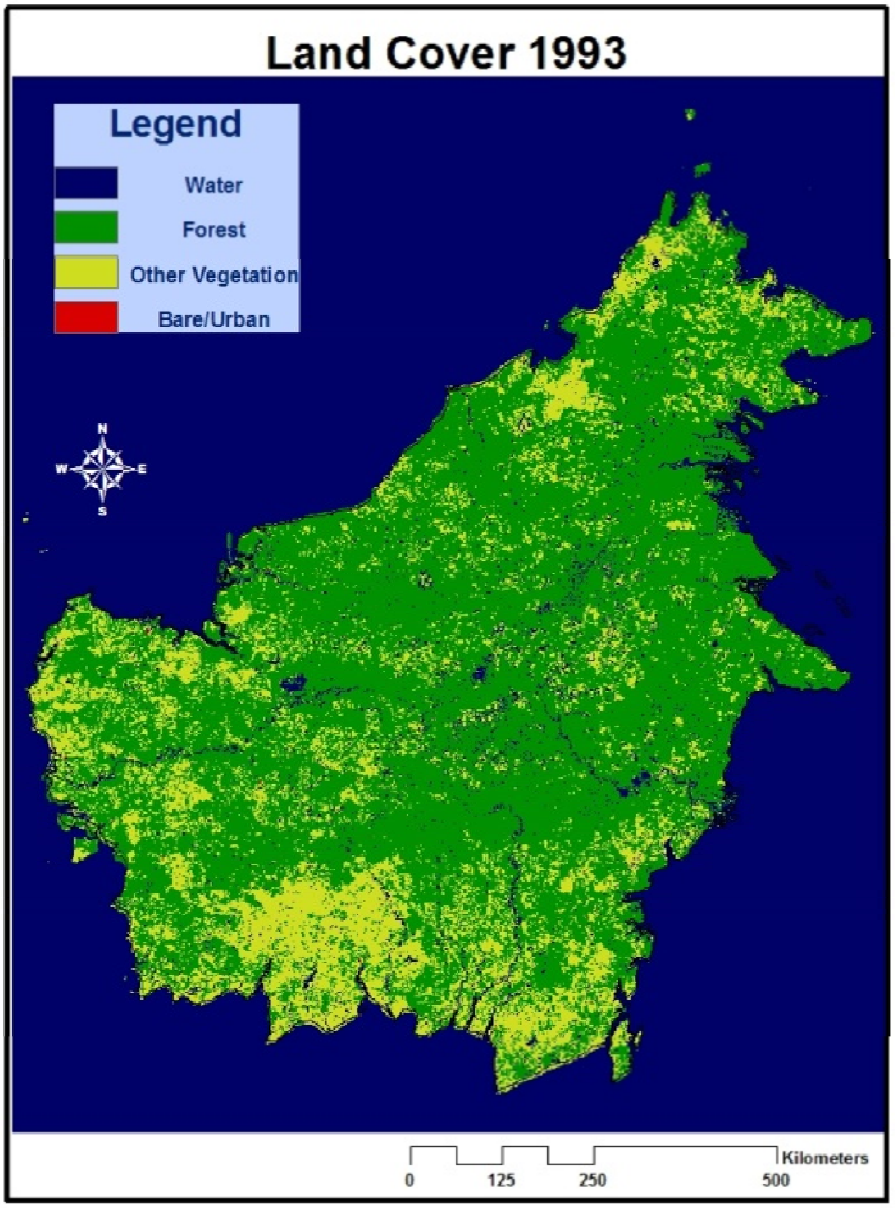

Figure 4.2a. Land cover map of Borneo derived from 1993 AVHRR data.

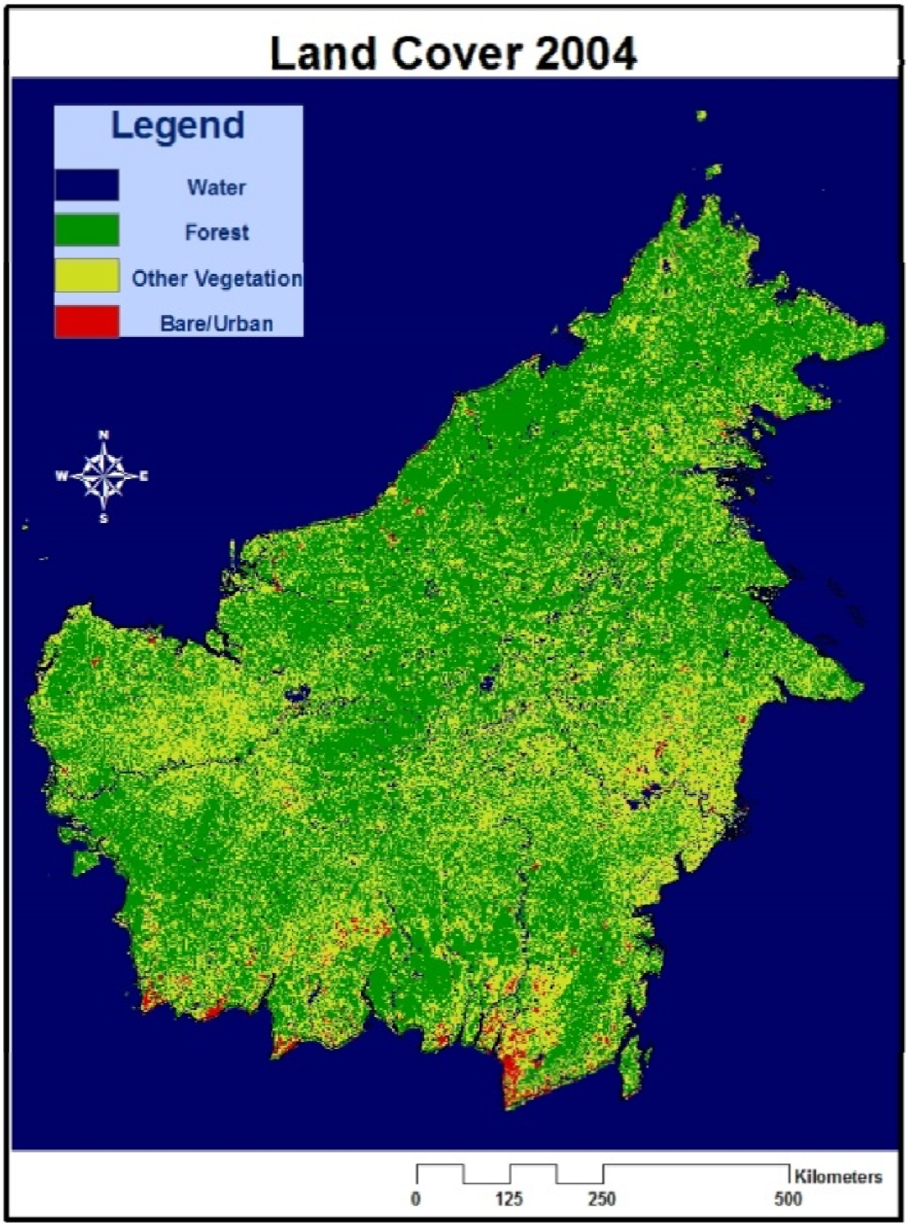

Figure 4.2b. Land cover map of Borneo derived from 2004 MODIS data.

4.2.3 Forest Cover Change Analysis

The land change modeler in IDRISI (Eastman, 2006) was used to map changes in forest cover that occurred between 1993 and 2004 within areas of potential *Ciconia stormi* presence from 1999-2007. This reveals the changes in forest cover that have occurred in locations where *Ciconia stormi* was recently seen, and may provide some indication of the level of forest disturbance the species is able to tolerate. The land cover change modeler outputs a classified change map where each pixel contains information on the change in forest that occurred between the two dates: forest loss, persistent forest, and forest gain.

4.2.4 Fragmentation and Landscape Dynamics

Forest fragmentation was assessed within areas of potential *Ciconia stormi* presence at 20%, 40%, 60% 80% and 90% confidence levels. There are a suite of spatial analysis packages that calculate landscape metrics which correspond to the groups of landscape change identified by Ritters et al. (1995) including FRAGSTATS, Patch Analyst and the r.le Programs (Baker and Cai 1992; Elkie et al. 1999; McGarigal et al. 2002). Patch Analyst was selected because it easily integrates into ESRI ArcView software and required no additional data transformations (Elkie et al. 1999). Both FRAGSTATS and Patch Analyst have been widely applied in similar studies of land cover change (Clayton et al. 2003; De Orio et al. 2005; Clerici et al. 2007; Cakir, 2008) and their continued use will allow for comparative work in the future. At least one metric from each of Ritters et al. (1995) separate groupings was selected from those available, with the exception of number of attribute classes group and nearest neighbor statistics (see table 4.1). The number of attribute classes group has been expanded from the single metric originally identified by Ritters et al. (1995) to include a variety of diversity measures (McGarigal et al. 2002). However, Patch Analyst only calculates these measurements at the landscape level, not the class or patch level. In this study, the landscape is of a derived nature and may be partially a facet of observer bias on the part of data collectors and the methodology involved in constructing the confidence of stork presence. Therefore, metrics that are applicable only at the landscape level were excluded. Nearest neighbor statistics measure the average of the distances between patches of like classes and were also excluded from this analysis. In situations where an area of potential presence contains only one patch of a particular forest class, the nearest neighbor measurement will extend to the next area of potential presence, a somewhat artificial measurement because the distribution of areas containing stork presence data may be affected by observer bias.

Table 4.1. Groupings of landscape metrics as described by Ritters et al. (1995) and Hargis et al. (1998), and the specific metrics used to represent those groups in this study.

Groupings of information described by landscape metrics	Metrics used in this study
Average patch compaction	Class area and Edge Density
Overall image texture	Mean Patch Size, Contagion
Average patch shape	Mean Shape Index
Patch perimeter-area scaling	Core metrics
Number of attribute classes	NA
Inter-patch distance	NA

The maps depicting forest cover in areas of potential *Ciconia stormi* presence were imported into ArcView. No smoothing filter was applied to the images because smoothing can increase errors in landscape pattern indices by overestimating total edge and mean patch size (Brown et al., 2000; Langford et al., 2006). Following the groupings identified by Ritters et al. (1995) and available in Patch analyst (Elkie et al. 1999), the following metrics were calculated for forest in the areas of potential *Ciconia stormi* presence:

<u>Class Area:</u> Class area is the amount of each land-cover type represented in the landscape. This metric is reported as the percentage of the total area covered by forest.

<u>Edge Density:</u> Edge density reports the amount of edge standardized per unit of area, in this case ha (McGarigal et al. 2002). It should be noted that this measurement is influenced by the resolution of the image. At finer resolution, edges contain more detail and record as higher values compared to those obtained from straight-line edges of coarse resolution imagery. In this study edge density is compared between two images of like resolution, so this limitation should not affect the results. Edge effects directly influence wildlife distributions as well as processes such as predator-prey dynamics and foraging strategies (Skoczylas et al. 2007; Christianini and Galetti, 2007).

<u>Mean Patch Size (MPS):</u> MPS is the average area of all patches in a particular class. This metric provides information at both the landscape and class level. A landscape with a smaller mean patch size than another landscape can be said to be more fragmented and a class with a smaller mean patch size than another class can also be considered more

highly fragmented (McGarigal et al., 2002). This measure has been shown to influence wildlife distributions in other studies (Liu et al. 2003; Matter, 2006; Huste and Boulinier, 2007). It should be noted that this measure is constrained by spatial resolution of a raster image, as no patch can have an area smaller than the area of a single pixel (25 ha in this instance).

Contagion: Contagion measures the level of aggregation of landscape elements (McGarigal et al., 2002). A landscape with large, contiguous patches will result in a higher contagion value than a landscape with many small, dispersed patches. Contagion is calculated as a probability (see McGarigal et al., 2002 for description) and reported as a percentile. This metric reveals something about the make-up of the landscape at the patch-level but it is influenced by the distribution of patches at a landscape level so caution should be taken when interpreting its results in this analysis.

Mean Shape Index (MSI): MSI is a measurement of the average perimeter-to-area ratio, or average patch shape within each class type. MSI equals patch perimeter (given in number of cell surfaces) divided by the minimum perimeter (given in number of cell surfaces) possible for a maximally compact patch of the corresponding patch area. MSI equals 1 when the patch is maximally compact (square or almost square), and increased without limit as shapes become less compact (McGarigal et al. 2002). Patch shape has been shown to influence wildlife foraging strategies (Forman and Godron, 1986).

Core Area Density and Mean Core Area: Core area density is the number of core areas per 100 ha. Mean core area is reported as the mean size of all core areas as compared to the mean core area per patch. Patches with zero core area are excluded from the calculation of this metric. *Ciconia stormi* is known to be sensitive to noise pollution (Danielsen et al. 1997). Because noise penetrates approximately 400m into forests (Pocock and Lawrence 2005), core areas were defined as areas at least 500m from the patch edge.

4.3 Results

4.3.1 Percent Canopy Cover from Hemispherical Photographs

Analysis of hemispherical photographs showed that the average percent tree cover in areas of primary forest, older selectively logged forest, older heavily logged forest, and recently heavily logged forests were 93.9 percent, 87.7 percent, 82.6 percent, 72.7percent respectively (figure 4.3). The average percent tree cover in each disturbance strata exceeded the 75 percent threshold identified as the amount of forest cover required by *Ciconia stormi,* except areas that were heavily logged within the past five years. Undisturbed primary forest and older selectively logged forest averaged greater than 86 percent canopy cover, the amount associated with sightings of *Ciconia stormi* feeding or roosting.

Perhaps the greatest difference between areas that experienced lower intensities of logging and those that experienced higher intensities, was the range of canopy conditions within those sites. Specifically, the standard deviation within primary forest and selectively logged forests was 2.3 and 10.3 percent respectively. In areas that underwent heavier commercial logging, the standard deviations were 21.6 percent for older logged forests and 29.4 percent in recently logged forests. This is in part due to the more elaborate road networks (both logging roads and skid trails) found in heavily logged areas, which result in an absence of tree cover over those roads (Cannon et al., 1994).

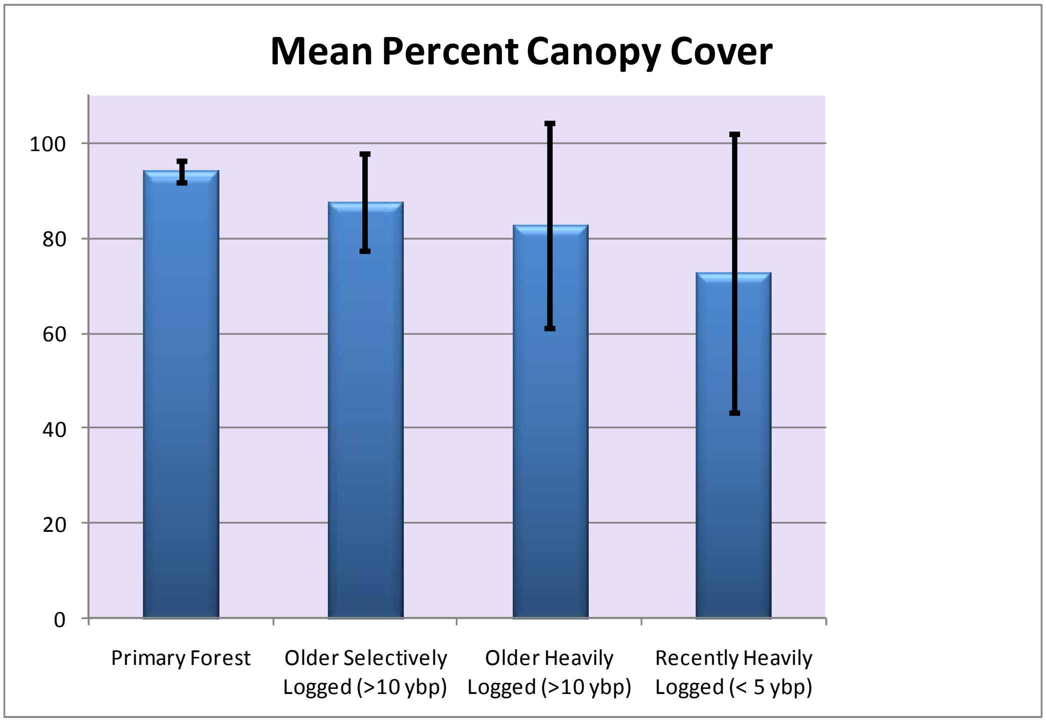

Figure 4.3. Mean percent canopy cover in forests following various logging intensities. Standard deviations were substantially higher following heavy commercial logging activities.

4.3.2 Land Cover Change and Forest Fragmentation

4.3.2a Method of reporting

There are two different ways to report landscape metrics for this study: analysis of *areas* where *Ciconia stormi* was potentially present, or analysis of the *forests* used by *Ciconia stormi*. The first method uses areas of potential *Ciconia stormi* presence as the areal unit, and analyzes the forest cover changes within those circular areas. Analysis of this nature needs to consider the Modifiable Unit Area Problem (MAUP), which is the problem of differing results due to different sizes or locations of constructed sampling units. As addressed in chapter two, the areas of potential *Ciconia stormi* presence are derived from user defined parameters and MAUP is a potential limitation to this research that will be addressed further in chapter five.

The alternative method is to use the entire forest patch as the areal unit, as opposed to only the subset that falls within the circular areas of potential presence. This method assumes the species may use the entire contiguous forest patch, and is not restricted to the portion of forest that falls within areas of potential presence. This introduces a potential source of error by including in the analysis sections of forest that the species does not actually use, perhaps due to inaccessibility (e.g. portions may exceed the upper end of the species altitudinal range) or the availability of more preferred habitat nearby.

Scaled indices (MSI, contagion) report values that are useful for comparative purposes, but are difficult to translate into management recommendations because they are not “real-world values” (e.g., mean shape index begins at 0 and increases without limit). Edge density, mean shape index and contagion are reported as values within areas of potential *Ciconia stormi* presence. Mean forest patch size and mean core area may provide important information to land managers, so those metrics are reported both using both areas of potential presence and forest patches used by Ciconia stormi as the areal unit of measure.

4.3.3 Land Cover Change Results

There was substantial forest loss and fragmentation in areas of potential *Ciconia stormi* presence between 1993 and 2004 (table 4.2). Forest Cover decreased from 76.9 percent to 61.9 percent (table 4.2). Of those forested areas, 78.1 percent was persistent forest (forest that did not change between 1993 and 2004), while reforested areas accounted for 21.9 percent of the forested landscape (figure 4.4). It is difficult to know what proportion of reforestation is forest regeneration following disturbance, and what proportion is plantation forest. This is an important distinction, as plantation forests are less likely to provide the same quality of habitat as naturally regenerating forests reasons that will be discussed shortly. Regardless of the state of regenerating forests, reforested areas constitute a minority (21.9 percent) of the forests in areas of potential *Ciconia stormi* presence.

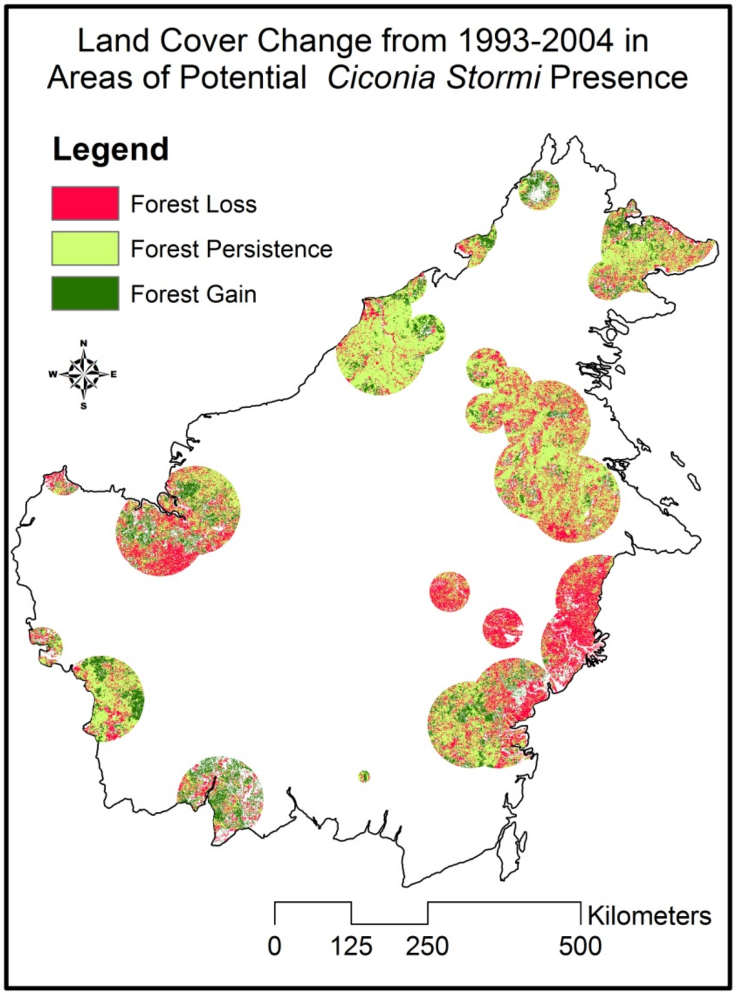

Figure 4.4. Forest change in areas of potential *Ciconia stormi* presence. Forest change occured between 1993 and 2004. Areas of *Ciconia stormi* presence identified from sightings that occurred from 1999 -2007.

All landscape indexes revealed increased forest fragmentation both in areas of potential *Ciconia stormi* presence, and in forest patches potentially used by *Ciconia stormi* (tables 4.3 and 4.4). While the values of landscape metrics varied at different confidence levels of presence, the trend of forest loss and fragmentation was present at all confidence levels (tables 4.5 and 4.6). The mean forest patch size in areas of potential *Ciconia stormi* presence was 1,332.6 ha, while the mean patch size of forests used by *Ciconia stormi* was 1,121.8 ha. Each of these figures is larger than the average forest patch size on Borneo of 452.0 ha. The average core area in areas of potential *Ciconia stormi* presence decreased between 1993 and 2004 from 7,601 ha to 802 ha. Forests potentially used by *Ciconia stormi* have average core areas of 7229.6 ha (figure 4.3). This number is larger than the average forest patch size, because many small forest patches do not contain core areas and are excluded from the analysis. Again, the average core in forests potentially used by *Ciconia stormi* is larger than the average core area of forests on Borneo of 2,143.8 ha (figure 4.3).

Table 4. 2. Areas of potential *Ciconia stormi* presence have undergone substantial forest loss between 1993 and 2004.

Landscape Metric	1993	2004
Percent of Landscape Forested	76.9 %	61.9 %
Mean Forest Patch Size	13,699 ha	1,332.6 ha
Mean Core Area	7,601.2 ha	801.9 ha
Edge Density (m/ha)	4.4	11
Mean Shape Index	1.46	1.34
Contagion	63.9 %	41.2 %

Table 4.3 Persistent lowland forest patches potentially used by *Ciconia stormi* are larger and have larger core areas than average forest patches on Borneo.

Landscape Metric	Forest Potentially Utilized by *Ciconia stormi*	Standard Deviation	Mean Value for Lowland Forests on Borneo	Standard Deviation
Mean patch Size (ha)	1,121.8	95,706.9	452.0	49,649.0
Mean Core Area (ha)	7,229.6	191,881.4	2,143.8	94,226.3

Patches of persistent forest where the species was seen foraging or roosting averaged 1,402.8 ha. However it is difficult to draw conclusions regarding minimum forest patch size required by the species from this figure, because of the small number of sightings recorded with precision (marked with a GPS unit) between 1999 and 2007, and the fact that many of these sightings fell within a single large forest patch covering 12,124,935 ha. The patch of persistent forest identified spanned from Central Kalimantan to Sabah. The size of this block of contiguous forest seems somewhat dubious considering the large amount of deforestation that has occurred on Borneo (Achard, 2002; Curran et al., 2004). Even heavily logged forests may maintain relatively high levels of tree cover (figure 4.3); a phenomenon which may lead to misclassifying highly disturbed areas as persistent forest.

Table 4.4. Values of landscape metrics vary depending on the level of confidence of *Ciconia stormi* presence.

	Mean Patch Size (ha)		Percent of Landscape Forested		Mean Core Area (ha)		Edge Density - m/ha		Mean Shape Index		Contagion	
Confidence Level	**1993**	**2004**	**1993**	**2004**	**1993**	**2004**	**1993**	**2004**	**1993**	**2004**	**1993**	**2004**
Any	13699.0	1332.6	76.9	61.9	7601.2	801.9	4.4	11	1.46	1.34	63.9	41.2
20	7615.5	1276.7	78.2	63.1	5093.3	571.0	7.0	18.3	1.6	1.53	64.9	43
40	5183.4	1130.9	80	59.8	3509.5	504.9	7.7	19.3	1.6	1.55	62.7	42.5
60	4785.6	1069.1	79.3	60.3	4613.0	454.2	6.8	19.6	1.36	1.60	58.8	42.7
80	3327.5	787.0	80.1	60.6	3184.2	334.2	7.9	22.8	1.32	1.56	59.7	43.8
90	1895.5	700.8	79.3	60.1	1900.5	281.5	7.4	14.4	1.41	1.49	58.1	42.6

Table 4.5. Forest patch size and core areas vary depending on which forest patches are included in the analysis.

Confidence Level	Number of Forest Patches	Mean Patch Size of Persistent Forest (ha)	Standard Deviation (ha)	Number of Core Areas	Mean Core area of Persistent Forests (ha)	Standard Deviation (ha)
Any	16509	1,121.8	95,706.9	1233	7,229.6	191,881.4
20	458	32,512.4	573,834.9	52	7,602.8	31,635.2
40	229	62,097.3	810,769.0	25	2,217.7	5,770.0
60	98	144,229.9	1,238,215.1	11	3,044.9	8,215.2
80	71	201,871.2	1,464,082.6	7	5,415.9	11,663.6
90	49	294,339.7	1,766,100.6	6	6,707.3	13,048.6

4.4 Discussion

4.4.1 Percent Canopy Cover

Surprisingly, few studies have addressed canopy cover conditions in SE Asian tropical forests following logging, with Cannon et al. (1994) being a notable exception. Cannon and colleagues assessed forest structure at six months, one year, and eight years after logging in a West Kalimantan forest. Their methodology separated short secondary growth from tall secondary growth and they reported canopy cover in only heavily disturbed portions of logged forest, making it difficult to compare their results to this study, which assessed all levels of disturbance found within a single logging regime. However, they did find approximately 82 percent canopy closure at eight years after logging. These findings are in line with those reported in this study, where forests having experienced heavy logging at least ten years prior exhibited 82.6 percent mean canopy closure. Cannon et al. also reported a progressive closure of forest canopy over time, similar to the results of this research (figure 4.3).

Finally, Cannon and colleagues noted a complex mosaic of canopy conditions within a single forest after logging. This is due to a combination of inaccessible areas within a timber concession that go unlogged, coupled with heavy extraction in other parts of the concession (Cannon et al., 1994). This thesis expands on their finding, by identifying a gradual homogenization of canopy cover over time (figure 4.3). Specifically, the standard deviation within logged forests decreased from 29.4 percent in recently heavily logged areas, to 21.6 percent in older heavily logged forests. The standard deviation further decreased in older selectively logged forests to 10.3 percent and was small at 2.3 percent in primary forests. While extraction of large trees clearly lowers mean canopy cover, the impact of road networks developed in logged forests should not be overlooked. Both logging roads and skid trails result in little to no canopy cover over portions of logged forests, which, combined with tree regeneration, results in a wide range of canopy conditions within a single logging regime. These findings indicate that commercially logged forests may provide sufficient average canopy cover for *Ciconia stormi* just ten years after logging. However, at ten years after logging large gaps in the canopy are still prevalent. *Ciconia stormi* may actively avoid canopy gaps when foraging (table 3.7), therefore while the average canopy cover ten years after logging may be sufficient, a large proportion of these forests is still unsuitable for the species. Forests selectively logged twenty years prior match levels of average canopy closure found in areas of potential *Ciconia stormi* presence, and have few areas of low canopy cover, indicating the bulk of these forests may be suitable habitat for *Ciconia stormi*.

4.4.2 Impacts of Logging on Avian Communities

Over time, some bird species are able to recover from selective logging of their habitat. Wong (1986), found that breeding occurrences of understory birds in SE Asian tropical rainforests 25 years after selective logging were not significantly different from breeding occurrences in virgin forests of similar structure (Wong, 1986). Other studies reveal that breeding occurrences, diets and food availability of understory birds were similar in forests 30 years after selective logging and unlogged forests (Yap et al., 2007).

A review by Lambert and Collar (2002) concluded that large areas of selectively logged forest will retain a majority of bird species that were present prior to logging, while forest fragments, even those of considerable size, will experience severe species loss (Lambert and Collar, 2002). Particular taxa (e.g. woodpeckers and hornbills) are more negatively affected by logging, as are species that utilize understory habitats (Lambert, 1992; Cleary et al., 2007). Logging seems to favor certain species that are able to exploit particular niches (Boulinier et al., 2001). At the same time, logging is most detrimental to species that rely on those processes compromised by logging activities (Lambert and Collar, 2002).

The work of Wong (1986) and Yap et al. (2007) found a similarity of bird communities in logged and unlogged forests. Their research was conducted 25 and 30 years after logging, as opposed to other studies that occurred more recently after logging (Wong, 1986; Lambert, 1992; Boulinier et al., 2001; Sekercioglu, 2002; Yap et al. 2007). This seems to indicate that levels of bird biodiversity may return close to pre-disturbance levels 25 years after logging. However, this may be dependent on the proximity of undisturbed habitat, which is important for the recolonization of disturbed forest patches (Wong, 1986; Yap et al., 2007). Additionally, the intensity and management of logging activities is likely to influence avian community recovery. Heavy logging may permanently change vegetation structure through climatic changes corresponding to increased canopy gaps and wind penetration (Geiger, 1965). Similar lasting alterations of habitat result from greater human use of an area brought on by increased access after development of road networks. These types of structural changes negatively affect particular bird species in tropical forests, particularly understory species (Sekercioglu, 2002).

Some understory species may be physiologically incapable of tolerating changes to light and humidity conditions that follow logging (Thiolay, 1992). Other species may be affected by decreases in available food resources (Frumhoff, 1995). *Ciconia stormi* may rely on closed canopy light conditions for breeding purposes as discussed in chapter three. Additionally, *Ciconia stormi* nests in large canopy trees overhanging rivers (Danielsen et al. 1997). Logging activities that target large old-growth trees may restrict nest site availability. Nest site selection is an important factor of habitat preference for the black stork a closely related forest-interior species (Lohmus and Sellis, 2003; Lohmus et al. 2005).

4.4.3 Impacts of Logging on Stream Biodiversity and Function

The impacts of logging on stream biodiversity and aquatic food resources are also likely to impact the benefits disturbed forests provide to *Ciconia stormi*. Erosion caused by logging activity increases levels of water turbidity (Clarke and Walsh, 2006; Harris et al., 2007). The gradual degradation of logging roads may lead to elevated levels of erosion and severe erosion events up to 15 years after logging (Clarke and Walsh, 2006).

Canopy cover also affects the amount of rain throughfall in South East Asian tropical forests, with large amounts of throughfall reaching the forest floor through canopy gaps (Konishi et al., 2006). The increased size and number of canopy gaps in logged forests likely leads to greater intensity of throughfall and increased erosion and soil influx into streams.

Increased sedimentation from logging-induced erosion decreases densities of certain fish taxa (Rodriguez and Lewis, 1997; Alin et al., 1999). This is caused by reduced light penetration, which, in turn, may decrease levels of benthic primary production (Alin et al., 1999). As discussed in chapter three, fish density is a key indicator of foraging success for storks (Kahl, 1964; Gawlik, 2002). Erosion from logging may lower fish densities, thereby reducing *Ciconia stormi's* ability to forage effectively.

Reduced canopy cover has also been linked to loss of species richness in aquatic invertebrate communities, a critical trophic level of riparian food-webs (Benstead et al., 2003). It is hypothesized that increased availability of basal food resources (streams with high canopy cover store greater amounts of biomass) is the primary driver behind high species diversity amongst invertebrate communities in heavily forested streams (Benstead et al., 2003). Other work has shown that deforestation, even at a watershed scale, may lead to reduced diversity among riparian fish assemblages (Harding, 1998). Because storks are tactile-feeders, they are unable to discriminate among prey once they are encountered; therefore *Ciconia stormi* is more likely to be affected by fish densities than fish community assemblages. At the same time, storks are known to be highly selective for particular fish species, indicating that some prey are more easily encountered (Gawlik, 2002). The loss of preferred prey is likely to challenge the species' ability to meet its caloric requirements.

4.4.4 Impacts of Fire on Avian Communities

Fires also contribute to the levels of forest loss and fragmentation on Borneo (Curran et al., 2004; Fuller et al., 2004). As with logging, fire decreases bird species diversity in Borneo (Slik and Van Balen, 2006). Slik and Van Balen (2006) found that the species most affected by fires were those that tend to prefer closed forests and persist in the understory, particularly predatory species. Sozer and Nijman's (2004) work on another endangered water-bird, the white shouldered-ibis (*Pseudibis davisoni*), revealed the species was absent in burned areas, despite it being relatively abundant in these same areas before the fire. Again, the loss of forest cover from fire can result in finer substrates and more eroded riverbanks (Iwata et al. 2003) potentially altering river fish and invertebrate communities, in turn reducing *Ciconia stormi*'s ability to forage effectively.

4.4.5 Plantation Forests

Conversion to plantation forest may have been classified as forest gain, persistent forest, or loss in this study depending on what stage in the conversion process the plantation plot was at the time of data acquisition. For this reason, it is difficult to address the prevalence or impacts of plantation forests within areas of potential *Ciconia stormi* presence. However, certain South East Asian bird species are known not to inhabit plantation forests (Sodhi et al., 2005; Peh et al. 2006), and many of the activities that lead to degraded riparian system function and structure (e.g.

construction of roads and loss of canopy cover) are likely to occur as part of plantation management.

4.4.5 Impacts of Forest Fragmentation

There has been a substantial amount of forest fragmentation in areas of potential *Ciconia stormi* presence (table 4.2). The fact that the species persists in these areas shows that it may tolerate some level of habitat fragmentation. It is interesting that despite net forest loss and increased habitat fragmentation, the mean lowland forest patch size in areas of potential *Ciconia stormi* presence is larger than the mean lowland forest patch size in Borneo (table 4.2). This may indicate that Storm's stork is selecting the largest forest patches of those available. Without absence data it is difficult to draw conclusions about how much forest loss and habitat fragmentation leads to loss of populations. Radio telemetry data of tagged individuals would be useful to gain information regarding the minimum forest patch size utilized by the species.

Despite *Ciconia stormi's* persistence in disturbed areas, there are several potentially negative impacts of this forest fragmentation. Basic concepts of island biogeography state that smaller patches of habitat maintain smaller populations of any given species (MacArthur and Wilson, 1967). Additionally, smaller populations have higher extinction probabilities (MacArthur and Wilson, 1967; Boyce, 1992). Based on these concepts, area-sensitive species should suffer higher extinction rates in landscapes composed of smaller forest patches (Boulinier et al., 2001). Given that *Ciconia stormi* likely employs a high-grading foraging strategy, and only a portion of Borneo’s rivers are accessible to at any given time (see section 3.4.1b), the species is likely to require large areas of habitat from which to select the highest quality foraging grounds. Thus, the degradation of riparian systems associated with deforestation is likely to further reduce the number of foraging grounds available to the species.

At a regional scale, closely clustered forest patches may allow *Ciconia stormi* to easily access other patches of viable habitat if the travel route does not pass through areas with too much human activity, as was shown with the white shouldered ibis (Sozer and Nijman, 2004). The decrease in contagion values between 1993 and 2004 show a greater segregation of forest patches. Land managers should monitor this trend to ensure *Ciconia stormi* has continued access to forest patches of high canopy cover. Radio telemetry data could also provide valuable information on the distance *Ciconia stormi* travels when prospecting for feeding sites.

4.4.6 Assumptions and Limitations

The results should be considered in light of the limitations of post-classification change detection, which relies on a per-pixel comparison of land cover classes for two distinct dates. This methodology can lead to greater rates of error propagation as an error in the accuracy of either date leads to a false indication of change (Singh, 1989; Serra et al., 2003). Regardless, post-classification comparisons are widely used in studies of land cover change (Belisle et al. 2001; Nagendra et al., 2006; Cakir et al. 2008).

Despite efforts by Fuller and Murphy (2005) to remove cloud cover from the MODIS images, some cloud and haze may have been present in the images. Persistent cloud cover may have affected the initial classification because clouds absorb and scatter electro-magnetic radiation, thus altering the level of radiance detected by the sensor. Furthermore, if clouds obscured forested areas in 2004, those areas would have been classified as non-persistent forest in the forest change analysis, which may result in over-estimating forest loss and forest fragmentation between 1993 and 2004.

Potential disagreement between the data used to quantify forest cover arises because the satellite-based instruments record forest cover from a downward pointing nadir-view while hemispherical photographs record forest cover from the lowest level of the forest canopy, which has both vertical and horizontal dimensions. This discrepancy may reduce agreement between soft-classification products derived from satellite imagery and hemispherical photographs because the hemispherical photographs capture canopy conditions at different zenith angles, potentially including more vegetation than optical satellite data which only records vegetation that is visible from above.

At this time it is difficult to speculate how canopy structure influences habitat selection by *Ciconia stormi*. Future work might take advantage of active remote sensing platforms, specifically the ability of RADAR and LIDAR to yield better information on sub-canopy structure found in areas the species occupies.

4.5 Conclusions

Many forest-interior bird species have become extinct in areas of intensive logging, or exist in such low densities that they are effectively ecologically extinct

(Redford, 1992; Bennet and Robinson, 2000). Certain species are particularly vulnerable to the effects of logging, particularly species that utilize understory habitats, as is the case for *Ciconia stormi* (Lambert, 1992; Ford and Davison, 1995; Johns, 1996). The effects of deforestation may be particularly detrimental to *Ciconia stormi* because it both nests and forages in forested landscapes, as compared to many other species that utilize forest interiors for nesting but not hunting (Jiguet and Villarubias, 2004).

This analysis shows that ten years following heavy logging, average canopy conditions match average canopy conditions in areas of potential *Ciconia stormi* presence. However, ten years after logging large gaps are still prevalent, which may limit *the species'* ability to utilize large portions of these forests. Twenty years after selective logging, the occurrence of large gaps are much less prevalent and average canopy conditions were greater than 87 percent, exceeding the 86 percent canopy cover associated with locations where *Ciconia stormi* forages (figures 3.7 and 4.3). This indicates that selectively logged forests may meet forest cover requirements of *Ciconia stormi* twenty years following logging.

The average forest patch size in areas of potential *Ciconia stormi* presence was over 1,300 ha, with the average size of forests patches used by the species was over 1,100 ha. Mean core areas of forest patches in areas of potential *Ciconia stormi* presence exceeded 7,000 ha. Problems with the 1,300 ha figure arise from the influence of MAUP and the development of areas of potential *Ciconia stormi* presence. At the same time, the average size of forest core areas used by the species may be overstated due to the presence of a single large forest patch containing a large core area. Portions of this forest may not be used by *Ciconia stormi* (i.e. portions of may not have rivers with high fish densities, or they may be at the upper edge of the species elevation tolerance). The minimum forest patch size required by *Ciconia stormi* may lie somewhere between the 1,100 ha the 7,000 ha figures, and future data obtained from radio telemetry work may help add a level of precision to these findings. This analysis showed approximately 50 percent of areas of potential *Ciconia stormi* presence are covered by at least 85 percent canopy cover (see section 3.4.5), indicating that maintaining this proportion of the landscape as viable foraging habitats may be important for the persistence of the species.

Chapter V

5.0 Final Conclusions and Further Discussion

This thesis assesses the habitat needs of an endangered bird, *Ciconia stormi*, on the island of Borneo. Because of the small number of precisely geo-referenced sightings, a methodology was developed that mapped a fuzzy-distribution of the species, where locations were assigned a value based on the confidence or likelihood that a sighting occurred at that particular location.

This thesis also quantified forest cover requirements of *Ciconia stormi* at three different scales. Average canopy cover is over 70 percent within areas of potential *Ciconia stormi* presence when assessed at an island scale using 500-meter MODIS data. Average canopy cover derived from 30-meter Landsat data at locations of precisely geo-referenced sightings (those marked with a GPS) is over 90 percent. Finally, hemispherical photographs obtained at locations where *Ciconia stormi* was seen foraging or roosting reveal an average canopy cover of approximately 86 percent. This multi-scaled approach is important, because wildlife select habitat at multiple scales, including selection of the geographical distribution of a species; selection of individual home ranges within the geographical distribution; selection of habitat components within home ranges; and selection of specific sites used for foraging or nesting (Johnson, 1980). Conservation efforts should address species habitat needs at each of these scales, because if even if habitat is protected at one scale, failure to protect sufficient habitat at other scales may still result in species extinction. For example, an area managed to meet the forest cover requirements of the species at a regional scale (70 percent tree cover) may still fail to contain a sufficient number of suitable foraging sites (over 85 percent tree cover).

The level of confidence in *Ciconia stormi* presence increases in areas with over 75 percent canopy cover, indicating this may be an important forest cover requirement of *Ciconia stormi*. A similar threshold was identified at 550 meters in elevation where confidence in presence begins to decline

Bird life international (2001) states that "Records from disturbed, recently burnt and logged habitats give no indication of the long-term value of such areas to the species, and until research proves otherwise it is sensible to assume that such areas are suboptimal, or at least offer only temporary benefits." Hemispherical photographs were used here to analyze forest canopy conditions at locations that had been logged at different times and at different intensities, and link those conditions to canopy cover requirements of *Ciconia stormi*. Disturbed forests displayed a progressive closure of canopy over time. Soon after logging, a complex mosaic of canopy conditions exists followed by a progressive homogenization of canopy conditions over time after logging. This thesis found that selectively logged forests have canopy conditions that may meet the requirements of *Ciconia stormi* 20 years after logging, although reduced stream biomass and biodiversity may still limit the species' ability to inhabit these forests.

Mean forest patch size in areas of potential *Ciconia stormi* presence was found to exceed 1,100 ha, although the mean core areas potentially used by *Ciconia stormi* were over 7,000 ha, making it difficult to assess minimum forest patch size required by the species (i.e. the species may not use the small and medium sized forest patches that fall within its range). Radio telemetry data should be able to provide the data on home-ranging behavior necessary to develop a complete management plan for the species.

5.1 Limitations and Assumptions

Efficient allocation of resources leads to greater aggregation of data at broader scales (Weiers et al., 2004). For example, limits of computational power mean that it would be impractical to assess worldwide deforestation levels at one meter resolutions. At the same time, management decisions that occur at a regional or sub-regional scale depend on higher levels of data accuracy and detail to carry out operational tasks such as the accurate delineation and enforcement of protected areas (Weiers et al., 2004).

Scale poses significant challenges to this research. Perhaps the most pressing is the influence of the Modifiable Areal Unit Problem (MAUP) on the analysis. As described in section 4.3.2, MAUP is the problem of obtaining different results through use of artificially designated sampling units of different size or location, rather than ecologically meaningful boundaries (Jelinski and Wu, 1996; Li et al., 2006). In the absence of precisely geo-referenced presence data, and lack of information regarding the species' home-ranges, a decision had to be made as how to best define the unit of analysis. To address MAUP, the presence data was mapped along a gradient of high to low confidence of presence. Results were presented at different confidence levels, each of which covered a different spatial extent. By presenting the results for each of these different sized areal units, the affects of MAUP were more clearly revealed.

5.2 Final Note

Presently, selective logging criteria for much of Borneo are based solely on a minimum basal diameter cutting limit (Sist et al., 2003). This approach is predicated on maximizing the economic return of timber resources and is unlikely to result in ecologically sustainable forests (Sheil and van Heist, 2000). A more complete conservation paradigm should include biological, social, political and economic criteria (Meffe and Carroll, 1997). However, balancing these issues is complicated and various approaches may be applicable depending on the context. This research attempts to find a balance between resource maximization and ecological approaches by identifying the amount of time after logging necessary that may meet the habitat requirements of a particular species.

However, even if more ecologically based guidelines are adopted, logging activities need to be carefully managed and implemented. The collateral damage caused by selective logging practices can be severe. For example, in a lowland forest in the Brazilian Amazon where only 2 percent of selected trees (those greater than 10 cm at breast height) were logged, 26 percent of the remaining trees of equivalent size were destroyed or damaged and canopy cover was reduced by almost 50 percent (Uhl and Viera, 1989). This is in addition to the damage brought on by the construction of logging roads, soil compaction and alterations to forest systems that continue to manifest themselves over time such as the increased risk of tree pathogens, and elevated rates of fire and wind damage (Frumhoff, 1995). Therefore it is imperative that any action taken to enhance the conservation value of harvested forests focus on the final conditions of forest structure, rather than on extraction practices alone.

Basing forest management criteria solely on the needs of *Ciconia stormi* is unlikely to convey protection to all species and processes present within those forests, particularly when the ecosystems are as complex and interdependent as tropical rainforests (Lindenmayer et al. 2002). For example, maintaining over 85 percent canopy cover via selective logging techniques and long fallow periods may still lead to the loss of important fruit trees that particular species are dependent upon. However, using charismatic species to spearhead conservation efforts on Borneo has been an effective strategy to raise funds and public awareness that are important for forest conservation (Rose, 2007). LUCC and forest loss on Borneo is driven in part by federal policies such as the BAL and transmigration (Siegert et al. 2001; Fearnside, 1997). These policies are structured to take advantage of Borneo's forests through settlement and resource extraction. CSS may provide a mechanism to increase the economic (e.g. eco-tourism) and cultural (presence of rare species) value of forests, with the stipulation that ecosystem services must be maintained. This may influence land management decisions made by the Indonesian government by providing incentives to manage forests as other than resource commodities.

Ultimately, the survival of *Ciconia stormi* will depend on conservation of sufficient blocks of lowland forest, undisturbed riparian habitats, and improved control of burning and logging practices. Given recent reports of deforestation, fires and illegal logging throughout Borneo (and Indonesian Borneo, in particular), it seems unlikely that the species will persist in the face of multiple anthropogenic pressures on its habitat. If the species becomes extinct, it will be one of a growing list of rainforest dwellers about which little is known. It is not only the birding enthusiasts who will mourn the loss of another avian species, but the also the forest-dwelling people of Borneo whose livelihoods depend upon the existence of intact, functional forest ecosystems to perpetuate their way of life.

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