A Landscape Analysis of Forest Loss and Land Cover Change, 1998-2008

in the Alexander Skutch Biological Corridor, Costa Rica

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Abstract

The Alexander Skutch Biological Corridor (ASBC), located in south-central Costa Rica, links forest fragments in high elevations to forest fragments in lower elevations through two protected areas- Las Nubes Reserve and Los Cusingos Bird Sanctuary. Given that lowland rainforests are one of the most threatened ecosystems in Central America and that the ASBC traverses across three Holdridge ecological life zones, the corridor represents an area of ecological significance (Powell et al., 2000). Despite its importance, the extent and location of remaining forest areas in the corridor has not been evaluated since the late 1990s. Current information is crucial for the conservation and sustainable management of lands within the corridor, which is the primary operating goal of the ASBC.

A 2008 analysis of forest cover and land use in the ASBC was undertaken using GIS and remotely-sensed data to assess the degree of forest loss and landscape change in the corridor since 1998. The overall goal of this study is to identify opportunities for ecological restoration in the corridor in order to strengthen the ecological integrity of the region. Interviews with corridor residents were conducted to gain insight into the socio-economic drivers of land use changes, while FRAGSTATS software was used to compute descriptive statistics of forest cover change between 1998 and 2008.

Study results reveal that the ASBC has lost 19% of its forest cover since 1998, with a corresponding decrease in average patch size from 92.14 ha in

1998 to 78.01 ha in 2008. Smaller forest patches in combination with a higher shape complexity have also lead to a 15% decrease in total core habitat area in the corridor. Remnant forest patches were shown to be concentrated in the northern or southern sections of the corridor, with less than 15% of remaining forest cover located in the central regions. In contrast, connectivity between remnant forest patches increased from 39% in 1998 to 73% in 2008, suggesting that the negative effects associated with habitat isolation may have been reduced over time.

Inconsistencies between data sources from 1998 and 2008 did not permit a direct comparison of land use changes over time. However, field observations, interview results, and external data sources indicate that shade-grown coffee plantations have decreased in the corridor at the expense of increasing pasture, sugar, and pineapple plantations.

The overall loss of forest since 1998, particularly in lowland regions, in combination with increasingly intense land uses, threatens the ability of native species populations to persist in the corridor. As such, the ecological restoration of key habitat areas is essential if the long-term protection of biodiversity is to be achieved. Areas prioritized for restoration include internal patches within large forest areas, buffering small forest patches from external stresses, and restoring vegetated corridors along major rivers in the ASBC. Market-based incentives, such as Costa Rica's Payment for Environmental Services (PES) program, and the efforts of community organizations will play key roles in strengthening the conservation ethic and success of the corridor.

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Foreword

When it comes to protecting natural ecosystems from human-induced threats, knowledge alone simply does not cut it. To me, it is like having only half a solution or having only half the tools. One of the main goals I set out to achieve in the Masters in Environmental Studies (MES) program was to translate knowledge into positive environmental change. The difficulty in this kind of approach was not only felt in navigating the gap between the ideal and the realistic in much of my coursework in the MES program, but also in orienting the results and recommendations from my Major Research Paper (MRP) towards on-the-ground applicability. Despite much trial and tribulation, I feel that this MRP has fulfilled many of the learning objectives outlined in my Plan of Study, and has enabled me to contribute something meaningful to the Alexander Skutch Biological Corridor.

At the very least, assessing ecological conditions on a landscape scale requires a multi-disciplinary approach. It is not enough to understand which landscape elements are most needed to uphold ecological integrity over time. Intimately tied to this is understanding how native biota respond to modified landscapes, and how protected areas can serve to prevent further degradation. The task of designing a restoration strategy for the corridor further illuminated the need to combine site-specific field studies with the values and socio-economic factors of landowners residing in the corridor. The heterogeneous nature of many landscapes that have been modified by humans demands a strong understanding how these elements fit together and interact with one another. The breadth of this kind of approach must be inclusive across time and space if ecological conditions are to be assessed effectively. As such, this paper contributes to the larger initiative of sustainable land use management for the ASBC by acknowledging and discussing how and why ecological conditions have changed over time, and under what conditions effective ecological restoration can take place.

Through this paper, I was able to combine the major theories, principles and studies related to landscape ecology, conservation biology, protected areas management, and restoration ecology. The use of GIS technology in this project also allowed me to integrate my passion for analyzing environmental trends on a landscape-scale with my desire for practical applications. It is my hope that the results from the present study will be used to enhance the environmental management of the ASBC and will lay the foundation for future research on opportunities for ecological restoration and conservation in the corridor.

The structure of this paper is divided up into five chapters. The first chapter introduces the problem of habitat loss in the tropics, outlines the major conservation initiatives in Costa Rica, and provides a description of the study goals and study area. The second chapter delves more in-depth into the biological and ecological consequences of habitat loss in the tropics by reviewing relevant theories and findings from recent studies. Chapter Three is a comprehensive description of the methods used to conduct the present study, including fieldwork and data processing steps. The first section of Chapter Four presents the results of this study through a series of GIS-produced maps, while the second section discusses the conservation implications of the study results. Finally, Chapter Five contains a set of recommendations for enhancing the ecological integrity and sustainable land use management of the corridor, based on the results presented in Chapter Four.

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Chapter 1. Introduction

1.1 Problem Statement

Habitat loss and the conversion of forested environments to human-dominated landscapes are inherently problematic for the conservation of terrestrial tropical ecosystems. The interrelated processes of habitat loss, degradation, fragmentation, and isolation are arguably some of the most pervasive and ubiquitous threats to the conservation of biodiversity worldwide (Develey & Metzger, 2006; Sánchez-Azofeifa et al., 2003). Forest fragmentation results from the dividing up of forested areas into smaller parcels, while isolation is a function of the distance between remnant forest patches (Forman, 1995). Both phenomena stem from the process of habitat loss and/or land transformation, and can contribute to the ecological degradation of a region (Forman, 1995).

According to the World Conservation Union (IUCN), these processes negatively affect between 83% and 91% of all mammals, birds, and plants (Dirzo & Raven, 2003). This is largely because most tropical species are highly specialized and tend to exist in patchy, low-density populations (Laurance et al., 2002; Kricher, 1999). Although these distribution patterns have produced some of the most biologically diverse areas in the world, it also means that tropical species tend to be intrinsically rare and endemic (Sodhi et al., 2007; Laurance et al., 2002; Kricher, 1999). Thus, any loss in forest cover is more likely to result in the loss of sensitive or specialized species in a tropical environment than in other environments (Turner, 1996). Coupled with the loss and degradation of essential ecological services such as water purification and crop pollination, disturbed natural landscapes are rapidly losing their ability to sustain species populations and ecological processes over a long period of time (Sodhi et al., 2007; Laurance et al., 2002).

Neotropical forests, in particular, are being lost at an alarming rate due to encroaching human settlements and to agricultural, pastoral, and silvicultural activities (Sekercioglu et al, 2007; Dirzo & Raven 2003). By 1980, for example, nearly 60% of lowland rainforests in Central America had been deforested, mostly to make way for cattle ranches (Kricher, 1999). These trends have transformed once-continuous forested landscapes into scattered forest patches, often embedded in an agricultural landscape (Daily & Ehrlich, 1995).

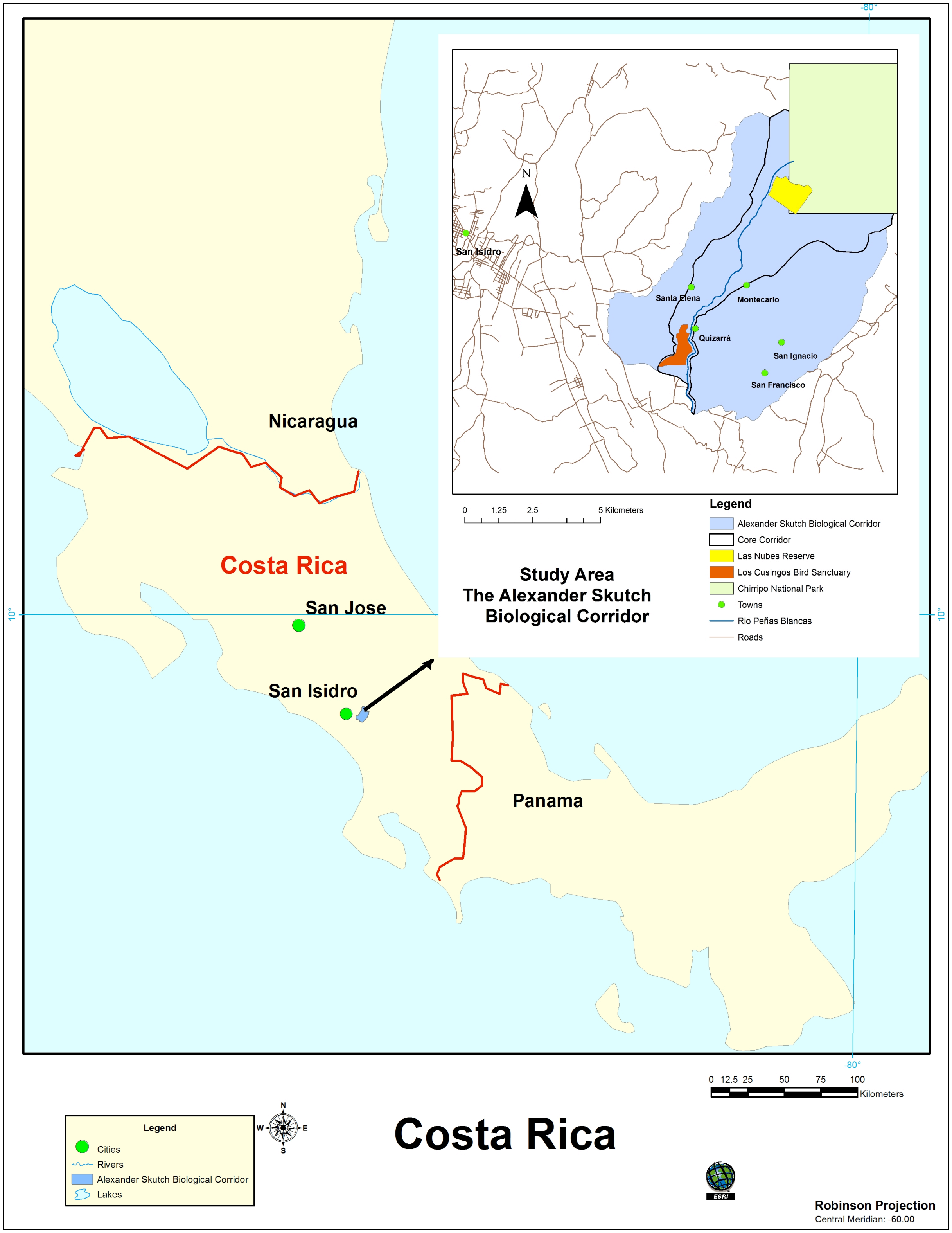
One approach to reducing habitat destruction and species loss is establishing protected areas that limit or prohibit destructive activities. However, as is the case in Costa Rica, many protected areas around the world tend to be too small, too isolated or are situated on marginal lands to adequately protect species populations over time (Lindenmayer & Fischer, 2006; Daugherty, 2005; Sánchez-Azofeifa et al., 2003; Powell et al., 2000). The increasing recognition that protected areas are not enough to guarantee the protection of species and ecological processes has led to a renewed emphasis on the sustainable management of lands outside of reserves (Gascon et al., 1999). The application of a landscape ecology approach for the management of fragmented landscapes has gained popularity as a means of assessing ecological conditions on a kilometres-wide scale (Forman, 1995). Although many definitions of landscape ecology exist, the discipline essentially involves the study of how the spatial arrangement and composition of habitat and non-habitat areas serve to influence, and are influenced by, organisms and their environment (Bennett, 2003; Forman, 1995). Landscape ecology recognizes that the effective conservation of habitats cannot be accomplished without taking into account the characteristics of an entire landscape mosaic (Lindenmayer & Fischer, 2006; Saunders et al., 1991). In other words, the type, number, size, proximity, shape, and location of habitat and non-habitat patches can affect the degree of degradation a particular landscape may experience because certain spatial patterns are more conducive to supporting ecological integrity than others. A system with ecological integrity has 'near-natural' levels of plant production, biodiversity, and soil and water characteristics, given the processes, cycles, and species historically associated with that system (Forman, 1995). Generally, the more disturbed a system is, the less ecological integrity it will have, and the less able it will be to deliver essential ecological services and to sustain viable species populations.

A landscape management approach requires looking beyond designated protected areas as the only mechanism for mitigating habitat destruction and species loss. As such, conservation opportunities and constraints present in a heterogeneous landscape must be identified before meaningful action can be taken (Jensen, 2007; Forman, 1995; Saunders et al., 1991). Action, in this sense, most often takes the form of sustainable land use management, such as the conversion of intensive crop monocultures to integrated crop polycultures, or through the process of ecological restoration. Ecological restoration is a means of assisting the recovery of a degraded ecosystem to a more natural or historically characteristic state (Lindenmayer & Fischer, 2006). Natural regeneration and planned reforestation are both methods of ecologically restoring a landscape, and can lead to improved levels of ecological integrity within a given area (Bennett, 2003).

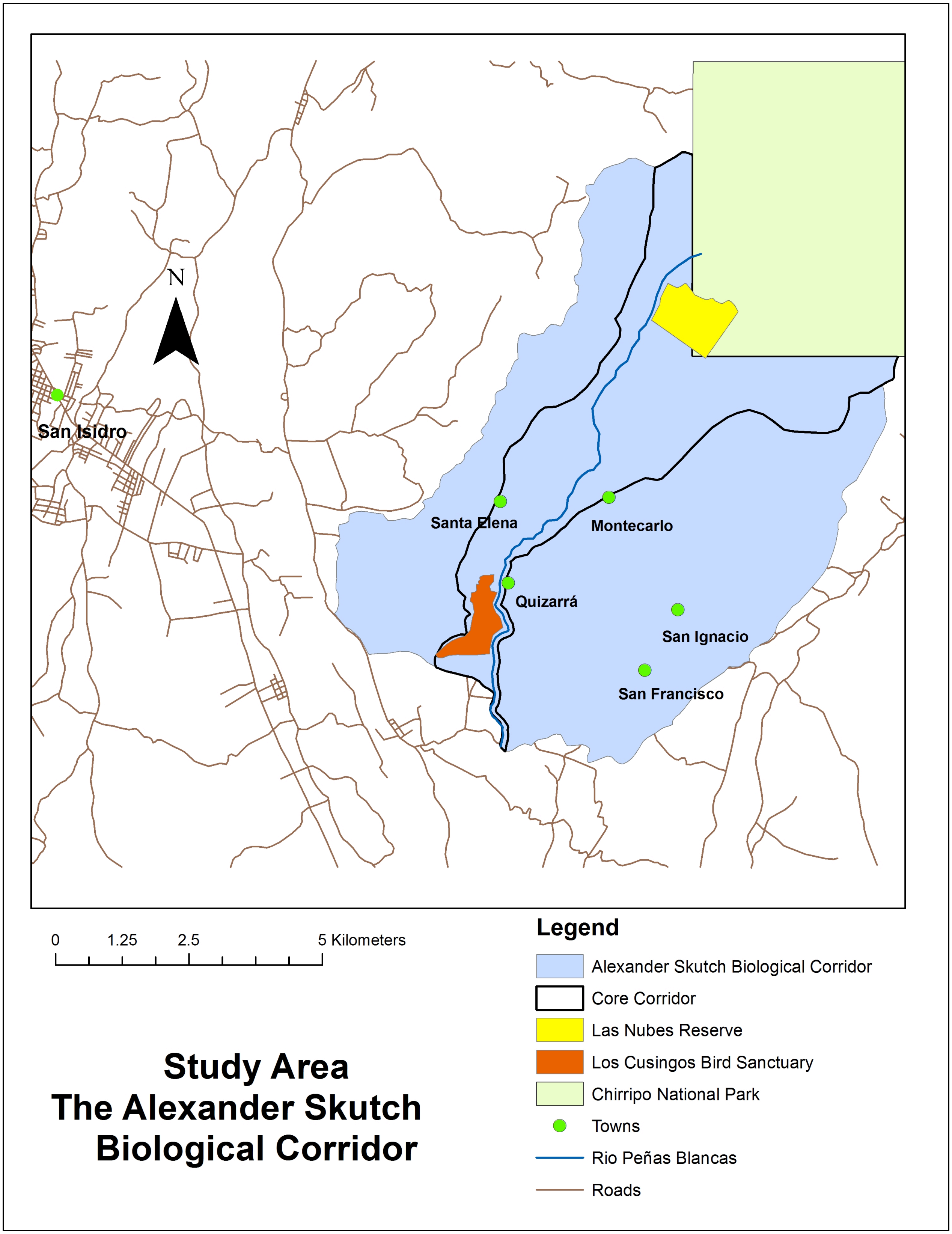
Typically, efforts to restore a fragmented landscape are directed towards re-establishing habitat patches, creating linkages between patches to reduce the effects of isolation, and/or buffering existing habitat patches from anthropogenic stresses (Lindenmayer & Fischer, 2006; Bennett, 2003; Forman, 1995). However, without knowing which landscape elements are most needed to maintain ecological integrity over time, restoration efforts can be misplaced. Thus, a central challenge in the conservation of modified landscapes is the ability to characterize and evaluate patterns of forest fragmentation and isolation in a way that is meaningful for conservation planning and management initiatives.

1.2 Background: Deforestation Trends and Conservation Initiatives in Costa Rica

Costa Rica is a small Central American country of 51,000 km^2 that lies between Nicaragua and Panama (Arroyo-Mora et al., 2005) (Map 1.1). Many of the conservation challenges typically faced by tropical developing countries can



Map 1.1: Map of Costa Rica showing the study area for the present study.



Map 1.2: The Alexander Skutch Biological Corridor. From here forward, the "Core Corridor" will be listed as part of the "Alexander Skutch Biological Corridor" in map legends.

be observed in Costa Rica's land use and settlement history since the 1940s (Sader & Joyce, 1988).

Between 1940 and 1984, Costa Rica experienced one of the highest deforestation rates in the world, resulting in the loss of over 60% of forested area within the country (Sader & Joyce, 1988). The principle cause of forest loss was the demand for land to accommodate increasing populations and agricultural expansion, namely beef for export (Kricher, 1999). During the 1960s and 1970s, the Government of Costa Rica took advantage of the increase in international beef prices and, with international assistance, offered financial incentives to farmers who cleared land for the production of cattle (Arroyo-Mora et al., 2005; Sánchez-Azofeifa et al., 2001). By 1989, the amount of pastureland in Costa Rica had reached 2.4 million hectares, or roughly 48% of the entire country (Arroyo-Mora et al., 2005). These deforestation trends were projected to seriously degrade or eliminate remaining forest patches by the year 2000 if preventative measures were not taken (Powell et al., 2000). Since the 1960s, the Government of Costa Rica has gradually designated over 25% of the country as protected areas, and has enlisted a number of policies to encourage the conservation and restoration of forest habitats on private lands (for example, payments for environmental services) (Sánchez-Azofeifa, et al., 2003).

Despite these efforts, habitat degradation and loss is increasing in the areas outside of reserves, effectively isolating remaining forest areas (Sánchez-Azofeifa et al., 2003; Sánchez-Azofeifa et al., 2001; Powell et al., 2000). For example, as of 1991, 71% of remaining forest habitats were found to be outside the boundaries of established protected areas (Sánchez-Azofeifa et al., 2001). A gap analysis conducted by Powell et al. (2000) revealed that only 9 out of 23 Holdridge life zones in Costa Rica had greater than 10% of their total area protected. Ecological life zones were defined by Leslie Holdridge in 1967 according to the temperature, precipitation and elevation gradients occurring in a particular area that determine the plant communities and species associations that will theoretically occur in that area (Powell et al., 2000). The most threatened life zones identified in the gap analysis were found to be seasonally dry habitats in the northwest, and Pacific slope habitats between 500 and 1500 meters above sea level (Powell et al., 2000). Given that Costa Rica contains 6% of the world's plant and bird species, and that many Neotropical species are not adapted to cope with fragmented landscapes, more attention must be paid to the conservation and ecological restoration of forest habitats occurring outside of reserves, especially in areas that are most threatened by anthropogenic activities (Arroyo-Mora et al., 2005; Sánchez-Azofeifa et al., 2001). Without these efforts, remaining forest habitats will continue to be lost and degraded, and vast assemblages of flora and fauna in Costa Rica will face a tremendous risk of extinction (Dirzo & Raven, 2003).

1.3 Research Question and Objectives

This study examines opportunities for ecological restoration in a modified landscape in south-central Costa Rica based on changes in land use trends over time. Specifically, the present study attempts to characterize the spatial pattern of remnant forest patches in the Alexander Skutch Biological Corridor (ASBC) in order to evaluate the degree of habitat fragmentation and isolation in the corridor (Map 1.2). In addition, this study seeks to identify overall land use changes that have occurred since 1998, when the concept of the ASBC was first proposed. Evaluating current and past land use trends in a heterogeneous landscape can provide insight into which areas may be most threatened by deforestation or degradation in the future, and which areas may be most limiting to the maintenance of ecological integrity. Thus, a temporal understanding of land use changes in the ASBC serves as a reference framework for situating areas to be prioritized for restoration.

Although identifying areas for restoration in the ASBC is an overall goal, the present study does not aim to draw any definitive conclusions about the most appropriate methods of restoration for the area. Rather, the purpose of identifying areas where future restoration efforts should be focused is merely to provide a set of recommendations about restoration output (i.e. enlarging a habitat patch vs. enhancing connectivity) from which further studies on ecological restoration in the corridor can take place.

With this in mind, the present study has the following objectives:

i) To assess the degree of forest fragmentation and isolation in the corridor over the past decade.

ii) To produce an up-to-date map of forest areas in the corridor and land uses immediately surrounding these areas.

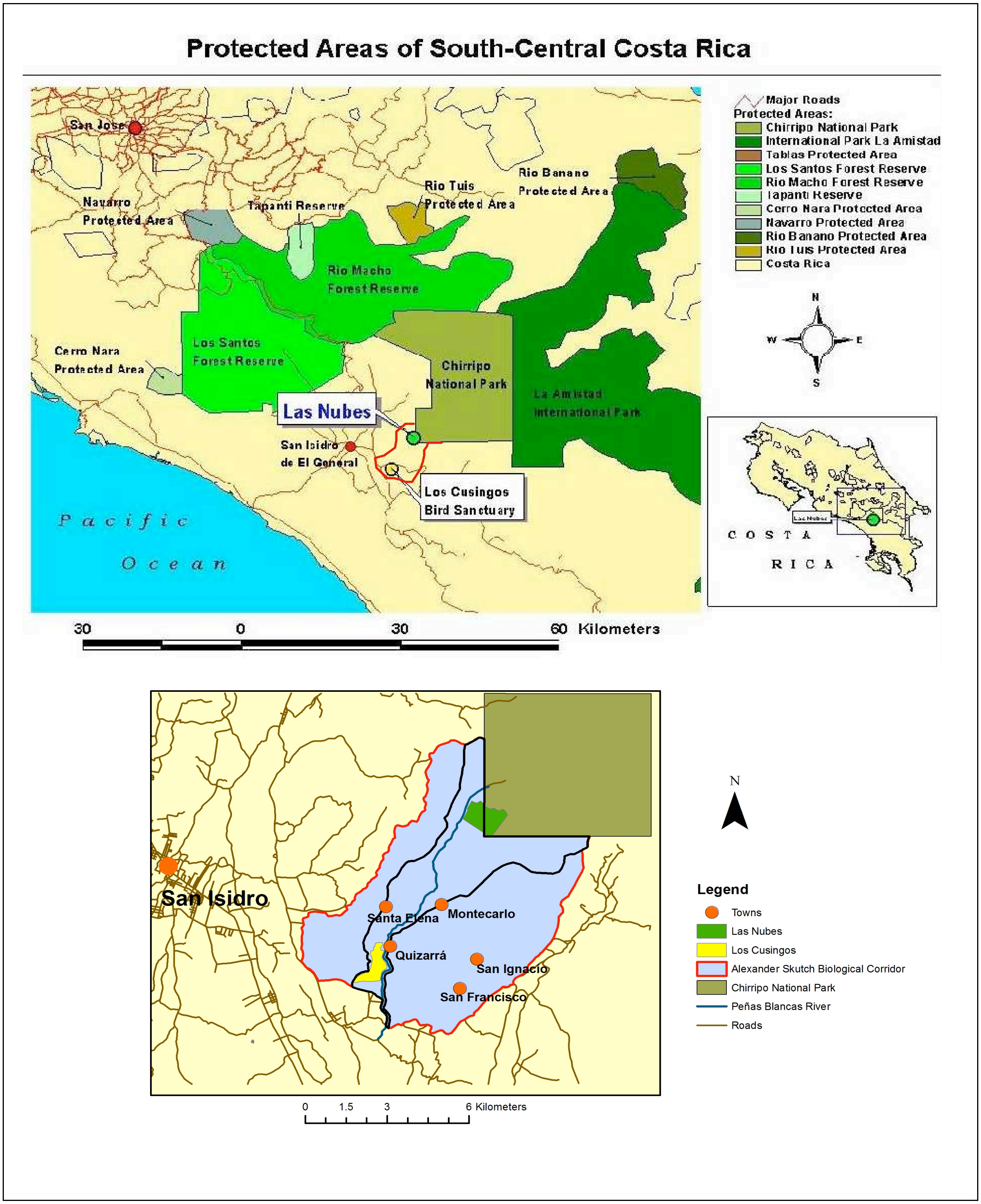
iii) To use this map to prioritize areas for ecological restoration in the corridor.

iv) To describe the general causes and drivers of land use change in the corridor over the past decade from a socio-economic perspective.

1.4 Study Area: The Alexander Skutch Biological Corridor

The Alexander Skutch Biological Corridor is situated on the Pacific side of the Talamanca Mountain range, in the central province of Pérez Zeledón, Costa Rica. The corridor is composed of a core zone, which links Las Nubes Biological Reserve and Los Cusingos Bird Sanctuary, and a buffer zone, which encompasses the watershed lands of the Río Peñas Blancas (Map 1.2).

Las Nubes Reserve is a 124-ha area of mostly primary montane rainforest, ranging in elevation from 1100 meters to 1500 meters above sea level (Daugherty, 2005). The reserve was donated to York University in 1998 by Dr. M.M. (Woody) Fisher, and is currently managed by the Tropical Science Center (TSC) in San José through and agreement with York University (Daugherty, 2005). The reserve is home to the headwaters of the Río Peñas Blancas, which supplies drinking and irrigation water to parts of the ASBC. Las Nubes also connects to Chirripo National Park, and by extension, La Amistad International Biosphere Reserve, which extends into Panama (Map 1.3). The mountainous area around Chirripo National Park is one of the few places in Costa Rica where one can find large-bodied mammals such as the jaguar, which require large tracts of forest to meet their resources needs (Laurance, 2002; Young, 2001).

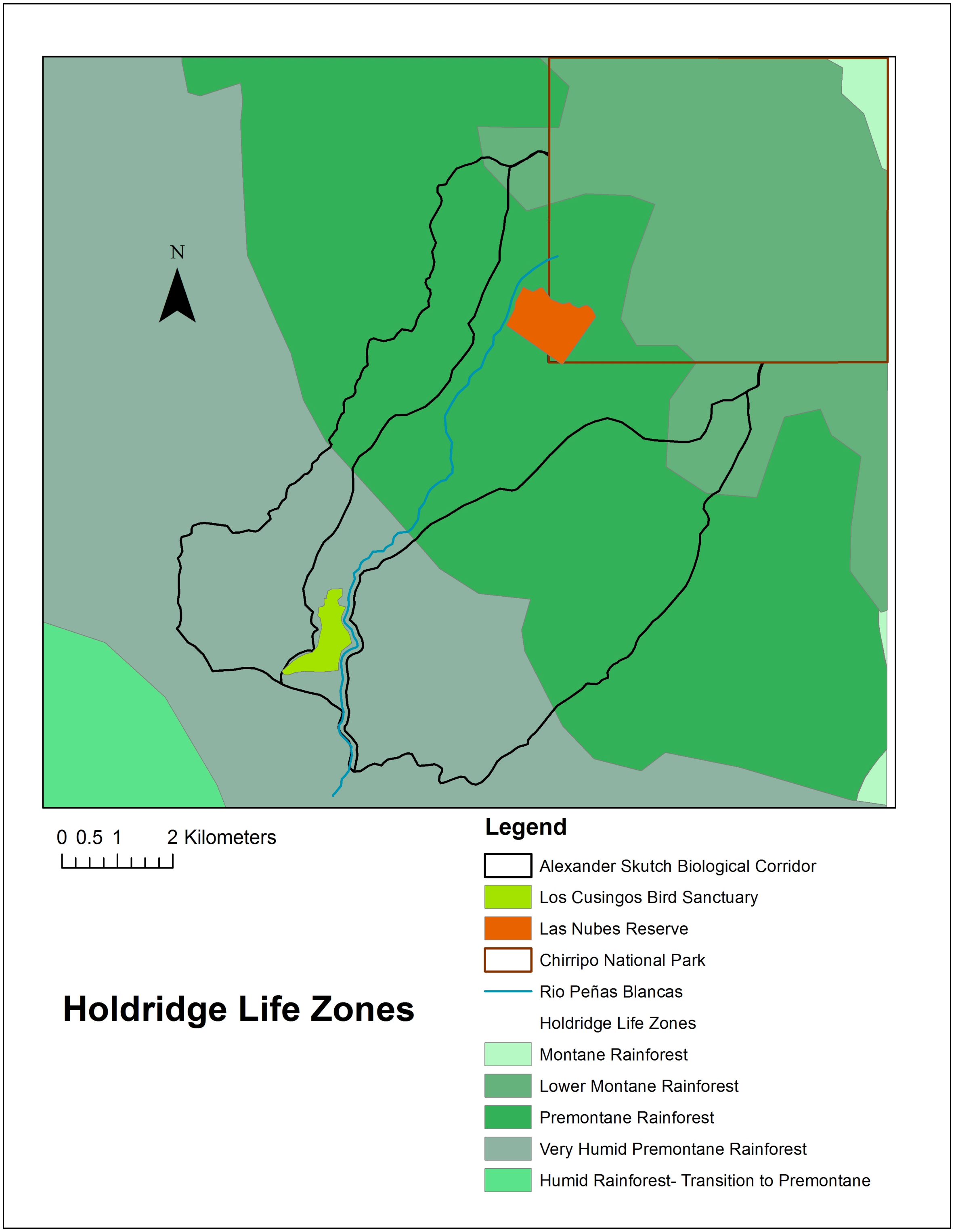


Map 1.3: The Alexander Skutch Biological Corridor and protected areas in the region. Context map from www.yorku.ca/lasnubes.

In contrast, at the southern end of the corridor is Los Cusingos Bird Sanctuary, one of the last remaining fragments of low-elevation rainforest in the region (Daugherty, 2005). This 77-ha reserve was once the homestead of Dr. Alexander Skutch, a world-renowned ornithologist and naturalist, until his death in 2004, whom the corridor is named after. Los Cusingos is situated between 600 and 750 meters in elevation, and is made up of mature and secondary-growth lowland forests. It is currently managed by the TSC as a protected area and as a potential tourist destination.

Collectively, Las Nubes and Los Cusingos create a reserve network for the region that links high-elevation and low-elevation forest fragments. Given that low-elevation forests are one of the most threatened ecosystems in Central America, the ASBC represents an area of ecological significance because it can provide connectivity across altitudinal gradients, which is important for many migratory species (Powell et al, 2000; Kattan & Alvarez-Lopez, 1996). The corridor itself stretches across three Holdridge ecological life zones (Map 1.4). The presence of these life zones may mean that certain species are at the limits of their geographical and altitudinal ranges, and are perhaps more sensitive to the effects of habitat fragmentation and loss (Turner, 1996).

Since the 1940s the area has experienced significant losses in forest cover (Skutch, 1971). When Dr. Skutch established Los Cusingos in 1941, he described the southern end of the corridor as being covered by “stately forests” aside from a few newly established farms (1971, p. 136). But, by the early 1970s, Skutch lamented, “only shreds and patches [of forest] remain in the valley



Map 1.4: Holdridge ecological life zones in the ASBC. That the corridor encompasses three different zones (Lower Montane Rainforest, Premontane Rainforest, and Very Humid Premontane Rainforest) testifies to its ecological significance, not only in terms of connecting habitats in these zones, but also in terms of supporting endemic or specialized species populations.

and on the slopes below four or five thousand feet….", Los Cusingos being one of the largest forest tracts left in the valley (1971, p. 337).

The intensification of agriculture over the past four decades, coupled with an increasing population, has resulted in the destruction and modification of the forests that once covered nearly the entire corridor. A variety of land uses now predominate in the ASBC, including pineapple and sugar cane plantations, sun and shade-grown coffee, pasturelands for livestock grazing, and settled areas. These uses have effectively fragmented and isolated remaining forest areas, particularly in areas below 1000 meters elevation, where gentler slopes have made for more desirable agricultural lands (Sodhi et al., 2007; Sader & Joyce, 1988).

The conservation and ecological restoration of forest habitats in the ASBC is crucial for the long-term maintenance of ecological integrity of the region. Forested areas and their associated species assemblages also hold importance for the social and economic sustainability of the corridor. In recent years, Los Cusingos has gained popularity as a destination for avid birders, and local communities have begun to work together to assess the potential for ecotourism of the entire corridor (Galaski, 2008). What is more, the conservation of riparian corridors and forest fragments are also becoming recognized for their importance in maintaining local water supplies and ability to maintain soil quality.

With these issues in mind, the ASBC was formally established in 2005 with the intention of protecting the biodiversity of the area through sustainable land use management and the ecological restoration of key habitat areas (Daugherty, 2005). Some of the operating goals of the corridor include encouraging farmers to convert sun-grown coffee farms to more sustainable shade-grown coffee farms; restoring riparian forests and degraded pasture lands; and the restoration of degraded lands for water-source protection and for erosion control (Daugherty, 2005). One of the central concepts behind these operating goals is the idea of regional landscape conservation, which Daugherty (2005) argues "may offer the best hope for protection of biodiversity in some tropical areas, including southern Costa Rica" (p. 158). Regional landscape conservation reflects many of the same ideas of landscape ecology where protected areas, ecological integrity, institutional and government policies, and sustainable community development are all viewed as components of an integrated conservation system, as opposed to their individual parts (Daugherty, 2005). However, there is a lack of up-to-date information on the location and extent of remaining forest areas in the corridor. In fact, the last land use studies were conducted in the late 1990s by the TSC and by a York University graduate student (see Young, 2001). Under the concept of regional landscape conservation, the above named operating goals cannot be achieved without current data on forest cover and land uses in the ASBC. Thus, the present study has important applications in achieving these goals, particularly the restoration of key habitats for biodiversity protection.

Chapter 2. Literature Review

2.1 The Ecological and Biological Effects of Habitat Loss

When forested habitats are reduced and fragmented as a result of anthropogenic activities, this process catalyzes the spatial configuration of remaining habitats, which in turn influences species abundance and distribution patterns (Develey & Metzger, 2006). Ecological processes such as disturbance cycles, seed dispersal, and moisture regimes are also impacted by the spatial arrangement of forest patches and the land uses present in the external landscape (Laurance et al., 2002; Endress & Chinea, 2001; Saunders et al., 1991). What this means for the ecological integrity of a landscape, particularly in a tropical environment, is that some landscape patterns are more conducive to sustaining viable species populations and/or characteristic ecological processes than others. In fact, the size of remnant habitat patches, the degree of connectivity between patches, and the type of matrix surrounding habitat patches will essentially dictate the biological and ecological responses to forest fragmentation and loss (Gascon et al., 2001; Gascon et al., 1999; Forman, 1995; Saunders et al., 1991). The matrix is considered to be the most dominant and extensive land use type in a given area (Forman, 1995). In a fragmented landscape, the matrix is typically human-modified areas such as urban or agricultural lands.

Studies in human-modified tropical environments have shown that native flora and fauna, in general, do not respond positively to these changes

(Sekercioglu et al., 2007; Sodhi et al., 2007; Develey & Metzger, 2006; Laurance et al., 2002; Daily et al., 2001; Gilbert & Setz, 2001; Gascon et al., 1999; Kattan & Alvarez-Lopez, 1996; Daily & Ehrlich, 1995). That is, many Neotropical species require large, intact forest areas that are linked to other forest areas in order to meet their resource and habitat needs (Sodhi et al., 2007; Van Dyke, 2003; Kricher 1999). In the absence of minimum habitat requirements, declines of sensitive or specialized species populations may occur that, without preventative action, can result in the local extinction of entire species assemblages (Sodhi et al., 2007; Van Dyke, 2003; Kricher, 1999). Since many Neotropical species have co-evolved ecological relationships, any loss in species is likely to result in secondary losses of species and ecological functions as well (Laurance et al., 2002).

Species most vulnerable to the effects of habitat loss, fragmentation, and isolation tend to be habitat specialists, species with high area and energy needs such as large-bodied mammals and predators, and those species that have limited dispersal abilities (Laurance et al., 2002; Kricher, 1999). Given the immense biodiversity and prevalent levels of endemism in Neotropical rainforests, one can expect a tremendous risk of local extinction and a disruption of historical ecological functions in a landscape that is highly fragmented and modified (Sodhi et al., 2007; Dirzo & Raven, 2003; Van Dyke, 2003; Kricher, 1999).

Understanding the ecological and biological effects of habitat fragmentation and isolation are central in devising strategies to mitigate those effects. Much of this effort has been centred on understanding how the spatial pattern of a landscape shapes the ecological character of a given area (Forman, 1995). A wide array of studies have been conducted in tropical America that examine such conservation issues as species-area relationships (or how much habitat is needed to sustain tropical flora and fauna), matrix/habitat-related edge effects, and the role of small vegetation patches and corridors in creating a landscape pattern that supports native species populations and processes.

2.2 Species-area Relationships

The idea that species abundance levels are inherently tied to the size and degree of isolation between habitat patches was originally founded by Robert MacArthur and Edward Wilson in 1967. Their theory, known as Island Biogeography Theory (IBT), sought to explain why large islands that are closer to the mainland have more species than smaller islands that are more isolated from the mainland. MacArthur and Wilson argued that the number of species on a particular island could be theoretically predicted by immigration and extinction rates, whereby islands closer to the mainland would have a greater colonization rate than those further away, and whereby larger islands were able to support higher colonization rates. MacArthur and Wilson's research contributed to many aspects of conservation biology, including reserve design principles, the effects of barriers and distance, the role of small vegetation patches, and species relaxation/compression hypotheses (Van Dyke, 2003).

Nevertheless, several shortcomings in their theory make applying IBT in a heterogeneous landscape inappropriate. Namely, the idea that habitat patches are figurative "islands" in an inhospitable landscape fails to acknowledge the role of the matrix in providing partial levels of connectivity between patches (Van Dyke, 2003; Forman, 1995). As Forman (1995) argues, "although isolation is a major problem for certain key species, most species can cross [the matrix] at least a low rates" (p. 57). Additionally, since the matrix environment influences many key processes in remnant habitat patches, such as edge effects and material flows, habitat patches can not be viewed in isolation from the surrounding landscape (Sodhi et al., 2007; Gascon et al., 1999; Forman, 1995).

In lieu of these shortcomings, a patch-corridor-matrix model has become more widely applied in examining species-area relationships in heterogeneous landscapes. This model acknowledges that the arrangement of patches and corridors within a matrix essentially determines the structure and flow of organisms, humans, and abiotic elements in a landscape (Forman, 1995). Species-area relationships are understood as a function of the characteristics and spatial configuration of a landscape rather than just a function of habitat patch size and isolation alone (Gascon et al., 2001; Forman, 1995).

One of the central challenges in assessing species-area relationships in a fragmented landscape is the question of habitat size: do large habitat patches sustain higher levels of biodiversity than small habitat patches? And, what is the minimum critical habitat size needed to support viable species populations over time? These questions are important for knowing how much habitat, and in what spatial arrangement, are optimal for the protection of biodiversity.

In tropical environments, a review of the literature makes it hard to refute the hypothesis that only extensive tracts of forest will contain a full complement of native flora and fauna (Daily et al., 2003; Van Dyke, 2003: Laurance et al., 2002; Gilbert & Setz, 2001; Turner, 1996; Kattan & Alvarez-Lopez, 1996). Generally, smaller forest remnants will contain a smaller subset of the original biota and a smaller sample of habitat diversity and resource availability (Laurance, 2002; Turner, 1996). As such, when forested habitats are broken up into smaller parcels, the loss of the original extent of forest will likely result in the decline of native species associated with that ecosystem because certain species can no longer meet their needs in a smaller fragment (Sodhi et al., 2007; Daily et al., 2003; Turner, 1996; Saunders et al., 1991). For example, in the Biological Dynamics of Forest Fragments Project (BDFFP), a 29-year-old project located in Amazonia that examines the effects of fragmentation on biota, researchers found that forest patches of less than 100 ha were too small to sustain three out of six species of primates in the region (Gilbert & Setz, 2001). Similarly, an examination of fragmented landscapes and biodiversity in the Columbian Andes found that only fragments in the range of several hundred hectares could maintain a significant part of the original biodiversity of the region (Kattan & Alvarez-Lopez, 1996). Although no set number exists as to “how big” forest patches should be, there is definite size threshold below which a particular species cannot survive (Van Dyke, 2003). This is not to say that species can only survive in extensive habitat patches, but that large and extensive forest tracts must form the cornerstone of any conservation plan within a fragmented landscape.

Smaller forest patches, in combination with the degree of isolation between remnant patches, may make small populations particularly vulnerable to stochastic events and to habitat degradation (Sodhi et al., 2007; Van Dyke, 2003; Gascon et al., 2001; Turner, 1996). That is, smaller populations are more susceptible to disease, natural disasters, inbreeding, pollution, and edge effects because they are less able to adapt to environmental variation (Van Dyke, 2003; Turner, 1996). Forest patches that are isolated from one another are also more likely to become ecologically depauperate because some species populations are no longer able to participate in genetic exchange or to access suitable habitat on a landscape-scale (Sodhi et al., 2007; Lindenmayer & Fischer, 2006). In another example from the BDFFP, researchers found that even small clearings of less than 100 meters are barriers for many rainforest organisms (Laurance et al., 2002; Gilbert & Setz, 2001). Thus, with regards to the conservation of biodiversity in tropical fragmented landscapes, the general consensus is that forest patches should be as large as possible and as contiguous as possible. A number of studies conducted in Costa Rica have made similar observations about species-area relationships in human-dominated tropical landscapes. Some of these findings are described below.

Case Study: Biodiversity and Agricultural Landscapes Around Las Cruces Reserve, Costa Rica

Las Cruces Reserve is a 227-ha area of primary mid-elevation rainforest, located on the Pacific side of southern Costa Rica (Daily & Ehrlich, 1995). The landscape around the reserve has been heavily modified by agricultural practices since the 1960s, and presently consists of small forest remnants of less than 35 ha amid coffee and pasturelands (Daily et al., 2003). A number of studies have been conducted in the region that explore the relationship between species abundance, habitat size, distance effects, and the conservation potential of agricultural lands (Sekercioglu et al., 2007, Daily et al., 2003; Daily et al., 2001, Daily & Ehrlich, 1995).

In a comparison of biodiversity levels in Las Cruces Reserve and remaining forest patches in the region using small-bodied organisms, Daily and Ehrlich (1995) found that twice as many butterfly species were found in Las Cruces Reserve than in seven remnant patches, ranging between 3 and 30 ha, that were located within 1 kilometre of the reserve. Similarly, when bird species were surveyed in the region, only 53% of the 251 bird species recorded in Las Cruces reserve were found in eight forest fragments ranging from 0.3 to 25 ha in the vicinity of the reserve (Daily et al., 2001). In fact, the forest fragment that shared the most similarity to the avian biodiversity levels found in the Las Cruces reserve was a 25-ha fragment that was connected to the reserve itself. This same study also concluded that 83% of avian species richness could be explained by fragment size alone. The importance of forested habitats to birds in the region was also demonstrated in a study conducted by Sekercioglu et al. (2007). The authors tracked three bird species in forest and non-forest areas in order to assess their habitat use, movement, and foraging patterns in different landscape elements. Despite the fact that only 11% of the study area was covered in forest, two out of the three bird species surveyed spent 69% to 85% of their time in a forested area. Additionally, of the daily movements of these birds, 97% were less than 500 meters in distance, while 74% were less than 100 meters in distance, suggesting that certain bird species are limited to dispersing between local forest patches (Sekercioglu et al., 2007).

A survey of mammalian species was also conducted in the region by comparing diversity levels between Las Cruces, forest remnants, and agricultural lands (Daily et al., 2003). Although the authors found that Las Cruces contained the most recorded species of the sites sampled, they also found that mammalian species richness varied more with habitat type as opposed to distance from the reserve. This finding suggests that habitat composition may be more crucial to the survival of tropical mammal species than connectivity between patches. As such, having one large reserve may not be enough to sustain tropical mammal populations; a variety of smaller reserves, stratified across different habitat types and altitudinal gradients, may also be necessary (Powell et al., 2000; Guidon, 1996; Kattan & Alvarez-Lopez, 1996). Of the 26 mammal species recorded in the study, 35% were found exclusively in forest habitats, 54% were found in both forest and agricultural habitats, and 11% were found only in agricultural sites (Daily et al., 2003).

Clearly, the above studies demonstrate the need for large, intact areas of forest to maintain species populations. What is more, a certain degree of connectivity between forest patches also appears to be necessary for the survival of some bird and butterfly species. However, given the dominance of human-modified lands around Las Cruces reserve, these studies also illustrate the need for sustainable land use management outside reserves in order to increase the habitat value of agricultural landscapes (Sekercioglu et al., 2007; Daily et al., 2003).

2.3 Edge Effects

One of the most prominent effects from the breaking up of habitat into smaller areas is the creation of edges along the boundary of a patch. The severe alteration of a once-intact forest area introduces a new set of ecological and microclimatic conditions along the boundaries of smaller patches, which can subsequently alter the ecological conditions within a patch (Sodhi et al., 2007; Van Dyke, 2003; Forman, 1995). These resulting changes, called edge effects, can include increased light and temperature levels along boundaries, decreased moisture levels, altered wind regimes, changes in seed dispersal and predation, and modified canopy-gap dynamics (Van Dyke, 2003; Laurance et al., 2002, Turner, 1996; Saunders et al., 1991).

Altered microclimatic conditions along the edges of forest patches means that a different assemblage of flora and fauna may become more prominent (Sodhi et al., 2007; Saunders et al., 1991). Specifically, studies of edge effects in tropical environments have found that edges tend to enable generalist or non-native species in a given area (Van Dyke, 2003; Laurance et al., 2002; Gascon et al., 1999; Turner, 1996; Saunders et al., 1991). Considering that most tropical forest species are adapted to interior-forest conditions, the introduction of increased light and heat levels along boundaries may decrease the suitability of that habitat, and instead, allow more tolerant species to survive along edges (Laurance et al., 2002). One of the most striking findings from the BDFFP project in Amazonia was elevated levels of tree mortality due to increased winds following edge creation (Laurance et al., 2002). Researchers demonstrated that abrupt microclimate changes could not be tolerated by certain interior-forest species of trees, resulting in changes in forest structure, composition, and diversity along edges (Laurance et al., 2002). Altered microclimates along edges have also been found to modify nutrient and decomposition cycles, which can lead to changes in resource availability and foraging opportunities for native species (Van Dyke, 2003; Saunders et al., 1991).

Another key finding from the BDFFP was that edge effects are not confined to the immediate boundaries of a patch. Rather, since the projects' inception in 1979, edge effects have been found to penetrate a forest fragment up to 400 meters (Laurance et al., 2002) (Figure 2.1). Although the majority of edge effects were detected within the first 100 meters from patch edges, elevated tree mortality and increased wind disturbance were detected at a distance of over 300 meters from the patch edge (Laurance et al., 2002).

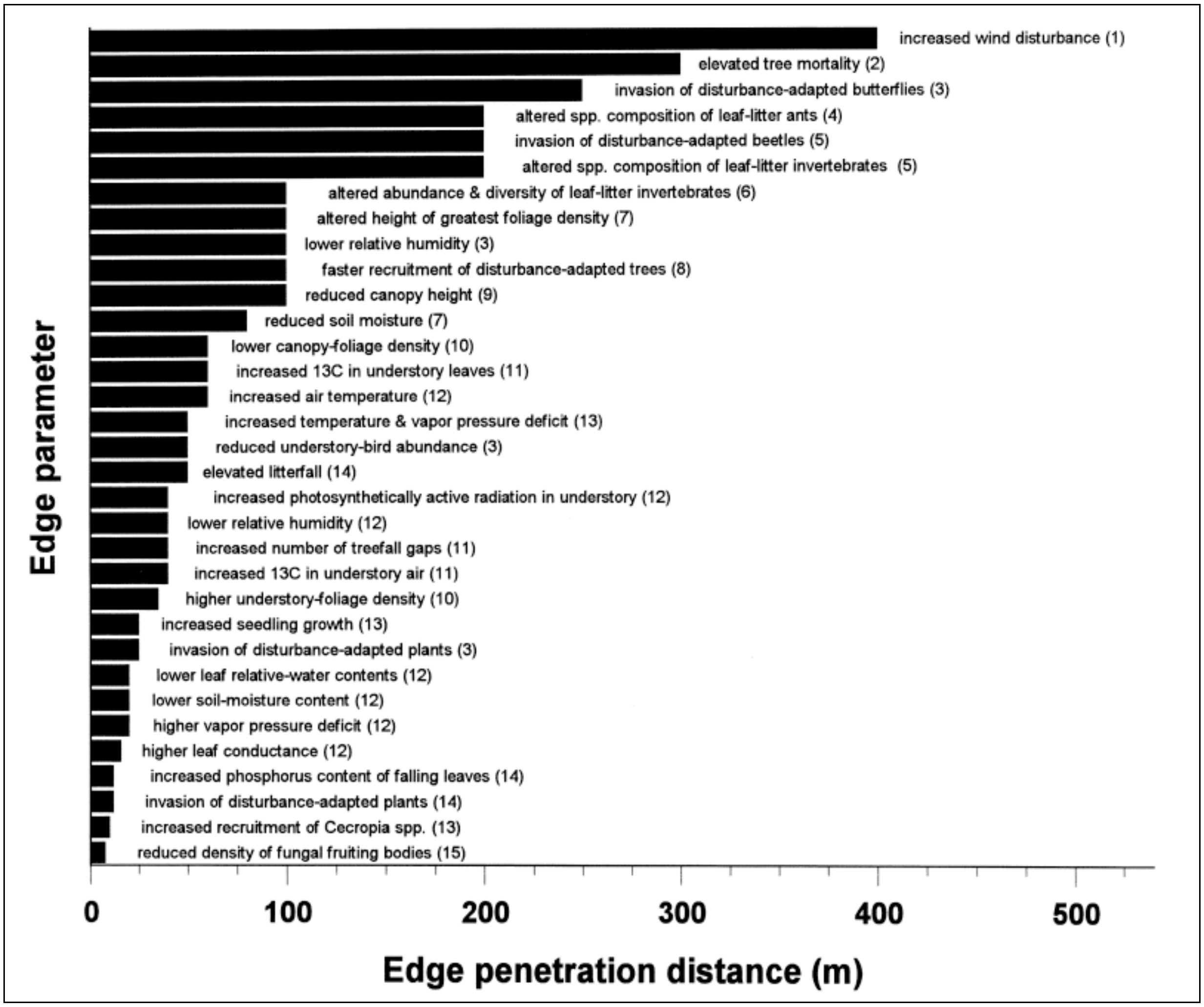


Figure 2.1: Findings from the BDFFP project on the penetration distance of edge effects. Numbers in parenthesis beside each edge effect correspond to the particular study from which those results were found. See Laurance et al., 2002 for study references. Source: Laurance et al., 2002, p. 608.

What this means is that small forest fragments could consist entirely of edge habitat. Since the absence of interior-forest conditions clearly does not bode well for the long-term viability of native species populations, the question then becomes: how do we manage a small forest patch in a way that minimizes edge effects (Saunders et al., 1991)?

In addition to enlarging or buffering a forest patch through ecological restoration, efforts can be directed towards creating more sustainable matrix conditions. The types and intensity of edge effects created from the boundaries between forest patches and the external matrix largely depend on the type of land uses occurring in the area (Sodhi et al., 2007; Laurance et al., 2002; Gascon et al., 1999). In other words, the more intensively the land is used and the more abrupt the boundary between forest and the matrix, the more likely edge effects will have significant impacts on the internal conditions of a forest patch (Laurance et al., 2002; Gascon et al., 1999; Daily & Ehrlich, 1995). As Lindenmayer and Fischer (2006) argue, "the magnitude of edge effects is often strongly associated with the level of contrasts in physical, structural, and other conditions between vegetation remnants and the surrounding matrix" (p. 201). External vegetation structure, therefore, is a key element in reducing edge effects in a fragmented landscape. Creating a "softer" matrix can include allowing natural regeneration to occur along the edges of forest patches, planting more trees and shrubs in agricultural or pastoral lands, and using the land less intensively (Lindenmayer & Fischer, 2006; Daily et al., 2003; Laurance et al., 2002; Gascon et al., 1999). A softer matrix can also provide some connectivity between forest patches, as well as foraging and/or habitat opportunities for species (Lindenmayer & Fischer, 2006; Gascon et al., 1999).

2.4 Small Vegetation Patches and the Role of Corridors

Among the key considerations for species conservation in fragmented landscapes is the role of small vegetation patches and vegetated corridors (linear habitats that connect two or more habitat patches) in reducing the effects of isolation and for providing additional resource and habitat sources (Sekercioglu et al., 2007; Lindenmayer & Fischer, 2006; Van Dyke, 2003). In fact, these features are considered to be nearly as important as the presence of a single large habitat patch for sustaining viable species populations over time (Bennett, 2003; Van Dyke, 2003; Forman, 1995).

In heterogeneous landscapes, one of the challenges in providing sufficient habitat for species is that large forest patches are not very common. In other words, due to pressure from human settlements and activities, large forest patches are likely to become disaggregated and separated by matrix lands over time (Sánchez-Azofeifa et al., 2003; Kricher, 1999). Large remnant patches that are present in a heterogeneous landscape are also likely to be situated on marginal or hard-to-access lands (Schelhas & Greenberg, 1996; Daily & Ehrlich, 1995). In many tropical hillside environments, for example, protected areas tend to be situated below 50 meters in elevation or above 1000 meters in elevation (Guidon, 1996). As a result, the diversity of habitat types present in a landscape may not be adequately represented by large habitat patches alone (Van Dyke, 2003; Laurance et al., 2002; Saunders et al., 1991). Both small vegetation patches and corridors hold importance for providing habitat heterogeneity and a degree of connectivity between larger patches in a landscape because they have greater flexibility in existing alongside human activities (Lindenmayer & Fischer, 2006). Namely, due to their relatively small size or linear shape, smaller forest fragments and riparian corridors are more likely to be incorporated and preserved in human-dominated landscapes (Schelhas, 1996).

Despite the fact that smaller forest patches are less likely to contain the full assemblage of native biota, small patches are still needed to increase the probability of survival for a portion of native biota (Turner & Corlett, 1996). Small vegetation patches have been shown to be important stopover points for migratory bird species, in particular (Sekercioglu, et al., 2007; Kattan & Alvarez-Lopez, 1996). In addition, small fragments can facilitate movement across the landscape for certain species, and provide habitat or breeding opportunities for others (Turner & Corlett, 1996). If a landscape is particularly devoid of adequate forest cover, the forest patches that still exist in the landscape can act as a seed source for reforesting degraded areas (Endress & Chinea, 2001; Holl et al., 2000; Turner & Corlett, 1996). The point is that small fragments in a landscape are better than no forest fragments at all, and, if those small fragments are scattered throughout a landscape that contains a number of large forest fragments, then native biota will have a greater probability of persistence (Turner & Corlett, 1996).

Theoretically, corridors offer a direct response to the effects of habitat isolation (Sodhi et al., 2007; Bennett, 2003). Commonly encountered corridors in heterogeneous landscapes include vegetated riverbanks (riparian corridors), wildlife conduits, powerline corridors, and hedgerows or living fences in agricultural lands (Forman, 1995). By linking two or more forest patches together, corridors are said to provide five key functions: corridors can serve as habitat, as a conduit or a barrier, and as a species source, or as a species sink (Forman, 1995). These functions generally hinge on the extent and length of gaps between forest patches, the external matrix, and the behavioural responses of native biota (Bennett, 2003).

Much controversy has surrounded the idea of corridors as a conservation strategy for fragmented landscapes (Lindenmayer & Fischer, 2006; Bennett, 2003; Van Dyke, 2003; Forman, 1995). Ideally, corridors offer enhanced biotic movement between patches, additional food and habitat resources for species, can promote genetic exchange between subpopulations, and can help to maintain biodiversity at a landscape scale (Lindenmayer & Fischer, 2006; Bennett, 2003; Saunders et al., 1991). The drawbacks of connecting two or more previously isolated forest fragments can include the spread of disease, non-native species or disturbances such a fire, can increase edge habitat, and can link a high quality forest patch to one of lower quality, thereby creating a habitat sink (Lindenmayer & Fischer, 2006; Bennett, 2003; Van Dyke 2003; Saunders et al., 1991). The difficulty in determining whether the benefits of linking forest patches outweigh the disadvantages is that scientific evidence is ambiguous: linking isolated forest patches does not necessarily provide "connectivity" (Bennett, 2003; Van Dyke, 2003). In fact, the conditions needed for a particular individual to actually use a corridor are often species-specific (Van Dyke, 2003). Thus, what some species may perceive as a barrier, others may not. Euglossine bees, for example, were reluctant to travel to forest fragments in the BDFFP that were separated by 80 meters or more from other fragments (Kricher, 1999). Conversely, the same project found that some ant-following birds would cross a clearing of 100 to 320 meters to access primary forest (Laurance et al., 2002).

The ambiguity surrounding the concept of corridors as a conservation strategy suggests that efforts or funds devoted to establishing or protecting corridors over other conservation strategies can be misguided (Lindenmayer & Fischer, 2006). At the same time, a number of studies have demonstrated that landscape patterns that promote some degree of connectivity for species and ecological processes are essential in fragmented tropical landscapes (Sekercioglu et al., 2007; Bennett, 2003; Daily et al., 2003; Laurance et al., 2002; Gilbert & Setz, 2001; Daily & Ehrlich, 1995). Both Lindenmayer and Fischer (2006) and Bennett (2003) argue that any decision to establish linkages between forest patches must consider the following criteria: the ecology of the species targeted for conservation, the kind of landscape changes a corridor is intended to mitigate, the goals of the landscape management plan, and the habitat quality of remnant patches. Bennett (2003) also emphasizes an explicit consideration of local and socio-political factors when designing a connectivity strategy. Considerations should include, but are not limited to, status and tenure of land, management responsibility and adequacy of resources, support from the local community, community education and awareness, and the present and future context of sustainable land management for the region (Bennett, 2003).

2.5 Summary

In considering the ecological and biological effects of habitat fragmentation and loss, it is clear that many tropical species require large tracts of forest with a certain degree of connectivity, and which are insulated from matrix conditions to persist over time. However, given the nature of many human-dominated landscapes, maintaining minimal habitat and resource requirements is not often possible. After reviewing the literature related to this crucial conservation issue, a number of themes can be extracted:

- Large forest patches, ideally greater than 100 ha in size, should form the core of any conservation network

- The majority of edge effects are detrimental for the ecological integrity of remnant forest patches

- Matrix conditions are an important determining factor in the intensity and extent of isolation and edge effects in a landscape

- Smaller forest patches scattered throughout the landscape are important for providing 'stepping-stone' connectivity as well as additional habitat and resources for some species

- Although the ability of corridors to provide connectivity is ambiguous, they may be necessary in some instances to provide direct linkages between patches in order to reduce the effects of habitat isolation

The specific quantity, size, shape, arrangement, and location of these elements essentially creates the spatial pattern of a given landscape. Understanding which spatial elements and in what arrangements are needed to sustain or enhance the ecological integrity in a landscape is fundamental for the conservation of fragmented lands.

Chapter 3. Methods

In recent decades, the application of geographic information systems (GIS) technology in conservation and environmental monitoring disciplines has become an increasingly popular method of extracting information related to landscape composition and configuration (Jensen, 2007). Specifically, the ability to conduct sophisticated inquires to assess and monitor changes in environmental conditions offers an efficient means of researching phenomenon over spatial and temporal scales (Greenberg et al., 2002; Young, 2001). By integrating multiple sets of data into user-defined and interactive maps, GIS software provides a basis from which landscape-scale information can be examined, including patterns, distribution, and various descriptive statistics.

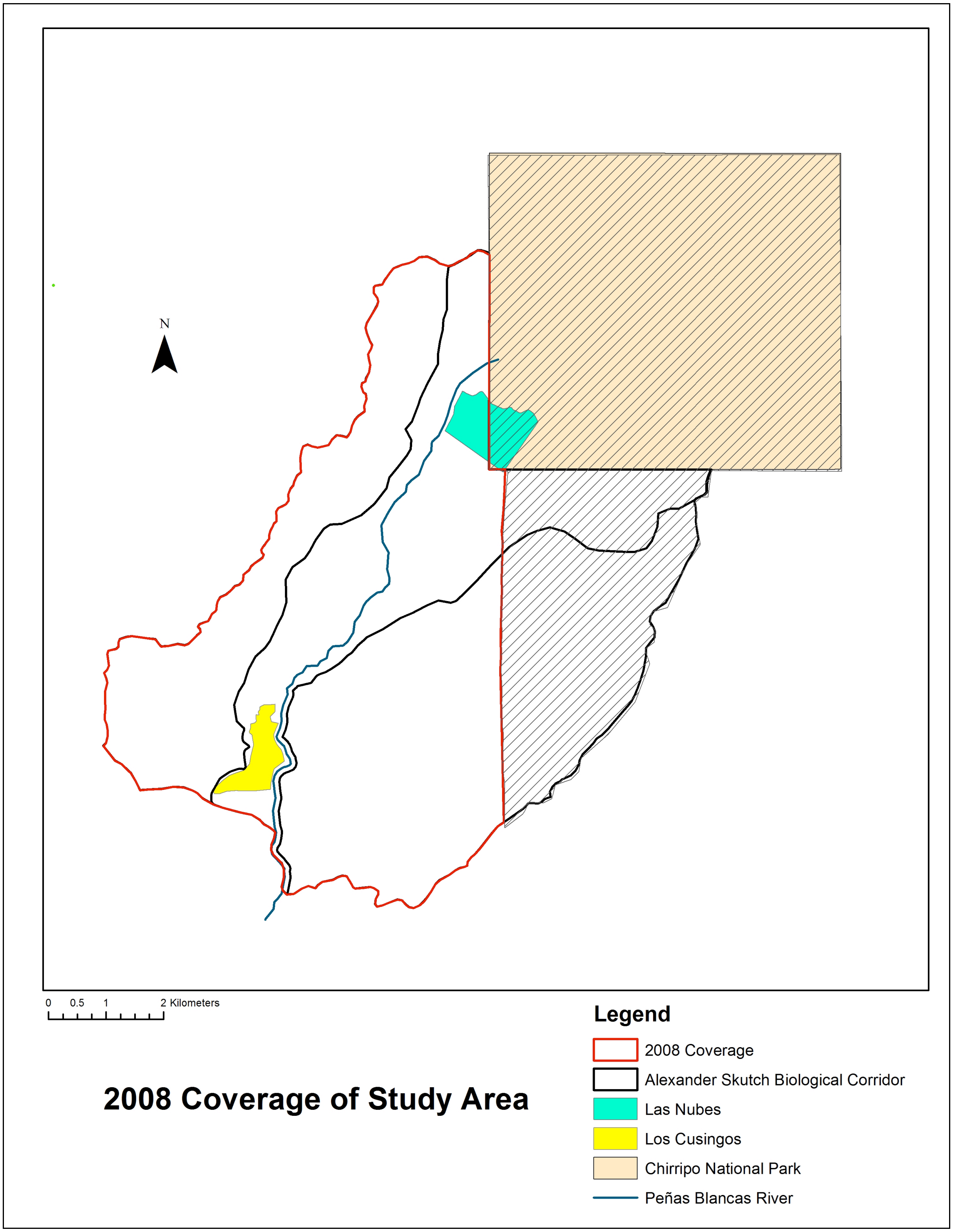
Given that one of the primary research goals of this project is tracking how remnant forest patches have increased, decreased, disappeared, or emerged over time as a result of changing land use practices, GIS technology readily enables this type of analysis. Assessment of forest cover change was measured by comparing landscape data in 1998 to data from 2008 using a ground-truth approach, geospatial data, and qualitative and quantitative measurements. The decision to compare land cover change between 1998 and 2008 in the ASBC was chosen as the coverage period because there is an abundance of GIS data available from the Tropical Science Center (TSC) between 1998 and 2000, and because a ten-year time period reflects the general length of time since the corridor began to form, and as such, can offer insight into the effectiveness thus far of operating goals within the corridor. The following sections describe how information was assembled and analyzed in order to examine forest cover and land use trends in the Alexander Skutch Biological Corridor.

3.1 Data Sources

The Alexander Skutch Biological Corridor is managed by the Tropical Science Center, a non-governmental organization dedicated to the sustainable management of reserve networks and the communities that reside within those networks (www.cct.or.cr). During the late 1990s, the TSC produced a variety of GIS data including land use and forest cover, river and road networks, ecological life zones, and reserve boundaries. Land use and forest cover data were originally obtained from a Landsat Thematic Mapper (TM) 30-meter resolution satellite image from the National Meteorological Institute (IMN) of Costa Rica in 1998 as part of a government initiative to assess carbon sequestration potential for the entire country (Jiménez-Salazar, personal communication; IMN, 2007). The imagery was geometrically corrected by the IMN, and registered to a Lambert Conformal Conic projection, based on the Clarke 1866 ellipsoid (Jiménez-Salazar, personal communication). Land cover data was derived from the satellite image by performing a supervised land use classification. A supervised classification is the process of extracting thematic information from multi-spectral imagery by assigning different land use categories to sample areas in the image based on the specific brightness value of pixels (Lo & Yeung, 2002). A supervised classification approach requires *in situ* knowledge of actual land cover in the field in order to train a remote sensing software program to classify the different spectral signatures in the image (Lo & Yeung, 2002).

Additional data used by the TSC was produced by digitizing topographic maps from the National Geographic Institute of Costa Rica (IGNCR) and by verifying geospatial information through a ground-truth approach (Jiménez-Salazar, personal communication). Ground-truthing is the process of verifying the spatial and attribute accuracy of real-world objects using a Global Positioning System (GPS) device. The generated data was intended to aid in the sustainable management of the corridor by providing information necessary to inform management and conservation decisions (Jiménez-Salazar, personal communication). The TSC, therefore, was the primary source of background data necessary to assess forest fragmentation trends over time.

Data for the year 2008 was obtained by purchasing a panchromatic (grey-scale), high-resolution (0.5 meter) satellite image taken February 12, 2008 from the World View 1 satellite. Only two-thirds of the corridor area was covered in the satellite image, and as such, the present study excludes the eastern and north-eastern sections of the ASBC (Map 3.1). This satellite image was the only digital imagery available that covered the majority of the study area, and that was taken within the past year. All other available imagery (satellite or aerial photo) of the corridor dated back to 2005. Data processing steps for this imagery are described in section 3.3.



Map 3.1: The area of the Alexander Skutch Biological Corridor included in the present study. The hatched lines represent the area not covered by the 2008 satellite image.

3.2 Land Cover Classification Schemes

TSC data relevant to the present study included the land use and forest cover data originally produced by the IMN in 1998 (later verified by the TSC). Table 3.1 outlines the land class categories used by the IMN to identify and map the dominant land uses in 1998.

With regards to classifying the 2008 imagery, observed land use classes in the ASBC were based on the categories used by the IMN and on the classification scheme used by Dean Young, a graduate student who performed a similar land use study in the corridor in 1999 (Young, 2001). Although the 2008 classification scheme is more detailed than the scheme used in 1998, it was more appropriate for describing the predominant land use classes encountered in the corridor. To produce greater consistency between the 1998 and 2008 classification schemes, one only has to combine the coffee, sugar cane, and agriculture (other) classes into the 'Permanent Agriculture' category used by the IMN and TSC. Table 3.2 describes the land use classes for 2008.

Table 3.1: Land cover classes and descriptions for 1998

Land Cover Class	Description
Permanent Agriculture	Agricultural lands devoted to the cultivation of year-round cash and food crops, including coffee, sugar cane, pineapple, bananas, citrus fruits, and agroforestry plantations.
Annual Agriculture	Agricultural lands devoted to the cultivation of crops that are harvested in less than a year, primarily fruits, vegetables, and grains.
Pasture	Areas of land dedicated to the cultivation of grasses for livestock consumption, and occasionally accompanied by shrubs and scattered trees. This category also includes inactive pasturelands.
Cleared Land	Non-urban lands naturally or artificially devoid of vegetative cover. Exposed rock and soil, including volcanic craters, eroded areas, and deposition from rivers and volcanoes fall into this category.
Mixed Use	Small areas of land associated with small farms and/or properties that are characterized by a mix of agricultural uses and grasses in similar proportions.
Early Regeneration	Lands characterized by low-lying, shrubby vegetation, 1 to 5 years after the original vegetative cover has been eliminated by natural disturbances or human activities. Young forests that develop during this stage are typically composed of fast-growing trees with a high demand for light, and can range from 5 to 10 meters in height. This stage preceeds the secondary forest stage.
Secondary Forest	Forested areas that have regenerated following the elimination of the original forest cover, generally caused by natural or human-induced disturbances, including natural disasters. Ecological communities in these forests are more advanced than in the early regeneration stage.
Primary Forest	Areas of land where the original forest still stands, characterized by the presence of trees with diverse heights and widths, and stratified vegetation layers. In Costa Rica there exists a great diversity of forest types due to variability in climate and relief. Flora and fauna exist in equilibrium in primary forest, subject to natural disturbance and successional cycles. When woody vegetation dies, they leave natural gaps and spaces in the forest, in which new vegetation can establish.

Source: IMN (1998). (*Translated from Spanish)*.

Table 3.2: Land cover classes and descriptions for 2008

Land Cover Class	Description
Coffee	Plantations of coffee, either sun-grown or that utilizes both single and multiple species of shade trees. Rows of coffee are typically interplanted with Eucalyptus, Poro, and Inga trees, though tree groves and hedgerows are not uncommon in a plantation.
Sugar cane	Monoculture plantations of sugar cane, ranging in height from 1 meter to over 3 meters. Prior to harvest, low-burning fires are set in cane fields to clear underlying vegetation, after which the ground may remain bare until the next crop sprouts.
Agriculture (Other)	Lands devoted to agricultural uses that do not fall under the previous categories. This includes agroforestry plantations, livestock farming other than cattle, and unharvested pineapple.
Pasture	Areas of land typically consisting of low-lying grasses and shrubs, either partially or completely cleared of woody vegetation. Scattered trees, tree groves, riparian areas, and hedgerows may also be present.
Cleared Land	Lands that lack vegetative cover, such as exposed rock, or more commonly, land cleared for housing construction or agricultural purposes. The majority of cleared lands encountered in the corridor were intended for cultivating pineapple or sugar cane.
Mixed Use	Areas made up of housing and/or farms that may contain several different land uses within a small area. This includes lawns and gardens typically associated with homes, and small agricultural or grassland areas. Forest patches and riparian corridors of under 5 ha that were present outside of larger forest patches are also included in this category.
Forest	Assemblages of mostly woody vegetation, characterized by the presence of trees over 5 meters, hanging vines, shrubby undergrowth, and a leaf litter layer (Wyser, 2003). Tree density and canopy cover generally vary with forest stand age, so two sub-categories of forest were recorded: • Primary/Mature: dense stands of mature, slower-growing trees estimated to have a canopy cover of greater than 80% (Guariguata et al., 1997). Primary forests are relatively undisturbed forest stands, having little or no previous impact from humans, whereas mature forest stands may have suffered from human-inflicted disturbance in the past but have since regenerated and are estimated to be at least 50 years old. • Secondary: Forest stands containing a lower density of tall, mature trees, characterized by young trees and dense under story growth (Guariguata et al., 1997; Kapelle et al., 1996). This category includes regenerating lands made up of pioneer species of trees, trees of diverse sizes, and a high density of shrubs and/or tall grasses. Secondary forest stands are less than 50 years old. Forest patches were grouped into these categories by consulting forest cover maps from the TSC, by talking with local residents who had lived in the corridor for more than 50 years, and by making observations related to tree height, basal thickness, undergrowth density, and the presence of hanging vines while in the field.

Source: Young (2001).

3.3 Field Approach

3.3.1 Ground-truthing

A ground-truth approach was applied in the field from January to April 2008 in order to generate up-to-date qualitative and quantitative data of land uses and forest cover in the corridor. Initially, fieldwork to obtain 2008 data was to be carried out using only a GPS device and topographic maps to map the boundaries of existing forest patches of over 5 hectares (ha) and the boundaries of land uses immediately surrounding these forest patches. Although the Food and Agriculture Organization (FAO) of the United Nations (UN) considers forest to be any forested piece of land at least 0.5 ha in size, a study conducted by a fellow student on avian diversity and habitat use in the corridor found that the habitat value of forest patches below 11 ha decreased significantly for bird species (FAO, 2000; Carter, 2007). Thus, a minimum mapping size of 5 ha for forest patches was selected for the present study for feasibility purposes, and to ensure that a sufficient level of forest patch data would be mapped to enable a landscape-scale analysis. In other words, 5 ha was deemed an appropriate size threshold given the time constraints of the field work period.

However, given the difficulty of the terrain and the amount of time it took to physically walk the boundaries of forest patches to record both coordinate and qualitative data, this method was decidedly inefficient for the desired scale and detail of mapping. As a result, a satellite image was purchased to aid in mapping the corridor (described previously). The decision to incorporate remotely-sensed imagery in the present study changed the field work approach significantly:

instead of mapping the physical boundary of patches with a GPS device, ground control points (GCPs) and the locations of clear examples of land use classes were recorded with the GPS device.

GCPs are easily identifiable reference locations, such as major road intersections or rivers, common to both the field and the satellite image (Lo & Yeung, 2002). GCPs are collected for the purposes of assessing the accuracy of satellite-derived data, and for referencing the image to a known geographic coordinate system (Lo & Yeung, 2002; Young, 2001). A total of 74 control points were collected from the study area. Although efforts were made to evenly distribute GCPs across the corridor, due to inaccessible roads or private property restrictions, there were significantly fewer control points collected from the north-east section of the corridor. Because this corresponds with the area of the corridor not covered in the satellite image, this did not compromise accuracy assessments or georeferencing abilities.

The classification of a panchromatic image requires that clear examples of land use types be identified and recorded in the field. Unlike the multi-spectral imagery typically used in a supervised classification (which consists of multiple colour bands that combine to create a colour-composite image), a panchromatic image only consists of one spectral band, which limits the classification procedure to visual interpretation using on-screen digitization in a GIS software program (Jensen, 2007). As such, even though land use class examples were recorded in the field, it was not for the purposes of 'training' the satellite image. Instead, ground-truth efforts were primarily directed at recording the locations and qualitative attributes of forest patches larger than 5 ha and the land use types surrounding these patches to guide the on-screen classification process.

A minimum of 25 land class points for the pasture, sugar cane, cleared land, and mixed use categories were collected, whereas for the forest category, a minimum of 25 points were collected for each sub-category. Upwards of 75 points were collected for the coffee category because varying levels of shade cover in a plantation can make it difficult to distinguish from forested areas in a satellite image (Young, 2001). Collecting a wide variety of coffee plantation types and locations aids in making classification decisions during on-screen digitization (Young, 2001). Less than a dozen field observations were made for the agriculture (other) category due to class rarity. Coordinate locations and qualitative data were collected from areas both immediately outside forest patches and from sample areas within the corridor because it was not always possible to visit every forest patch over 5 ha and surrounding land uses. Furthermore, since the satellite image was not obtained until nearly the end of the field work period, it was not possible to re-visit areas in the corridor that may have needed verification. Thus, 'sample areas' were collected for the purposes of making educated observations and classification decisions regarding land uses in the satellite image when direct field observations had not been recorded.

The GPS device used in the field was a Garmin eTrex Vista personal mapping unit. This device is capable of measuring elevation, recording routes and paths with an accuracy of up to 3 meters, contains an electronic compass, and can store attribute data in addition to location coordinates. GPS data was collected in a latitude/longitude format, using the WGS datum of 1984. Qualitative data pertaining to each land use patch was also recorded, including general location, compass bearing, land use type, vegetation data, and site sketches (see Appendix A). Recording qualitative data in conjunction with a ground-truth approach aids in corroborating locational accuracies recorded by the GPS device, and provides key information for classifying land uses later on.

3.3.2 Interviews

In addition to conducting field mapping, structured interviews were conducted with residents living within the boundaries of the Alexander Skutch Biological Corridor. The purpose of these interviews was:

1) To survey the predominant types of land uses occurring on individual properties, and the residents' motivations for choosing these land use types.
2) To identify land uses changes that have occurred within the corridor in the past decade, and to gain understanding of the drivers of these changes.
3) To inquire about the locations of recently re-forested or deforested areas within the corridor for the purposes of guiding ground-truth visits.

A minimum of 25 potential participants were selected based on a minimum length of residency within the corridor of ten years, and/or a minimum property size of 3 hectares. These criteria were chosen to ensure that a participant had resided in the corridor for a long enough time to be able to witness land use changes, and who could also provide insight into their own motivations for utilizing specific land uses on their properties.

With the aid of a local resident, informed consent was gained orally through informal introductions with potential participants, and a full explanation of the research project and the participants' rights were provided prior to the start of each interview. Questionnaires consisted of approximately 15 questions, and individual interviews ranged in time from 20 minutes to over an hour (see Appendix B for a copy of the questionnaire). Interviews were conducted primarily in Spanish and then later translated to English, with assistance. Data generated from the interviews was sorted, categorized, and computed in a Microsoft Excel program.

3.4 Data Processing

3.4.1 Field Data

During the field work period, collected GPS data was uploaded into a Map Source software program, which comes with any Garmin GPS package. This program enables the user to upload, store, and display coordinate, route, and attribute data on a computer. Map Source software was used to edit and arrange field data for the purposes of uploading data into a GIS program, in this case, ESRI ArcGIS 9.2 software. All GPS point data was converted from a latitude/longitude format to decimal degrees, and the default coordinate system converted from the WGS datum of 1984 to a custom Universal Transverse Mercator (UTM) grid for Costa Rica, also based on the WGS datum. These data conversions are necessary for the proper display of GPS-derived data in a GIS software program (ESRI, 2007a). Additional qualitative information associated with GPS data was stored in an Excel format, which was later appended to the GIS shapefiles. GIS point files were created for each of the land use class categories and for the GCPs, and later displayed in ArcGIS software. The purpose of this process was to prepare field data to be used as a guide to define and map corresponding land uses in the satellite image, as well as to perform an accuracy assessment based on GCPs and their referenced locations in the image.

3.4.2 Remotely-sensed Data

With any remotely sensed digital imagery, pre-processing is necessary to correct errors that may have been introduced by the remote sensing system and atmospheric conditions (Lo & Yeung, 2002). Sensor design, movement of the sensor platforms, albedo, the presence of clouds, and relief displacement are just a few factors that may serve to introduce errors and/or distortion in a satellite image (Lo & Yeung, 2002; Jensen, 2007). The processes of orthorectification and georeferencing are often employed to remove these errors. Orthorectification is the process of removing distortion caused by relief displacement in a given landscape by applying a digital elevation model (DEM) to correct height data (Lo & Yeung, 2002). In contrast, georeferencing is the process of assigning a known coordinate system to the X,Y coordinates present in an image through the use of GCPs (Lo & Yeung, 2002; Jensen, 1996). Both processes aim to produce an image that most accurately reflects real-world conditions, both geometrically and pictorially (Lo & Yeung, 2002).

Although the World View 1 satellite image was purchased already orthorectified and georeferenced, there was some inconsistency between the collected GCPs and the corresponding locations in the image. Namely, corresponding locations were separated by up to 150 m between the two sets of data so it was necessary to resample the image in accordance with the locations of the GCPs collected in the field. The apparent misalignment between the satellite image and the GCPs was likely the result of the two different coordinate systems used to define each data set. As mentioned previously, the original satellite image was defined using to the WGS 84 datum, however it lacked the custom UTM projection for Costa Rica. Despite the fact that the image was re-projected in accordance with the CRTM/WGS 84 parameters, the misalignment still persisted and further georeferencing was needed.

In assessing acceptable standards for georeferencing, Lo and Yeung (2002) suggest that a minimum of 20 control points be distributed throughout the map area, and that the square root of the average error of spatial discrepancies (root mean square error, or RMSE) between sample points and the control points be less than the average mapping error of the original control points. Of the 74 GCPs collected in the field, a total of 37 control points were assigned to the 2008 satellite image, using the “Georeference” toolbar in ArcGIS 9.2. A RSME of 6.77 meters was deemed an acceptable error range for the process, given that the average accuracy of GPS points collected in the field was 6.8 meters. A cubicconvolution re-sampling method was employed to rectify the image to match the corresponding control points. Following these procedures, the image was ready to be converted into vector (point, line, polygon) format via on-screen digitization.

3.4.3 Creation of 2008 Forest Cover and Land Use Data

"Heads-up" or on-screen digitizing is the process of creating vector data from digital raster imagery by electronically tracing the geographic boundaries of patches and/or objects within a particular landscape (ESRI, 2007b). This process can be readily accomplished using the "Editor" and "Advanced Editor" toolbars in ArcGIS 9.2 software.

GPS-derived point files of each land use class were used to guide the digitization process by providing either known locations of forest patches and other land uses, or by providing known locations of clear land use examples which could be used to visually interpret land use types that were not ground-truthed while in the field. Boundaries of forest patches and other land uses were digitized at an average scale of 1:1700 to ensure sufficient mapping detail to distinguish between land uses. Associated attribute data such as land use type, patch size in hectares, and other notes were later appended to each vector file produced.

All forest patches greater than 5 ha were digitized, including any riparian corridors or forested 'fingers' extending from patches. Land uses immediately surrounding each digitized forest patch were digitized according to the designated classes. For feasibility purposes, only land uses immediately outside forest patches were digitized, as the process of on-screen digitizing can be tedious and time-consuming. In total, seven land class files were created: forest, pasture, coffee, sugar cane, cleared land, mixed use, and agriculture (other). All produced vector files were defined according to the CRTM/WGS 84 coordinate system, and were clipped to fit the boundaries of the study area using the “clip” tool in the ArcGIS toolbox.

3.4.4 TSC 1998 Data

Because GIS data from the TSC was already in a vector format, very little pre-processing was necessary to prepare this data set for results processing. Existing data files relevant to the study, such as road and river networks, corridor boundaries, forest cover, and land use cover files, were converted from a Lambert projection/Clarke 1866 datum to the CRTM/WGS 84 coordinate system using the “project” tool provided in the ArcGIS toolbox. A geographic transformation using Molodensky parameters was also used to convert between the Clarke 1866 datum and the WGS 84 datum.

In order to achieve a sufficient level of consistency between the 1998 forest cover data and the 2008 forest cover data, all forest patches smaller than 5 ha were deleted from the 1998 forest cover file. Additionally, all relevant vector files were clipped to fit the area of the corridor that was covered in the 2008 satellite image using the “clip” tool in ArcGIS toolbox.

3.5 Accuracy Assessment

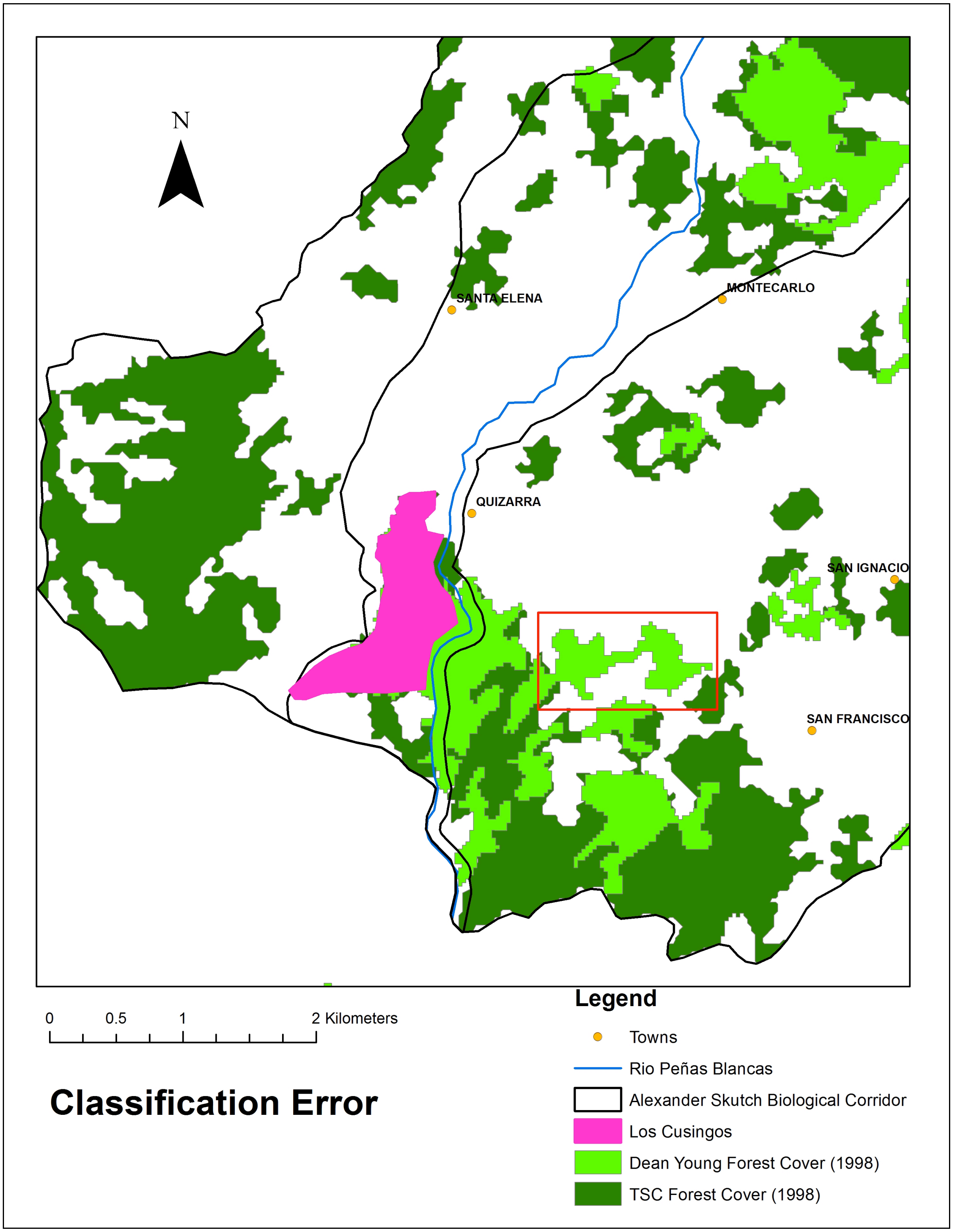
Given the nature of the present study, one of the main concerns for maintaining acceptable levels of accuracy is the level of consistency across data sources. Variability between data sources includes, but is not limited to, the type of remotely sensed imagery and classification procedure used, geometric correction techniques, spatial resolution, and field verification methods.

As already stated, the forest cover and land use data from 1998 and 2008 were obtained and processed in different manners, due to the nature of the original satellite image used. Although the outcome of both procedures is intended to be the same (i.e. a land use map for the designated year), a number of factors may present uncertainties in the accuracy of these maps.

First, the original satellite images were taken using different spatial resolutions: 30 meters for the Landsat TM image, and 0.5 meters for the World View 1 image. Clearly, a coarser spatial resolution can increase the chances of incorrectly classifying certain types of land uses that have similar spectral signatures in the image. For example, Dean Young (2001), in his 1999 study of the relationship between land uses and water quality in the ASBC found that an important source of error in performing a supervised classification of Landsat TM imagery was the inclusion of forested areas in the shade-coffee class. Young concluded that areas of forest were likely to be under-estimated as a result, while shade-coffee areas over-estimated within the corridor. Evidence of this classification error is illustrated in Map 3.2, where Young's forest data (of the

Peñas Blancas watershed only) is overlaid with the forest data produced by the TSC.

Although the satellite images used to generate these layers were both obtained in 1998, and may even be from the same image, it is clear that several relatively large tracts of forest present in Young's forest data are absent from the TSC's forest data. When questioned on this matter, both Young and the GIS technician at the TSC assured me that their own data sets had been verified in the field using a GPS device in the late 1990s (Young, personal communication; Jiménez-Salazar, personal communication). Upon further examination of published datasets, Young's land use data was found to have an overall accuracy of 72%, whereas the land use data produced by the IMN and TSC had an accuracy of 89% (Arroyo-Mora et al., 2005). With regards to the present study, this uncertainty, although unresolved, does not pose a significant impediment to producing data results because overall forest cover and land use data from 1998, as opposed to a patch-by-patch analysis, is used to present a frame of reference for characterizing forest fragmentation trends over the past decade. That being said, data results and recommendations for areas that should be prioritized for ecological restoration are based on the 2008 forest cover and land use maps produced with a much higher resolution, thus greatly reducing the risk of misclassifying forest areas and shade-coffee plantations.



Map 3.2: A potential classification error between two sets of satellite-derived data from 1998. The highlighted area in red shows two forest patches, classified as "primary" forest from Young's (2001) study that are not present in the forest cover from the TSC, also from the same year.

Second, the ability to map more details with a higher resolution image has enabled a more ‘fine-scale’ 2008 forest cover and land use map to be produced. That is, because progressively finer details can be readily defined and classified in the 0.5-meter resolution 2008 imagery, this map has an inevitably ‘patchier’ appearance (see Maps 4.5 and 4.6 in *Results and Discussion*). Although this does not necessarily have a significant bearing on the data results, it does mean that smaller patches, likely less than 1 ha, in the 1998 data were generalized and incorporated into a more dominate proximate land use. The 30-meter resolution of the 1998 image may also mean that narrow riparian corridors and forest ‘fingers’ were excluded from the forest layer, even though a linear forest patch may have amounted to greater than 5 ha in size.

Given both of the above accuracy issues, I argue that the full extent of forest cover in the 1998 data is probably under-estimated and, as such, comparisons between changes in forest cover and land uses between 1998 and 2008 may also be somewhat under-estimated.

Finally, despite the 0.5-meter resolution of the 2008 satellite image, a few classification discrepancies were encountered. Because the image was only available in a grey-scale colour scheme, the classification of different land uses in the corridor was primarily made using referenced GPS locations from the field and by conducting a textural comparison between land uses if any uncertainty arose during the digitalization process. Referenced field locations served as benchmarks for characterizing the tone, grey-scale pixel value, and texture of various land uses.

For the most part, the majority of patches within the corridor were classified with little or no uncertainty. Where uncertainty arose, this pertained mainly to smaller land uses, such as those under 0.5 ha in size, and when patches with similar tone and texture were encountered. For example, pasturelands and sugar cane fields with low vegetation cover appear very similar in the image. As Figure 3.1 illustrates, both patches appear to have a relatively smooth texture and a lighter-grey colouring. Careful scrutiny of ground-truthed pasture fields absent of woody vegetation and sugar cane fields with new re-growth revealed that sugar cane fields are more likely to display subtle ‘till’ marks from where farmers built up rows or mounds to plant the cane. In addition, the majority of pasturelands encountered outside of forest patches had scattered trees and hedgerows planted in them, whereas sugar cane fields did not. Another point used to distinguish between pasture and sugar cane was elevation. There is a distinct trend for sugar cane to be present in the lower elevation areas of the corridor and for pasture lands to be more concentrated in the higher elevations (see Map 4.6 in *Results and Discussion*). This is likely due to the fact the sugar can grows best with maximum sunshine levels, which are markedly reduced as one goes higher in elevation due to the increased presence of clouds over the Talamanca Mountain range.

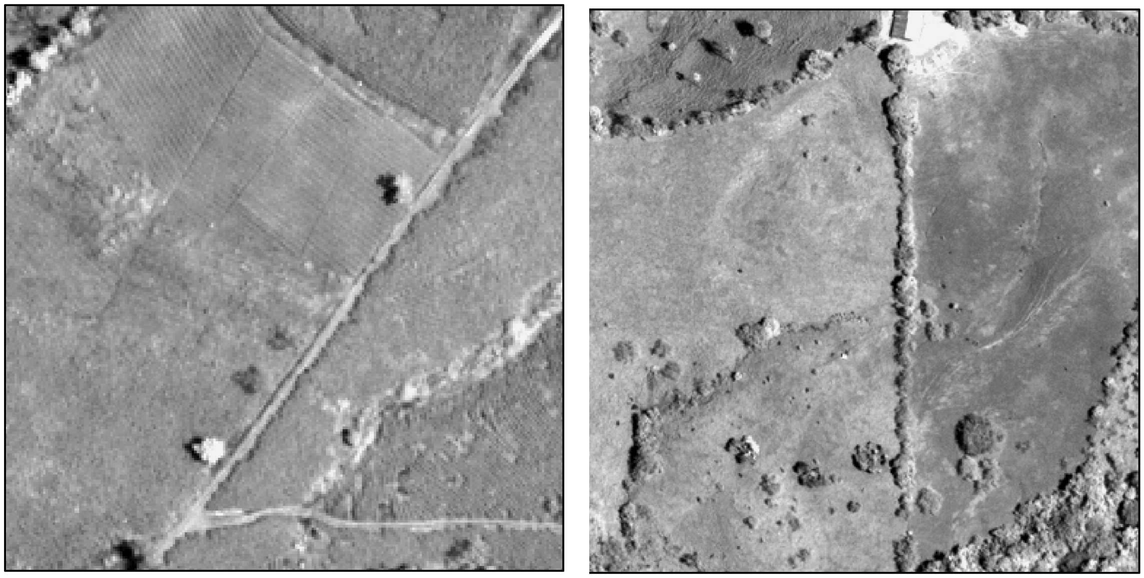


Figure 3.1: A visual comparison between sugar cane (left) and pasture (right). Similarities between tone and texture can make it difficult to differentiate between these two land uses in a panchromatic image. Sugar cane is more likely to display till marks and a 'fuzzier' texture, while pasture is often accompanied by hedgerows and scattered trees, as is evident in the right panel.

3.6 Generation of Results

3.6.1 Forest Cover Change Between 1998 and 2008

Landscape-based studies of the relationship between habitat loss and ecological integrity or biological diversity are often concerned with quantifying the spatial composition and configuration of a landscape through various descriptive statistics, including fragment size and proximity, core and edge habitat measurements, species counts, and matrix contrast indices (Develey & Metzger, 2006; Laurance et al., 2002; Guidon, 1996; Forman, 1995). These types of statistics are designed to provide insight into the environmental conditions of a landscape and, ideally, implications for conservation and/or environmental management objectives (Laurance & Peres, 2006; Schelhas & Greenberg, 1996). Accordingly, a similar approach was taken for the analysis of forest cover in 1998 and in 2008.

In order to characterize changes in forest cover in the ASBC it was necessary to compute statistics related to the extent of forest cover, average forest patch size, distance between forest patches, and core and edge area measurements. FRAGSTATS 3.3 software was used to generate this information (MacGarigal & Marks, 1995). FRAGSTATS is a free, downloadable program produced by Oregon State University in the mid-1990s, intended to provide students, researchers, organizations, and others with the capability of computing landscape metrics from categorical map patterns, specifically maps produced with GIS software (MacGarigal & Marks, 1995). Spatial patterns and configurations computed through FRAGSTATS are user-defined and can be used to tabulate any landscape-scale phenomenon on a patch, class, or landscape level (MacGarigal & Marks, 1995). The decision to use any of these scales is based solely on the type of data and results desired by the user.

Maps of forest cover in 1998 and 2008 of the ASBC were exported into an ArcGrid format (compatible with FRAGSTATS software), and patch and class level statistics were computed for each analysis year. Tables 3.3 and 3.4 list the FRAGSTATS parameters and accompanying equations used to generate data results (MacGarigal & Marks, 1995).

Table 3.3: Patch-level Metrics

Metric	Code	Equation	Description
Patch Area	AREA	AREA= area of patch (m^2) / 10,000 (to convert to hectares)	The size of each forest patch, in hectares.
Shape Index	SHAPE	SHAPE= 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 (in a square raster format) of the corresponding patch area.	SHAPE equals 1 when a patch is maximally compact (i.e., square or almost square) and increases without limit as patch shape becomes more irregular. Another measure of shape complexity.
Core Area	CORE	CORE= the area (m^2) within the patch that is further than the specified depth-of-edge distance from the patch perimeter, divided by 10,000 (to convert to hectares).	The area within a patch that is beyond the user-defined distance from patch edge. In this case, an edge distance of 100 m was selected. [1]
Core Area Index	CAI	CAI= the patch core area (m^2) divided by total patch area (m^2), multiplied by 100 (to convert to a percentage).	CAI represents the percentage of the patch that is comprised of core area.
Euclidean Nearest-Neighbour Distance	ENN	ENN= the distance (m) to the nearest neighbouring patch of the same type, based on shortest edge-to-edge distance.	Measures the distance, in meters, between patches.

[1] The Biological Dynamics of Forest Fragments Project (BDFFP) in Amazonia found that the majority of edge effects occur within the first 100 m of a forest patch, including higher light levels, lower moisture levels, and reduced understory-bird abundance (Laurance et al., 2002). (See also Figure 2.1 in *Literature Review*).

Table 3.4: Class-level Metrics

Metric	Code	Equation	Description
Class Area	CA	CA= the sum of the areas (m^2) of all patches of the corresponding patch type, divided by 10,000 (to convert to hectares)	Measures total class area of the input class type.
Number of Patches	NP	None	The number of individual classes found in the input class type.
Total Core Area	TCA	TCA= TCA equals the sum of the core areas of each patch (m^2) of the corresponding patch type, divided by 10,000 (to convert to hectares)	The amount, in hectares, of total core areas across the input class type, as designated by the user-defined distance from edge (100 m).
Core Area Percentage of Landscape	CPLAND	CPLAND= the sum of the core areas of each patch (m^2) of the corresponding patch type, divided by total landscape area (m^2), multiplied by 100 (to convert to a percentage)	The percentage of the landscape of the input class type that is comprised of core area.
Clumpiness Index	CLUMPY	CLUMPY= the proportional deviation of like adjacencies involving the corresponding class from that expected under a spatially random distribution	Measures the degree of aggregation or disaggregation of patches within the input class type. When CLUMPY equals -1, the patches are maximally disaggregated; CLUMPY equals 0 when patches are randomly distributed; and CLUMPY equals 1 when patches are maximally aggregated.
Connectance Index	CONNECT	CONNECT= equals the number of joinings between all patches of the corresponding patch type divided by the total number of possible joinings between all patches of the corresponding patch type, multiplied by 100 to convert to a percentage	The percentage of patches in the input class type that are either joined or within the user-defined distance from a patch edge. In this case, connectivity was defined as being within 100 m of a patch edge. [2]
Distribution Statistics	MN, MD, RA, and SD	Standard summary equations for mean, median, range, and standard deviation.	Distribution statistics were computed for AREA, PARA, CORE, CAI, and ENN.

[2] One of the key findings in the 29-year BDFFP was that even small clearings of less than 100 m between forest patches act as dispersal barriers for some tropical forest species (Laurance et al., 2002). Similarly, in their study on the effects of forest fragmentation patterns and bird diversity in Brazil, Develey and Metzger (2006) used a 100 m connectance threshold for processing their GIS files through FRAGSTATS.

3.6.2 Land Uses Outside Forest Patches

The inconsistencies between the classification of land uses in 1998 and 2008, combined with varying spatial resolutions, presented a significant challenge in characterizing land use changes over time in the corridor. As a result, mapping and analyzing data related to land uses outside of forest patches in the corridor was restricted to 2008 data only. The intention of this analysis was to:

1) Gain a visual understanding of the predominant types of land uses present in the ASBC as of 2008;
2) Evaluate land uses outside of forest patches greater than 5 ha in terms of their ability to provide landscape connectivity (i.e. which types of land uses are more likely to allow species to disperse between forest patches?)

It must be noted that the idea of 'landscape connectivity' does not attempt to view connectivity from a species and/or ecological processes perspective (Lindenmayer & Fischer, 2006). That is, it does not attempt to generate guidelines about a species or taxas' ability to move from fragment to fragment based on dispersal capabilities, behaviour, and habitat needs (Lindenmayer & Fischer, 2006). Rather, landscape connectivity focuses on the degree of connectivity provided by the matrix based on the similarity of the vegetation structure present in the matrix compared to the vegetative structure of a forest patch (Lindenmayer & Fischer, 2006; Gascon et al., 1999). In general, the more similarly land uses in the matrix reflect that of a forest patch, the more likely species will be able to travel between habitat patches (Lindenmayer & Fischer, 2006; Gascon et al., 1999).

Although assessing the ability of taxa to move between forest patches is beyond the scope of this project, evaluating which matrix areas of the ASBC are most likely to enable species dispersal is an important aspect of measuring habitat isolation in the corridor. The predominant land uses observed in the ASBC clearly vary in the level of vegetative structure they provide. As such, a weighted analysis was used to assign a numeric value between 1 and 5 (1= most conducive to providing connectivity; 5= least conducive to providing connectivity), using ArcGIS (Table 3.5). ArcGIS was used to produce a colour-coded map of land uses in 2008 displaying the weighted values (see Map 4.7 in *Results and Discussion*).

Table 3.5: Landscape Connectivity Values

Connectivity Value	Description
1	Coffee plantations, agroforestry plantations, tree groves and riparian corridors outside of forest patches larger than 5 ha
2	Pasturelands with trees and/or shrubs
3	Homes, small buildings, and associated lawns or gardens ("Mixed Use")
4	Sugar cane plantations
5	Cleared lands for agricultural or construction purposes

Chapter 4. Results and Discussion

i. Results

GIS technology in conjunction with remotely sensed imagery and FRAGSTATS software was used to illustrate how the spatial characteristics of forest cover have changed between 1998 and 2008 in the area now designated as the Alexander Skutch Biological Corridor. The purpose of characterizing these changes was to provide insight into the ecological conditions of the corridor, based on the principles and concepts of landscape ecology and conservation biology. Forest cover and land use data were used to assess the degree of forest fragmentation and isolation over time in the corridor by computing a variety of statistics related to forest patch size, overall extent, shape, core-to-edge ratio, and connectivity indices (*Objectives I & II)*. The data generated from structured interviews conducted with residents in the ASBC contributed to understanding the socio-economic drivers and motivations of land cover changes between 1998 and 2008 (*Objective IV)*. Overall, this information can be used by the TSC, community groups within the corridor, student researchers, or other responsible authorities to make informed decisions about the future environmental management of the ASBC, including opportunities for strategically placed ecological restoration efforts, which are outlined in Chapter 5 (*Objective III*).

4.1 Forest Cover Change, 1998 - 2008

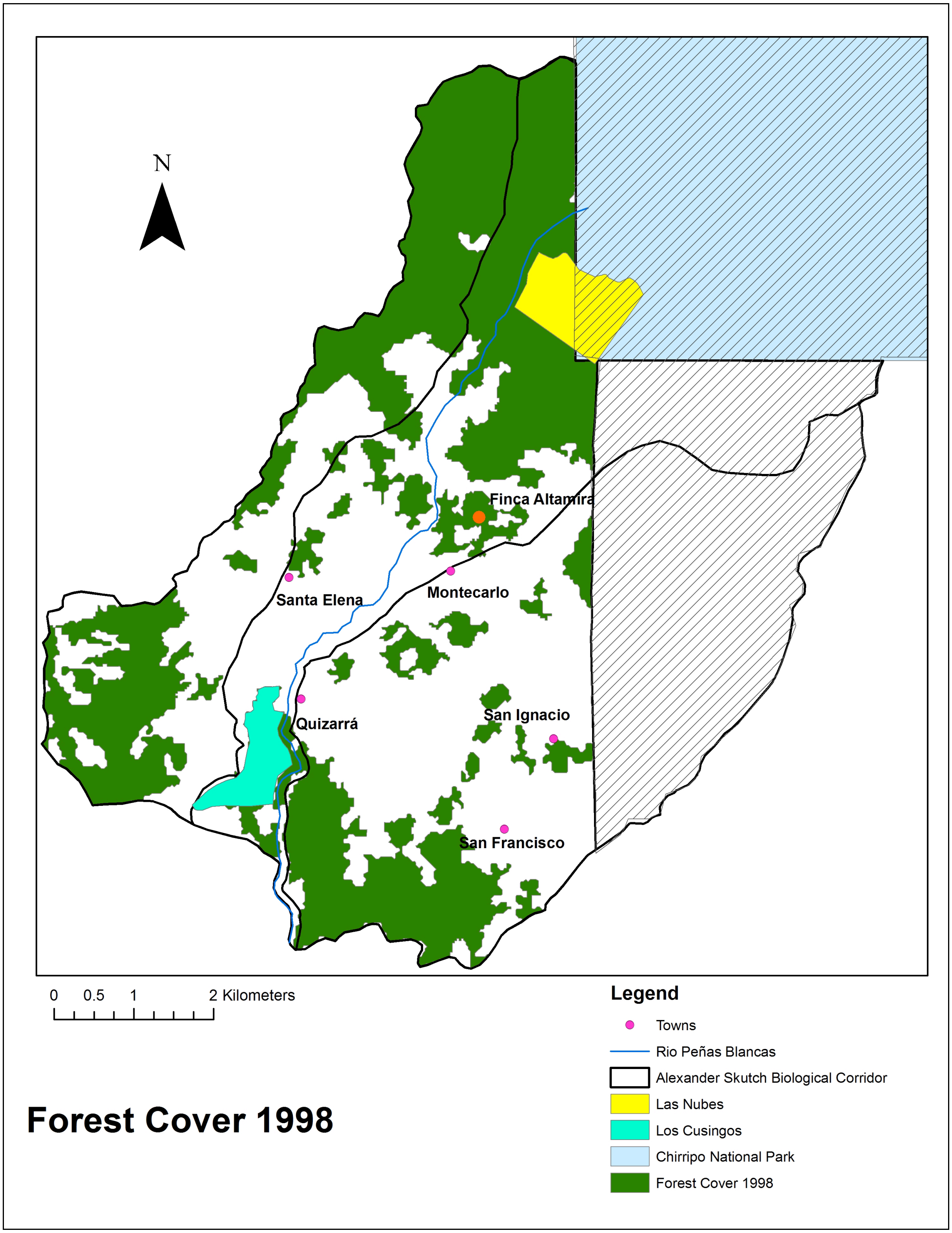
4.1.1 Forest Extent and Patch Size

Within the two-thirds of the ASBC that was included for analysis in the present study, the number of hectares covered by forest in 1998 was 2,119 ha, or roughly 46.7% of the study area (Map 4.1). A forest cover of 46.7% in 1998 is similar to the findings from a national forest inventory for Costa Rica in the year 2000 (46.3%) (Kleinn et al., 2005). In contrast, the 2008 satellite imagery revealed that the number of hectares covered by forest in the ASBC dropped to 1,716 ha, or roughly 37.8% of the study area (Map 4.2). As such, over a ten-year period, the amount of forest in the ASBC decreased by an alarming 19%; an average rate of 1.9% per year. (See Appendix C for the results from FRAGSTATS computations).

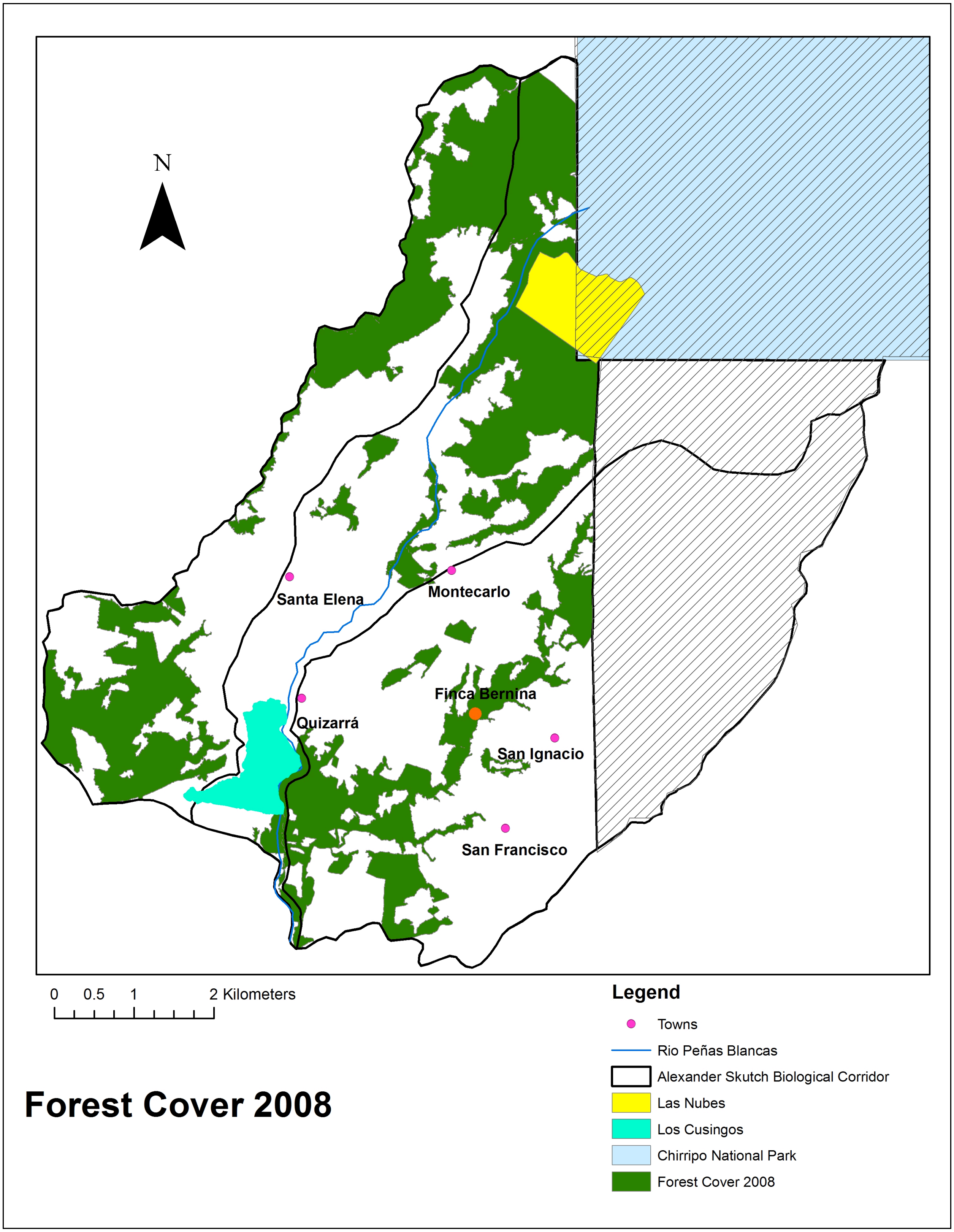
Although the number of forest patches over 5 stayed relatively the same between 1998 and 2008 (23 versus 22 patches respectively), the average size of forest patches decreased from 92.14 ha in 1998 to 78.01 ha in 2008. This represents an average decrease of 14.13 ha, or 15%, per forest patch since 1998, suggesting that external land uses have steadily encroached upon remaining forest patches in the corridor.

In both 1998 and 2008, the location of the largest forest patches remained relatively unchanged, being located either in the vicinity of Las Nubes Reserve in the north or in the vicinity of Los Cusingos Bird Sanctuary in the south (Maps 4.1 and 4.2). When added together, the top five largest forest patches in the corridor in 1998 amounted to an area of 1,868.66 ha, or 88.18% of forest area for that year. Similarly, in 2008, the top five largest forest patches amounted to 1,471.51 ha, or 85.74% of all forest area for that year.

Taken together, these numbers show that less than 12% of forest patches in 1998 and less than 15% of forest patches in 2008 were distributed in the area between Las Nubes and Los Cusingos. In fact, the largest forest patch in this area in 1998 was found to be Finca Altamira, an area of 49.17 ha to the south of Las Nubes (Map 4.1), whereas the largest forest patch in this area in 2008 was found to be a 62.41 ha regenerating forest area on Finca Bernina, east of the Peñas Blancas River (Map 4.2). The fact that forest patches for both analysis years were not distributed evenly is reflected in the Clumpiness index (CLUMPY): in 1998 forest patches had clumpiness index of –1.00, which means they were maximally disaggregated, whereas in 2008 the clumpiness index was measure to be –0.16 for forest patches, suggesting that the current distribution of forest patches in the corridor is somewhat more aggregated (MacGarigal & Marks, 1995). A quick look at Maps 4.1 and 4.2 reveals that forest patches are clearly concentrated in the north and south ends of the corridor.



Map 4.1: Forest cover for the year 1998 for the ASBC. The light-grey hatched lines indicate the area of the corridor not included in the present study.



Map 4.2: Forest cover for the year 2008 in the ASBC. From here on, the light-grey hatched lines indicate the area of the corridor not included in the present study.

4.1.2 Edge and Core Areas

The amount of edge-to-interior habitat in a given patch is partially a function of patch shape (Sodhi et al., 2007; Forman, 1995; Saunders et al., 1991). Large patches that have a compact shape (i.e. a square or circular shape) are believed to support more interior-forest conditions than elongated or convoluted patches (Forman, 1995). While the average patch shape in 1998 was 2.13, in 2008 that index increased to 2.70 for forest patches. The value of the Shape Index equals one when a patch has a compact shape, and increases as the patch shape becomes more irregular (MacGarigal & Marks, 1995). These numbers show that forest patches have become more irregular in shape over time, which indicates that the amount of core habitat within patches has decreased over time as well.

This indication is confirmed upon examination of the Core Area (CORE) and Core Area Index (CAI) metrics. Total core area in the study area decreased from 1,170.03 ha (55% of total forest area) in 1998 to 704.12 ha (40% of total forest area) in 2008. This reflects a loss of 15.2% of core habitat area over a ten-year period, under the assumption of a 100-meter core-to-edge distance from patch boundaries. The average amount of core habitat per forest patch in 1998 was 50.87 ha, whereas in 2008 this number dropped to 32.01 ha per patch. Furthermore, in 1998 the average proportion of a forest patch that was considered “core” area was 9.85%. In 2008, this number decreased to 6.64%, nearly a 33% drop in the proportion of core area per forest patch. Not surprisingly, the forest patches with the greatest proportion of core area corresponded to the largest forest patches in the corridor (see Appendix C). As such, it appears that larger patches are more likely to contain some core area than smaller patches.

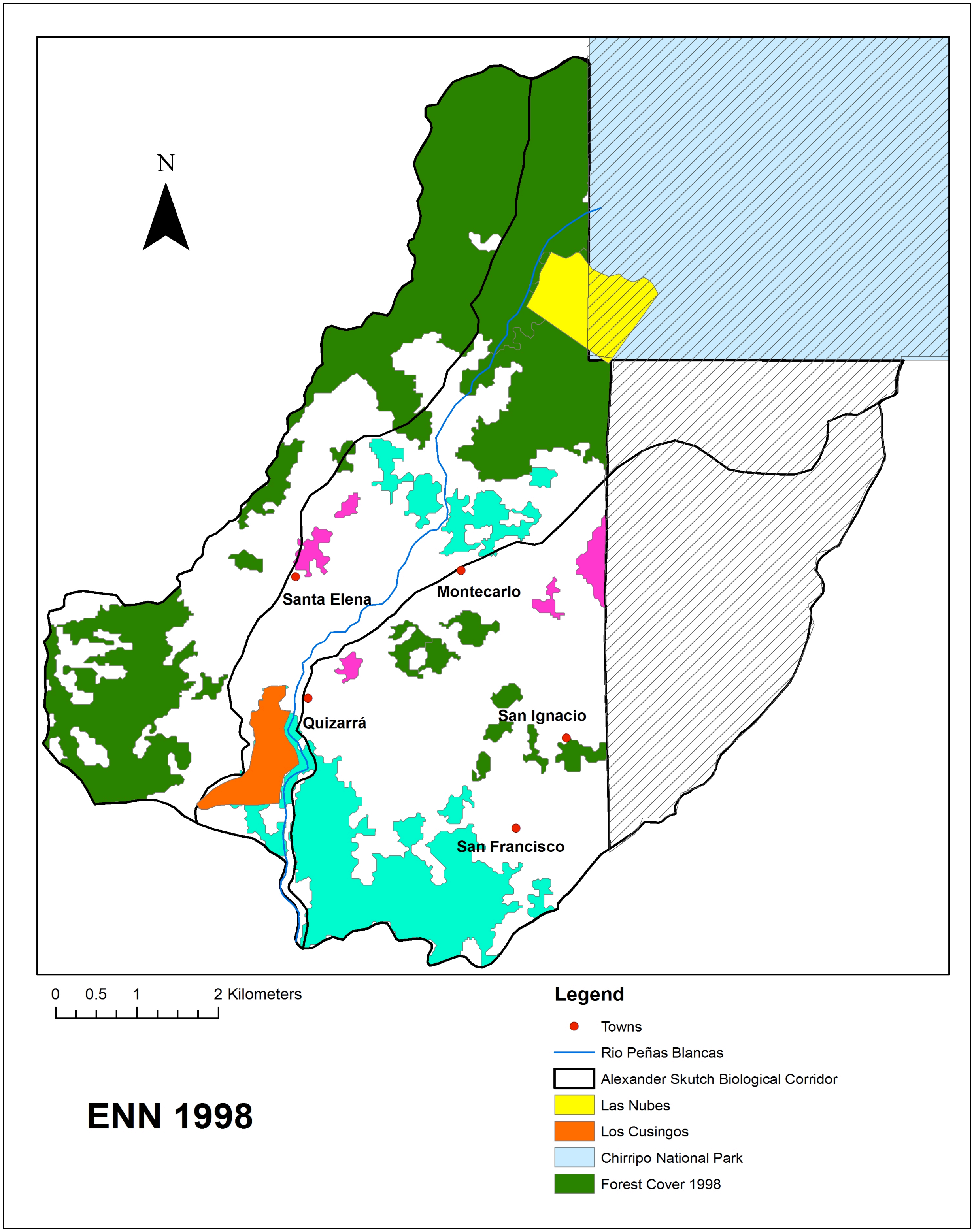
4.1.3 Connectivity Indicators

Two connectivity indices were calculated with FRAGSTATS software in order to gauge the relative distance and isolation of forest patches in 1998 and 2008: the Euclidean Nearest Neighbour Distance (ENN) metric and the Connectance Index (CONNECT).

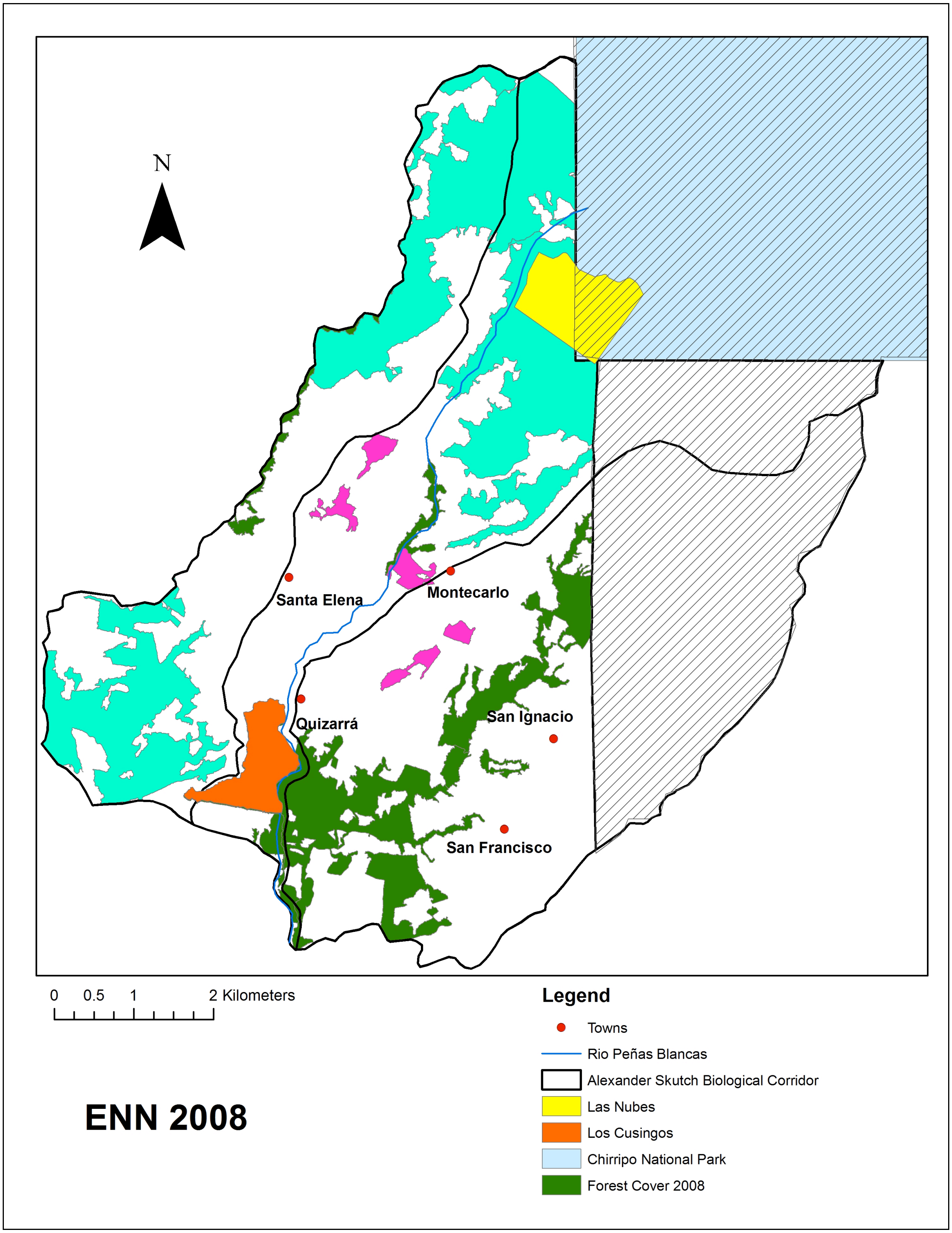
In 1998, the average ENN distance between forest patches in the study area was 144.31 meters and ranged up to 332.42 meters. However, in 2008, the average ENN distance between forest patches actually decreased to 64.50 meters, with a range of up to 169.71 meters. This finding is consistent with the calculations from the Clumpiness Index metric in showing that forest patches in 2008 are more aggregated than in 1998. Forest patches that had the greatest ENN distance between them in 1998 were generally located near the towns of Montecarlo or Santa Elena (Map 4.3). The forest patches with the least distance between them in 1998 were located in the southern end of the corridor and south of Las Nubes Reserve (Map 4.3). In 2008, the forest patches that had the greatest ENN distance from each other were similarly located near the towns of Montecarlo and Santa Elena. The forest patches with the least distance between them in 2008 were primarily found north-west of Las Nubes Reserve, the reserve itself, and in the south-west region of the corridor (Map 4.4).

In terms of “connectivity” between forest patches, in 1998 39.13% of forest patches were within 100 meters of one another. In contrast, this number rose to 72.73% of forest patches in 2008. This means that the degree of connectivity between remnant forest patches in the study area increased by over 33% between 1998 and 2008. However, this apparent increase can be misleading: given that forest patches in 2008 are more aggregated than in 1998, the increase in connectivity may be attributed to forest patches having a clumped distribution. A clumpy distribution can mask isolation that may be occurring in a particular area; in this case, there are fewer patches larger than 5 ha south of Las Nubes and north of Los Cusingos in 2008 than in 1998. Thus, “connectivity” between patches appears strong on an east-west axis in the northern and southern ends of the corridor compared to the central regions, which could make overall measurements of forest patches seem more connected than they actually are.

In short, the above data indicates that changes in forest cover between 1998 and 2008 have, for the most part, been negative ecological changes. Since 1998, forest patches in the ASBC have decreased in size, overall extent, and the amount of core-to-edge habitat. In contrast, forest patches in 2008 are somewhat more aggregated and connected than in 1998, though are not evenly distributed.



Map 4.3: Euclidean Nearest-Neighbour distance between forest patches in 1998. The blue areas represent the forest patches with the least distance in meters from other patches, while the pink areas represent the forest patches with the greatest distance in meters from other patches.



Map 4.4: Euclidean Nearest-Neighbour distance for forest patches in 2008. The blue areas represent forest patches with the least distance in meters from other patches, while the pink areas represent the forest patches with the greatest distance in meters from other patches.

4.2 Land Use and Landscape Connectivity, 1998-2008

Unlike forest cover data, it was not possible to directly compare how land uses outside of forest patches larger than 5 ha had changed between 1998 and 2008. This was primarily due to the fundamental differences in the original satellite image used for both years, namely stark contrasts in resolution levels, and the classification procedures and land use categories used. Additionally, there were areas of missing data in the 2008 land use map due to the absence of forest patches larger than 5 ha in the corridor, creating a difference in the percentage of the study area that actually contained data between 1998 and 2008 (Maps 4.5 and 4.6). This would not necessarily be an issue if land uses were not typically stratified across altitudinal gradients. For example, sugar cane is usually cultivated at lower elevations, coffee at mid-elevations, and pasture is more commonly found at higher altitudes than other land uses because steeper slopes are less suitable for intensive agriculture (Young, 2001). Considering that the missing areas of the corridor in the 2008 map correspond to mid-elevations, drawing a comparison between the proportion of land uses occupying the corridor in 1998 versus 2008 may inadvertently under-represent the proportion of coffee in the corridor (Map 4.6).

Furthermore, since the coffee, sugar cane, and other agriculture categories are classified under the broad “permanent agriculture” class according to the IMN and TSC, the actual proportion of these categories that occupy the “permanent agriculture” class in 1998 is unknown. This makes it nearly impossible to quantify the degree to which land uses have changed over time.

That being said, personal observations and interviews with residents in the corridor revealed that cane fields are rarely cultivated above 800 meters elevation. Coupled with the fact that international coffee prices in 1998 were the highest they had been in a decade, before falling sharply in 1999, it is very likely that the majority of lands classified under the “permanent agriculture” category in 1998 were, in fact, coffee plantations (Hallam, 2003) (Map 4.5).

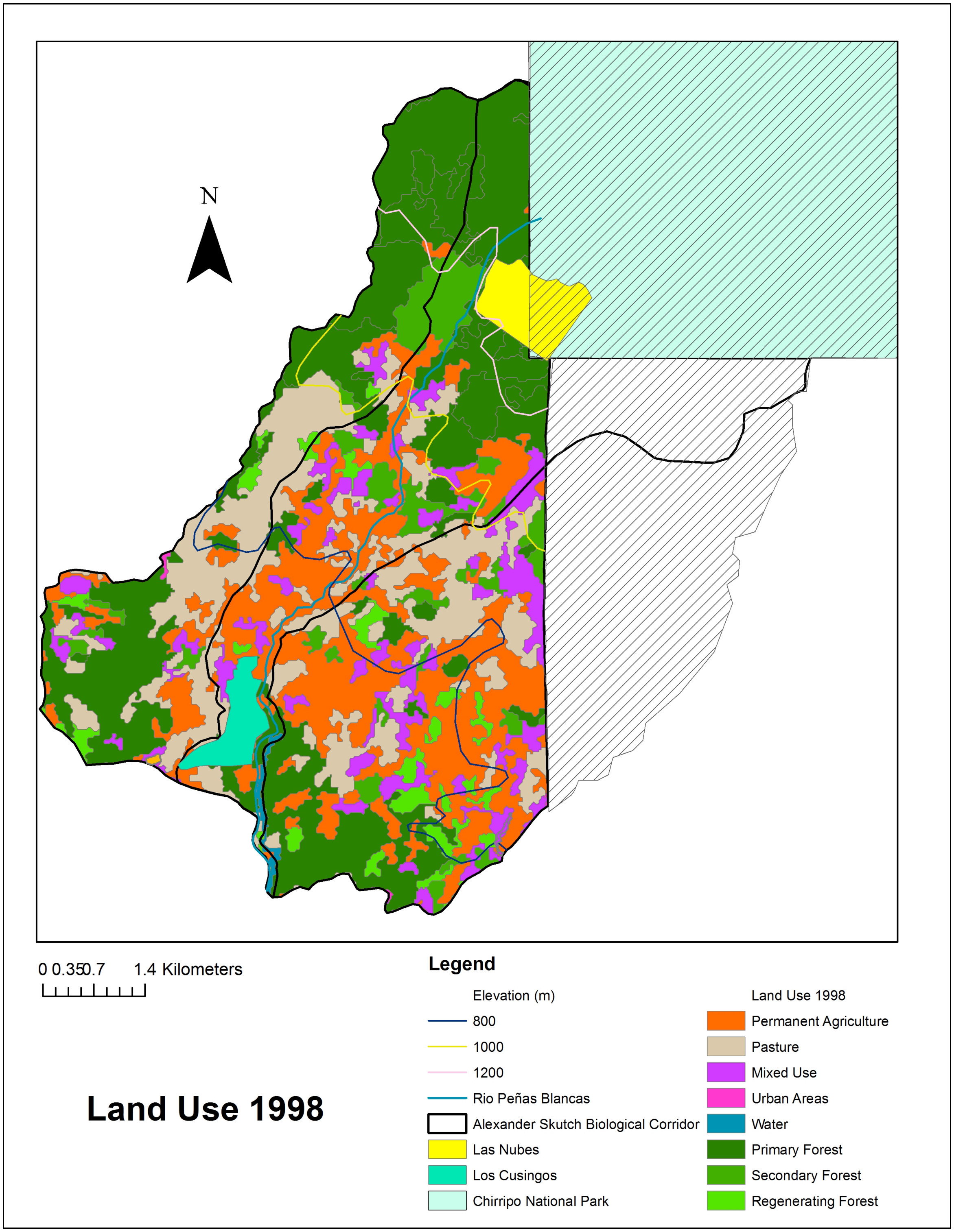


Figure 4.5: Land use in 1998 in the ASBC. Land use categories were derived from Landsat TM images from the IMN and TSC.

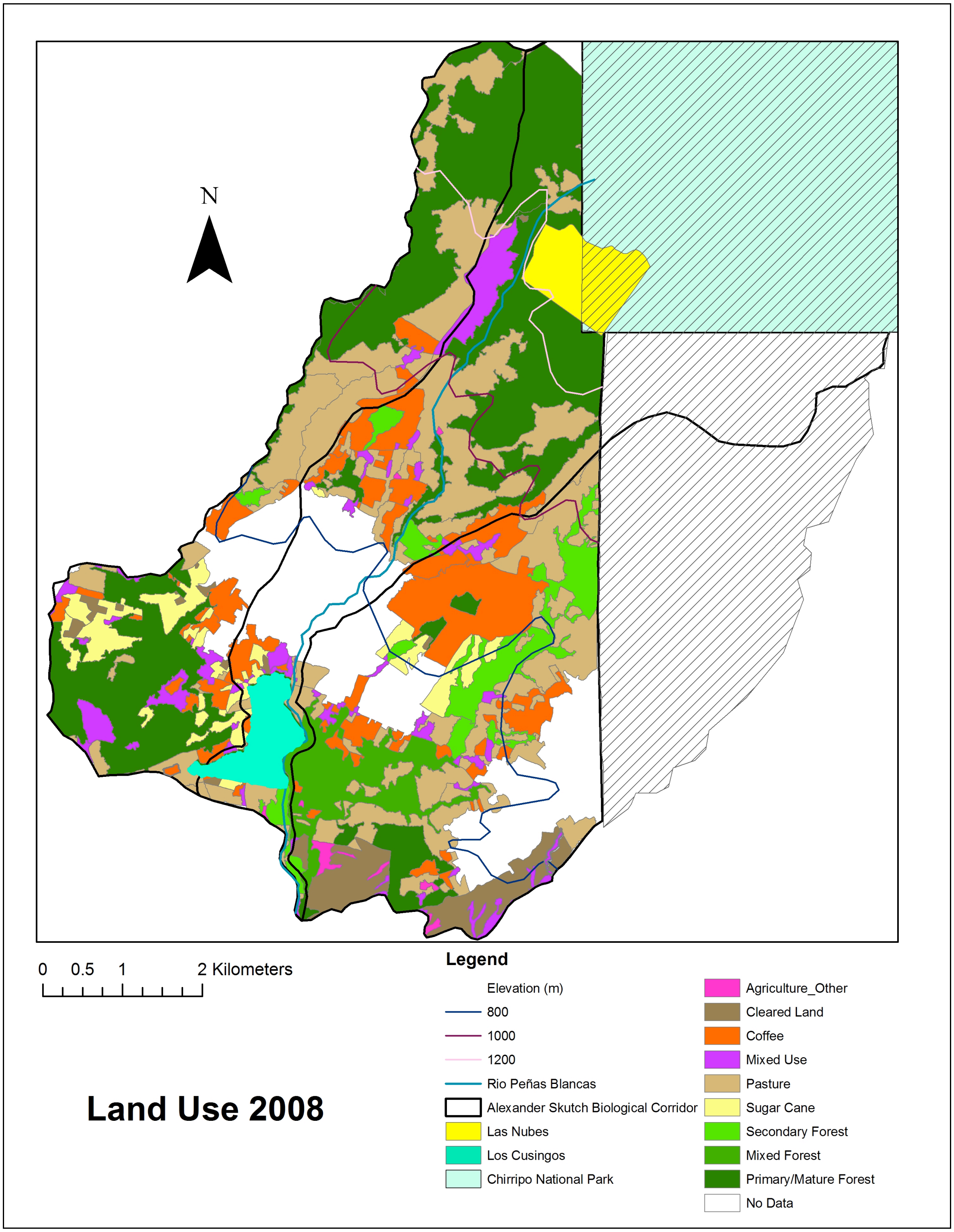


Figure 4.6: Land use in 2008 in the ASBC. The white "No Data" areas in the map represent areas within the corridor that did not have forest patches larger than 5 ha, and thus were not mapped while in the field.

Table 4.1 displays the proportion of land uses recorded for the 3,809.59 ha of the ASBC that contained data for 2008. These figures exclude the ‘no data’ areas from calculations.

Table 4.1: The distribution of land uses for the year 2008.

Land Use	**Total number of hectares**	**Percentage of total area (/ 3809.59 ha)**
Forest (> 5ha)	1716.19	45.05%
Pasture	1061.37	27.86%
Coffee	492.72	12.93%
Mixed Use	179.32	4.71%
Sugar Cane	179.22	4.70%
Cleared Land	157.90	4.15%
Agriculture (other)	16.87	0.44%
Total	**3809.59**	**100%**

Due to the uncertainties described above, a land use analysis of the connectivity potential of different land uses in the corridor could only be performed for the 2008 data. As described in the *Methods* chapter, landscape connectivity is a measure of the similarity between the vegetation structure of the external matrix to that of a forest patch (Lindenmayer & Fischer, 2006). An analysis of landscape connectivity in 2008 reveals that potential connectivity is strongest in the areas north of Santa Elena and just south of Montecarlo. In contrast, landscape connectivity is weakest in the south-west and south-east corners of the corridor where cleared lands, pineapple plantations, and cane fields are present outside forest patches (Map 4.7).

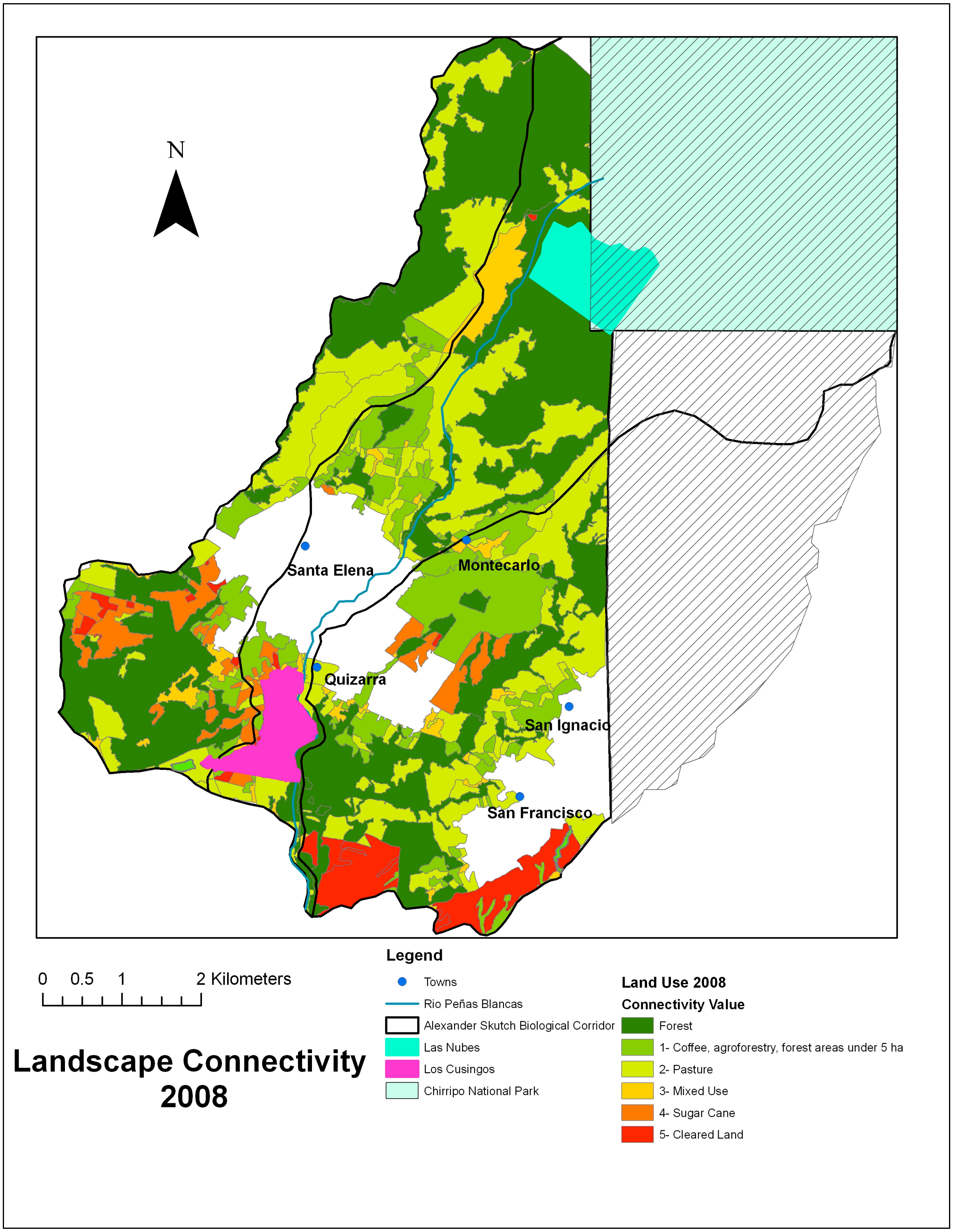


Figure 4.7: Potential landscape connectivity in 2008. Landscape connectivity values are based on the degree of similarity between the vegetation structure of land uses in the matrix to that of forest patches, a value of "1" being the highest, and "5" being the lowest.

4.3 Interview Findings

Although the purpose of conducting interviews with residents was to add additional information related to land uses and land use changes in the ASBC (*Objective IV)*, the survey sample was not necessarily representative of the actual demographic profile for the corridor. This was largely because participants were selected based on specific criteria, which generally hinged on the knowledge of the local resident hired to assist with field work. In order to better represent the demographic profile of the ASBC, a more developed social science survey is needed. Nevertheless, the information provided by the 26 participants in the present study still holds value for identifying land use changes, socio-economic drivers, and overall trends that have occurred since 1998.

4.3.1 Social Data

More than two-thirds of interview participants were male and over the age of 55. Participants had lived in the ASBC for an average of 41.5 years. Of the 26 respondents, just over 60% had a highest level of completed education at the elementary level, while another 19% said to have some post-secondary education. Nearly 58% of respondents classified their current occupation as 'agricultural'; 18.46% of respondents classified their current occupation as 'other' (which included work in the home, managing restaurants, cafes, or small stores); 15.39% of respondents classified their current occupation as 'commercial' (the sale of goods and foodstuffs); and, about 7% classified their current occupation as 'administrative/managerial' for farms within the corridor. Over 40% of respondents make less than $6000 USD per year, while approximately 30% of respondents could not specify an income for the past year due to diverse sources of income and variable crop prices.

4.3.3 Land Use Changes Observed in the Corridor

Respondents were asked to list the predominant type of land use changes they had observed over the past 10 years within the corridor. Respondents could list multiple changes, and a total of 57 responses were recorded for this question. Of the responses, 18% indicated that there are now more properties devoted to pasturelands. Over 11% and 7% of responses indicated that fewer coffee farms and sugar cane fields, respectively, were present in the corridor, and a further 7% of responses suggested that pineapple farms have increased in recent years. Over 12% of responses indicated that there a fewer agrochemicals being used on farms within the corridor.

More than 27% of responses cited that the number of forested areas in the corridor had increased in the past decade, whereas nearly 10% of responses suggested that there were fewer forested areas. The apparent contradiction of these responses may be attributed to the increased reforestation initiatives by Cocoforest, a local organization devoted to increasing forest cover on farmlands and private properties in the corridor. A number of residents indicated that they had seen small areas (<2 ha) being replanted, particularly in the towns of Santa Elena and Quizarrá, which is where Cocoforests' efforts are typically concentrated. Loss of forest in the past decade was usually in reference to a specific event or geographic area, meaning that residents living or working within observable distance from a specific area within the corridor may have collectively been referring to the same event. These factors could explain the difference in responses.

Over 42% of respondents thought that most land use changes in the ASBC were a result of increased environmental awareness and education. In contrast, nearly 35% of respondents believed land uses were changing (specifically in reference to the conversion of coffee plantations to pasture) due to unstable crop prices and/or low returns for labour input. Over 15% of respondents indicated that policies and management practices promoted by the Costa Rican government could account for these trends.

4.3.4 Payment for Environmental Services Program (PES)

Respondents were questioned about their awareness of the PES program offered by the Costa Rica Government, and were also asked to offer comments about the program's benefits and applicability within the corridor. The PES program "allows private forest owners to be compensated for the services they provide through such activities as reforestation, sustainable forest management, and forest conservation" (Zbinden & Lee, 2005). A more detailed description of the program is provided in the *Recommendations* chapter.

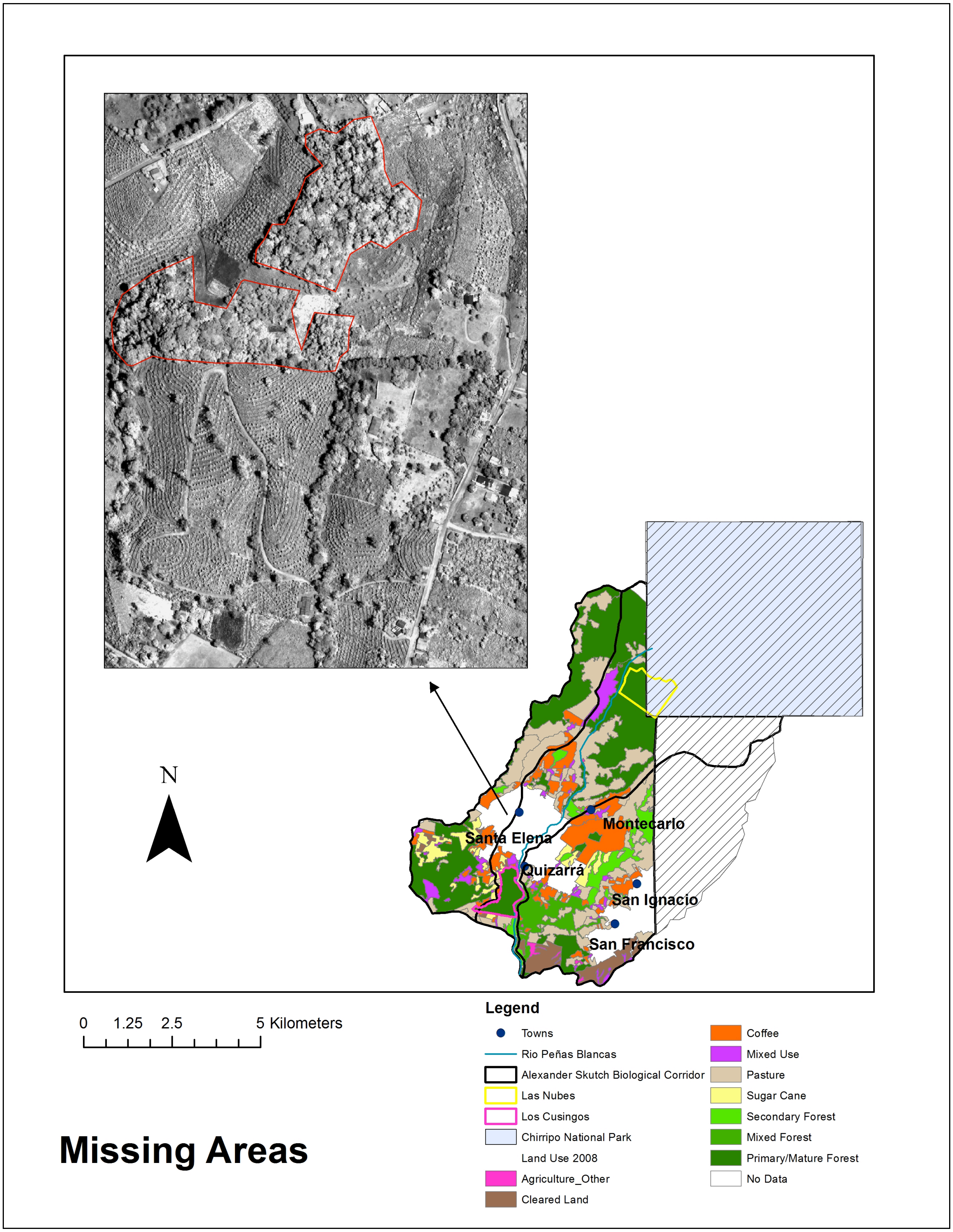
Only 22 out of 26 respondents were asked this question due to issues of respondent availability and time/travel constraints. Of those respondents, about 82% said they had heard of the PES program, or a program similar to the one offered by the Costa Rican government (for example, the local farmer's cooperative, CoopeAgri, offers a similar reforestation-incentive program). Of these respondents, one-third commented that a lengthy bureaucratic application process, including contractual obligations (which several respondents cited as intimidating), hindered participation. Nearly 20% of respondents mentioned that participation in the program was limited by insufficient compensatory funds to farmers. However, about 31% believed that the PES program was a good alternative to land uses in the corridor, and could be useful for encouraging reforestation and conservation initiatives. Approximately 17% of participants thought that participation in the PES program would likely increase if more education and information was available to the community.

ii. Discussion

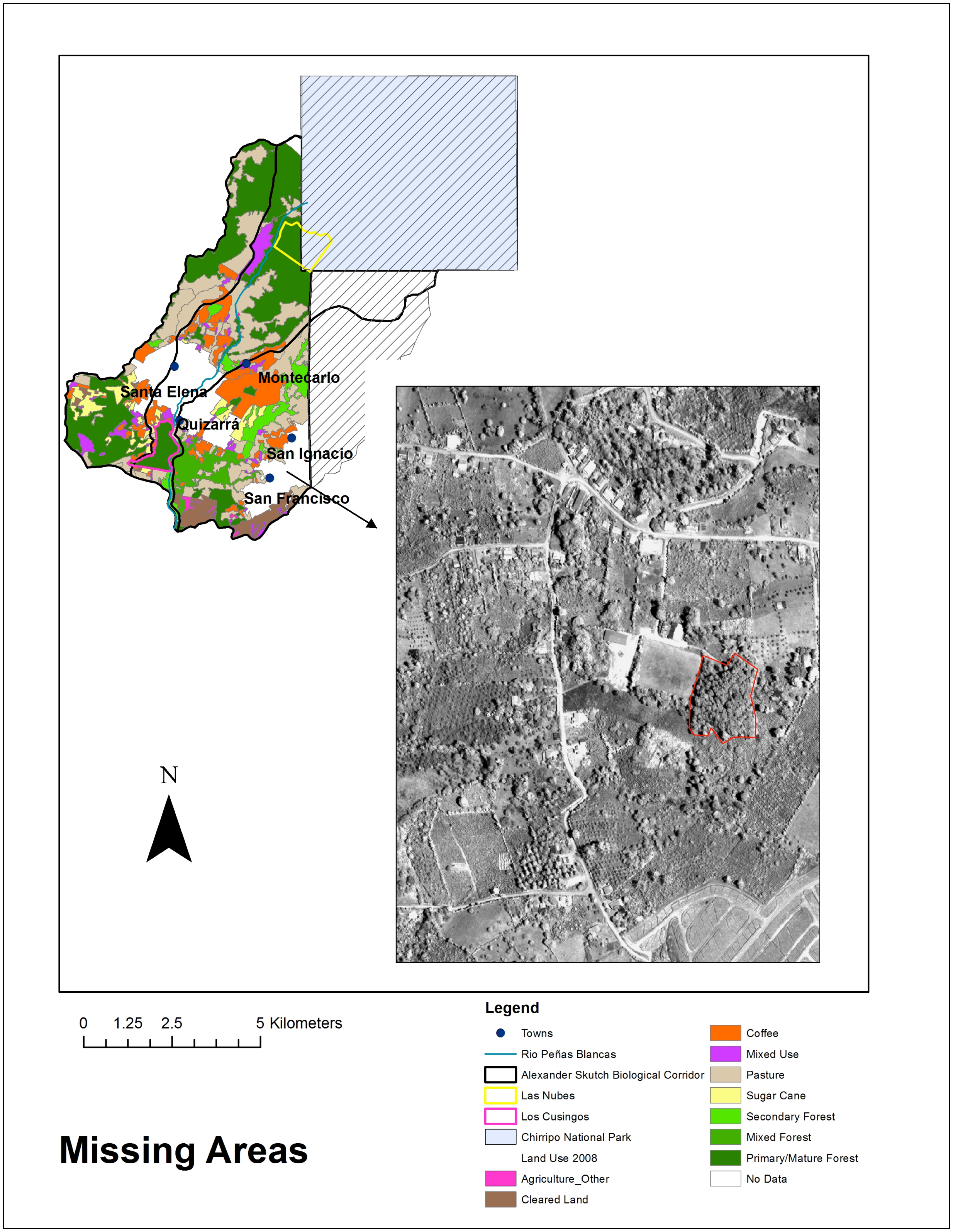
4.4 Evaluation of Forest Cover Change 1998 – 2008

To the extent that the Alexander Skutch Biological Corridor is intended to protect biodiversity and key ecological processes by linking forest fragments, and through sustainable land use management, the recorded 19% decrease in forest area over a ten-year period is alarming, to say the least. At an average deforestation rate of 1.9% per year, this rate of loss is higher than the national average rate of 1.5% recorded for the late 1990s, and even higher for the rate recorded by the FAO of -0.3% for the year 2005 (FAO, 2005; Sanchez-Azofeifa et al., 2003). Theoretically, if this deforestation rate were to continue unabated, the forest in the ASBC would be completely gone within 20 years.

However, the deforestation rate for the corridor is likely over-estimated for the study period because only forest patches larger than 5 ha were recorded. Although it is possible that the inclusion of these smaller forest patches could off-set the amount of forest that has been lost since 1998, as Maps 4.8a and 4.8b illustrate, these areas are dominated by patchy, heterogeneous land uses, with only the occasional forest patch. In fact, the inclusion of the < 5ha forest class would only add an estimated 45.5 ha to forest cover data for 2008, and thus would likely have a significant impact on deforestation rates in the corridor.



Map 4.8a: Sample image of one of the "no data" areas in the corridor for the year 2008. Images are taken from the World View 1 Satellite. Forest patches under 5 ha in the no data areas are outlined in red.



Map 4.8b: Sample image of one of the "no data" areas in the corridor for the year 2008. Images are taken from the World View 1 Satellite. Forest patches under 5 ha in the no data areas are outlined in red.

Upon examination of changes in forest cover between 1998 and 2008, it is clear that large areas of forest have been lost from the highlands in the north-west of the corridor and in the lowlands in the south-east (Map 4.9). Smaller forested areas have also been lost or reduced north of the town of Santa Elena and south of the town of Montecarlo. Thus, it appears that forest loss and fragmentation are occurring nearly throughout the entire corridor. The corresponding decrease in average forest patch size and core area, coupled with a higher forest patch shape complexity indicates that the ASBC has undergone a significance amount of negative environmental change in the last decade.

On the other hand, forest patches in 2008 appear to have a lower degree of isolation than in 1998. Physical connectivity between forest patches, in particular, has emerged in the forested areas east of the Río Peñas Blancas. As Maps 4.2 and 4.9 illustrate, patch connectivity can be observed north-east of Montecarlo, extending south to the large forested areas near Quizarrá and San Francisco. This increase in physical connectivity between previously separated forest patches in this area of the corridor can be attributed to a large, 62.41 ha regenerating forest area on the property of Finca Bernina (Map 4.2). In addition, because narrow riparian corridors and forested ‘fingers’ extending from patches can be distinguished in the 2008 0.5-meter resolution imagery, the actual edge-to-edge distance between patches in 2008 is reduced. Accordingly, this may also mean that edge-to-edge distance of forest patches in 1998 is not as great as initially calculated since it is likely linear forest areas were not mapped with the 30-meter resolution Landsat TM imagery.

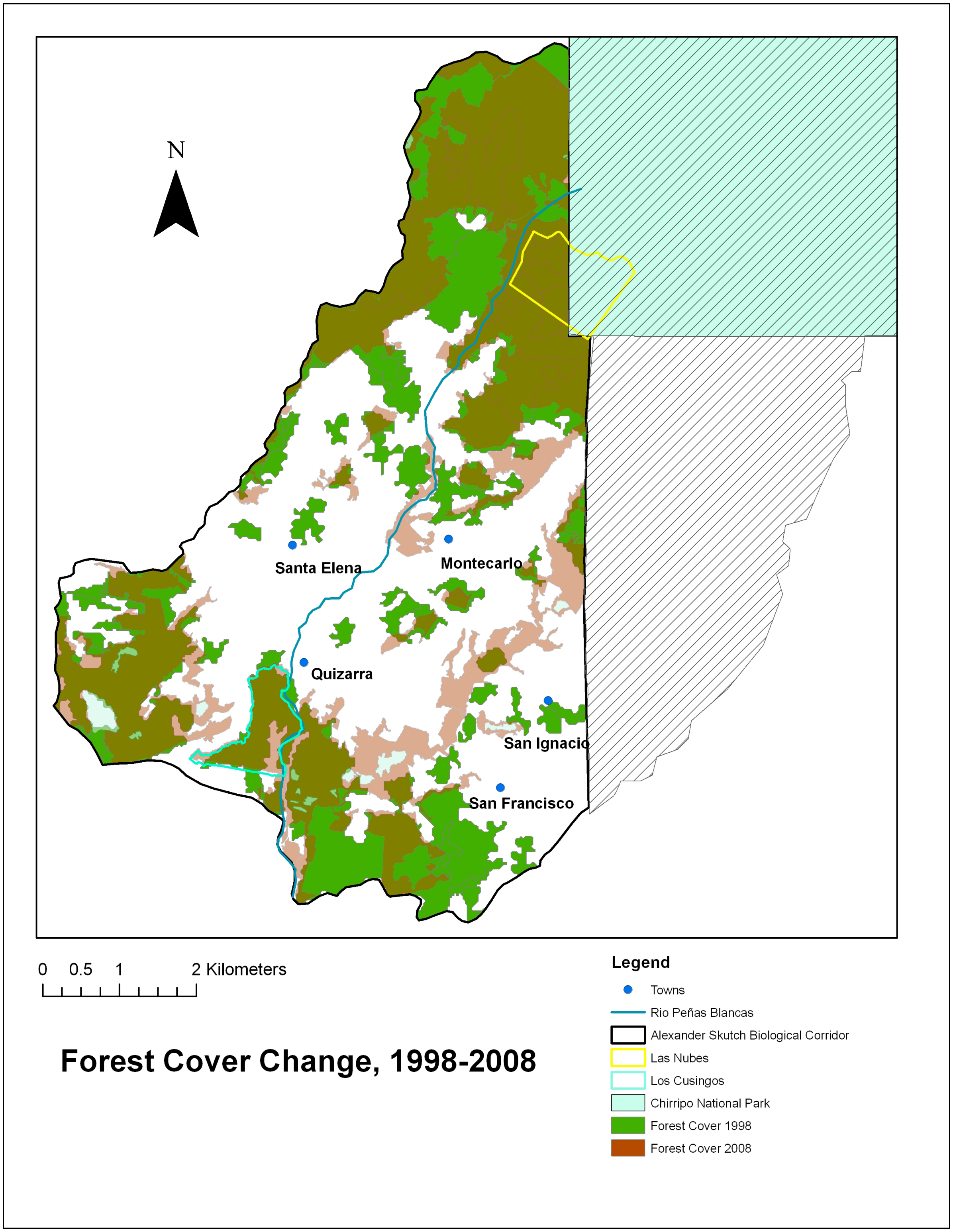
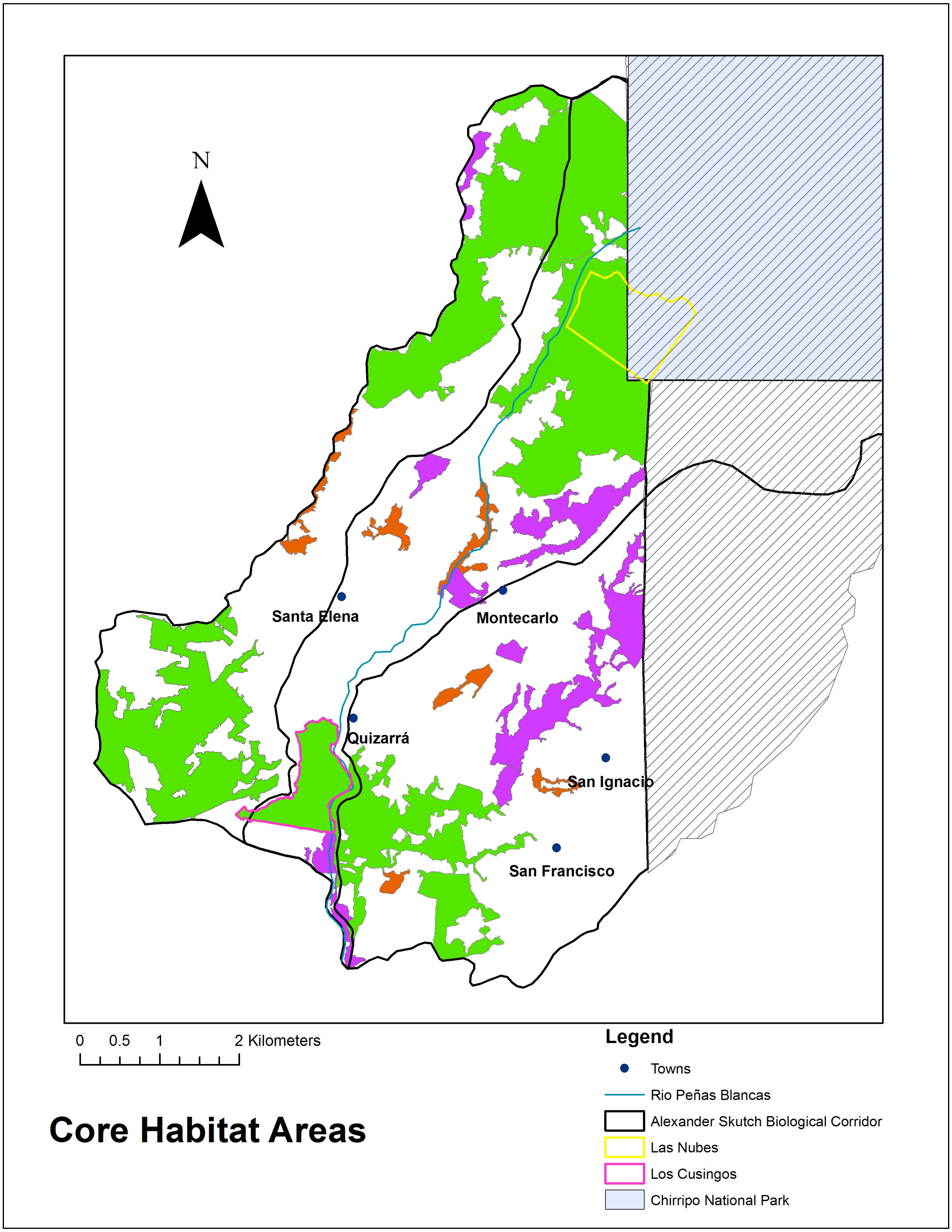


Figure 4.9: Forest cover change between 1998 and 2008. The 2008 forest cover layer is overlaid on top of the 1998 data, and faded out. Green indicates areas where forest has been lost; brown indicates no change, and the rose colour indicates either where forest has been gained, or where forested areas were present in 1998 but were not mapped due to mapping inaccuracies (see section 3.5 *Accuracy Assessment)*.

Despite this apparent decrease in isolation, the overall loss of forest and reduction in average forest patch size as well as core area is a serious cause for concern for the long-term survival of native species in the corridor. As Fahrig (1997) demonstrated in her examination of the effects of habitat loss versus habitat fragmentation and isolation alone, the effects of habitat loss far outweighed the effects of fragmentation and isolation. That the major loss of forest has primarily occurred in the largest forested areas in the corridor means that these essential conservation building blocks may be less able to sustain species populations and ecological processes than in 1998. The need for large and extensive forested areas in the corridor is reflected in the findings of a three-year monitoring project in the ASBC that examined the relationship between avian diversity, species abundance, and habitat type (Valdez, 2008). Between 2005 and 2008, the project found that, of the eleven study sites in the corridor, the sites with the highest species diversity were the largest, least-disturbed forest areas- Las Nubes Reserve and the forests near Los Cusingos (Valdez, 2008). Furthermore, the proportion of endemic species was also found to be higher in these sites than in other sites, implying that forest tracks of this size are essential for providing habitat to specialized or rare bird species (Valdez, 2008).

The changes in forest cover since 1998 are not just a cause for concern for overall loss of habitat, but also for the increased edge habitat in the corridor. This change is both a function of increased shape complexity of remnant patches (i.e. an increase in linear or convoluted patches), and a proportional reduction in average patch size. Although the distance beyond which edge effects are detected and begin to have a detrimental effect may vary from patch-to-patch, species-to-species, and ecosystem-to-ecosystem, the study results indicate that the proportion of forest patches that are exposed to potential edge effects has increased markedly since 1998. Other studies have shown that edge effects and area-related extinctions have a progressively greater impact as forest patches decrease in size, particularly those under 100 ha (Sodhi et al., 2007; Laurance et al., 2002; Saunders et al., 1991). In fact, out of the 22 forest patches identified in the corridor in 2008, only 16/22 or 73% contained some core area (Map 4.10). Only six out of these 16 patches had more than 20% of their total area classified as core area, and these patches were all larger than 55 ha.

Patch size was not the only predictor of core area for 2008: patch shape also influenced core-to-edge ratios (Sodhi et al., 2007; Daily & Ehrlich, 1995; Forman, 1995). For example, the 11.28 ha forest patch on the El Grano Tico coffee farm has 9.21% of its area classified as core area, whereas the 11.53 ha forest patch immediately to the south of it has 0% core area (Map 4.10). Despite the similarity in size, the differences in core area must exist because the El Grano Tico patch has a lower shape complexity index (1.88 versus 3.28). As such, the higher the shape complexity and the smaller the forest patch size, the less likely a patch will contain any core habitat, based on the 100-meter edge threshold. As the habitat quality of small patches is heavily affected by edge effects, patches below a certain size threshold may consist only of edge habitat, and therefore may not support forest-interior or sensitive species (Daily & Ehrlich, 1995).



Map 4.10: Core vs. non-core habitat areas in the ASBC for 2008. The orange forest patches do not contain any core habitat area; the purple patches contain between 0-20% core area, and the green patches contain more than 20% core habitat area.

4.5 Evaluation of Matrix Conditions, 2008

Examining the degree to which landscape conditions outside forest patches may or may not permit dispersal or provide some other ecological value for species is wrought with uncertainty, and even more difficult to generalize. Nevertheless, there is an overall consensus that the more closely the external vegetation structure reflects that of a forest patch, the more likely edge effects will be reduced and that some species will be able to use the matrix (Sekercioglu et al., 2007; Sodhi et al., 2007; Lindenmayer & Fischer, 2006; Daily et al., 2003; Laurance et al., 2002; Gascon et al., 1999). For example, findings from the BDFFP suggest that species that are able to use the matrix for movement, escape/hiding, foraging, or breeding tend to remain stable, whereas species who do not use the matrix are more likely to decline or disappear (Gascon et al., 1999). Studies of tropical species near Las Cruces Reserve in southern Costa Rica have shown that as much as 75% of bird species will use the matrix to meet some of their needs if forest patches are nearby (Sekercioglu et al., 2007; Daily et al., 2001). Specifically, shade-grown coffee plantations adjacent to forest patches in the Las Cruces region were found to have higher species richness levels than pasturelands adjacent to forest patches (Daily et al., 2003). Although coffee plantations alone were not found to be suitable habitat for many species, when the coffee was grown under a diverse canopy of trees, Daily et al. (2003) found that these types of plantations could facilitate movement between remnant forest patches. Similarly, the avian monitoring project in the ASBC revealed that many migratory bird species, in particular, preferred shade-grown coffee farms adjacent to secondary forest patches as opposed to mature forest patches alone (Valdez, 2008).

Shade-grown coffee is also believed to provide better erosion control, improve soil quality, and provide resources for other surface-dwelling fauna (Daugherty, 2005; Hall 2001). Based on these findings, shade-grown coffee plantations adjacent to forest patches have a greater ecological value than other agricultural uses such as pasture and sugar cane. Thus, although it cannot be concluded one way or another which species and under what conditions will travel between forest patches in the ASBC, it can be surmised that species, overall, are more likely to survive in a landscape that has partial tree cover incorporated into agricultural areas than none at all.

The obvious increase in pasturelands in the highlands of the corridor over the past decade, in conjunction with an apparent decrease in coffee plantations, suggests that matrix conditions have become more ecologically stressful since 1998 (Maps 4.5 and 4.6). Responses from interviews also reflect that more farmers are converting their coffee plantations to pasturelands as a result of greater economic returns and a lower labour demand. Upon examination of commodity exports from Costa Rica according to the FAO for the years 1998 and 2004 (the most recent data), coffee exports have indeed decreased nation-wide (FAO, 2008). While coffee commodities have experienced a 19% decrease in the number of metric tons exported since 1998, the amount of exported sugar has increased 133%, and exports of pineapple increased a dramatic 250% between 1998 and 2004 (FAO, 2008). Within the corridor itself, one of the greatest losses of forest since 1998 has been the result of a massive pineapple plantation south of the towns of Quizarrá and San Francisco, as cited by several interview respondents. A comparison of land use maps between 1998 and 2008 clearly shows that a large area of cleared land (harvested pineapple) has emerged in the south-east (Maps 4.5 and 4.6). In fact, when measurements of forest and cleared land are compared for this area, results show that the pineapple plantation has caused over 100 ha of primary lowland rainforest to be lost over the past decade. Of the 403 ha of forest lost between 1998 and 2008, the pineapple plantation alone accounts for nearly 25% of forest area lost.

According to the 2008 landscape connectivity map (Map 4.7), the south-west and south-east sections of the corridor are the least conducive to permitting use of the external matrix due to minimal vegetation cover in these areas. Sugar cane fields and large-scale pineapple plantations in the hundreds of hectares dominate these areas of the corridor. The forest patches in these regions, as a result, may be more susceptible to environmental degradation and species loss. Considering that these areas are one of the few places in the corridor where forest patches are larger than 100 ha, the combined loss of forest and a more intensively-used matrix may impair genetic exchange between species populations and result in smaller populations that are more vulnerable to stochastic environmental events. Even if the majority of native biota and characteristic ecological processes are present in these areas, overall ecological integrity may be undergoing a slow decline. The effects of hunting, pollution, habitat loss, climate change, and the encroachment of humans on remnant forest patches tend to act synergistically, thus remaining forests may be under assault from multiple threats (Peres & Michalski, 2006; Daily et al., 2003).

In short, results from the 1998 and 2008 data and interview responses indicate that those land uses that may have some ecological value to species, such as shade-grown coffee plantations, have decreased over time. It is clear that if forest loss and land use trends continue along this trajectory, the ecological integrity of the ASBC will decline, perhaps irrevocably. The continued and on-going pressure from anthropogenic activities in the corridor thrusts upon the TSC, student researchers, and the communities within the corridor the responsibility of taking strategic and appropriate action to prevent further loss of ecological integrity and to restore habitat, where possible.

Chapter 5. Recommendations

As the ecological restoration of key habitats and degraded lands are part of the strategic goals of the Alexander Skutch Biological Corridor, the 2008 forest cover and land use data offers valuable insight into where restoration efforts are most needed. On the other hand, the restoration of tropical lands is not a simple process. Many studies have shown that the success of restoration projects are often influenced by the intensity of previous land uses, proximity to existing forest patches, soil quality, and competition from non-native species (Lamb et al., 2005; Endress & Chinea, 2001; Holl et al., 2000). For example, Endress and Chinea (2001) found that over 90% of naturally regenerating pasturelands occurred within 100 meters of already established forest patches in the Republic of Palau. Additionally, Holl et al. (2000) demonstrated that seedlings planted under remnant trees on pasturelands in southern Costa Rica had a higher survival rate than those planted in the open because microclimate conditions were more favourable under partial canopy-cover. The intensity and duration of previous land uses also affects the availability and quality of soil nutrients, as well as soil hydrological conditions (Peñula & Drew, 2004; Holl et al., 2000).

These and other factors can vary from site-to-site and from ecosystem-to-ecosystem, which makes it difficult to produce a set of best-practice guidelines for restoration techniques. Consequently, small-scale testing should be done in advance in order to understand the specific ecological conditions and potential constraints at a particular site (Van Dyke, 2003; Holl et al., 2000). Further studies may also be needed to determine what form and extent of restoration is most appropriate for a given site, particularly if localized restoration efforts are intended to contribute to a larger-scale restoration initiative (Lamb et al., 2005). These considerations might include enlarging a small forest patch versus establishing a vegetated corridor to connect remnant patches; or, choosing among direct seeding, planted seedlings, or natural regeneration as the desired technique (Lamb et al., 2005; Van Dyke, 2003). Because these considerations are beyond the scope of this study (though are crucially important for any restoration plan) the following recommendations for the ASBC are solely based on observations where the ecological restoration of non-forest land is anticipated to lead to improved ecological integrity or biodiversity protection.

5.1 Restoration Applications in the ASBC

5.1.1 Repair Internal Patchiness

Saunders et al. (1991), in their review of tropical forest fragmentation, asserted that the management of large forest patches should focus on controlling internal dynamics, while the management of small forest fragments should aim to control external dynamics. This approach has the overall goal of increasing core habitat area in large forest patches and insulating smaller fragments from edge effects and anthropogenic stresses. Given the well-documented negative effects of forest loss and edge effects on species populations and ecological processes, this philosophy should inform the restoration approach for the ASBC.

Non-forest patches that exist within established forest patches such as those on Schoeder's property in the south-west (Map 5.1), the area south of

Quizarrá, the north-west corridor, and the area north of Montecarlo, could greatly benefit from reforestation or natural regeneration, especially considering that the internal land uses of these areas are primarily pasture or sugar cane (Map 4.6 in *Results and Discussion*). Theoretically, if these lands were reforested, they would contribute an additional 146.13 ha of forest to the corridor, increasing total forest cover from 37.8% to 41% (Map 5.1). Moreover, the restoration of lands in the south of the corridor would increase the amount of forested area in the threatened lowland rainforest ecological life zone. Although the success of any restoration effort cannot be predicted without further studies, each of these proposed areas is within an existing forest patch, which means that seed dispersal from neighbouring areas would be readily available and in close proximity, thereby enhancing the probability of success.

5.1.2. Enlarge/Buffer Small Forest Patches

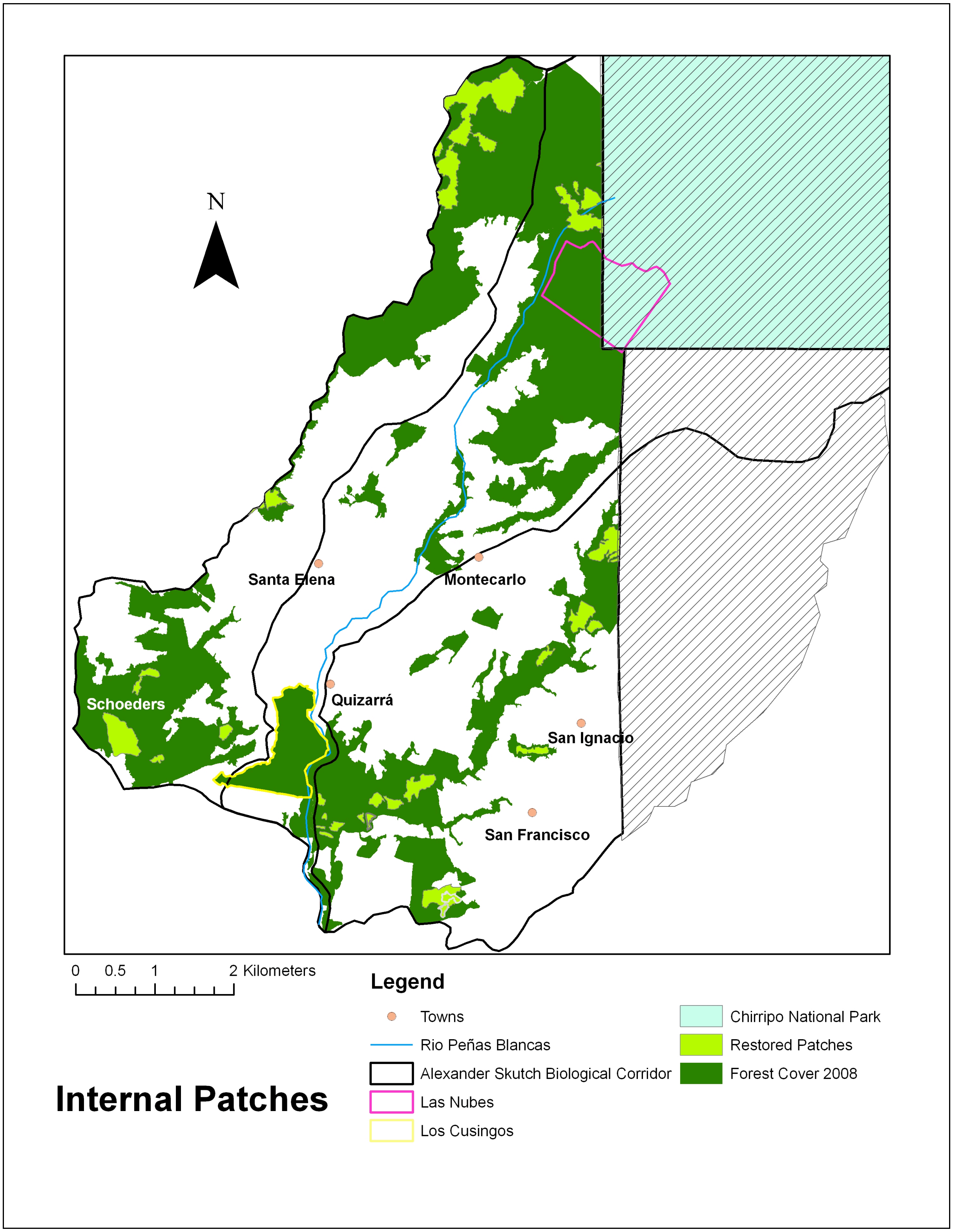
The absence of forest patches larger than 5 ha in the 'no data' areas of the corridor is certainly a cause for concern when it comes to the availability of suitable habitat for species, dispersal opportunities, and overall ecological health. As such, restoration efforts in these areas should focus on enlarging these small patches. Allowing natural regeneration to occur along the boundaries of patches is not only a relatively low-cost and low-maintenance way of increasing forest cover in the corridor, but it can also help buffer forested areas from external stresses and reduce the distance between remnant forest patches (Lamb et al., 2005) (Map 5.2).

5.1.3. Enhance Vegetated Corridors

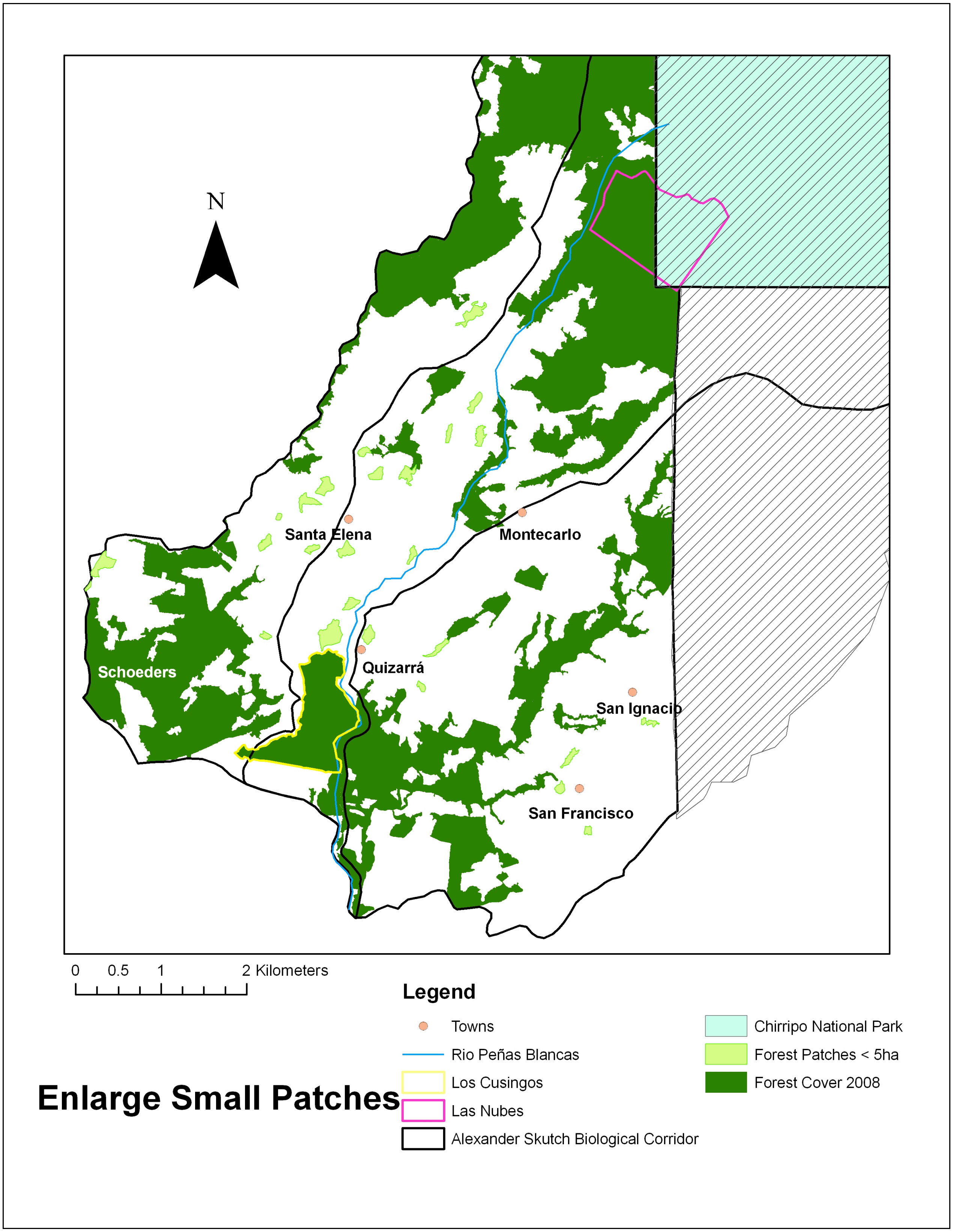
The concept of vegetated corridors as a conservation strategy is highly contentious, despite the fact that "landscape patterns that promote connectivity for species, communities, and ecological processes are a key element of nature conservation in environments modified by human impacts" (Bennett, 2003, p.8). A great deal of consideration must go into the planning and design of corridors if they are to have a beneficial impact on regional ecological integrity.

If deemed appropriate, a vegetated corridor should be implemented to link Schoeder's' property to Los Cusingos (Map 5.3). A linkage between these areas would create a series of continuously linked patches that would extend north-east of Montecarlo to the southern regions of the corridor. Vegetated corridors could also be strengthened between the forest patches south of Quizarrá (Map 5.3).

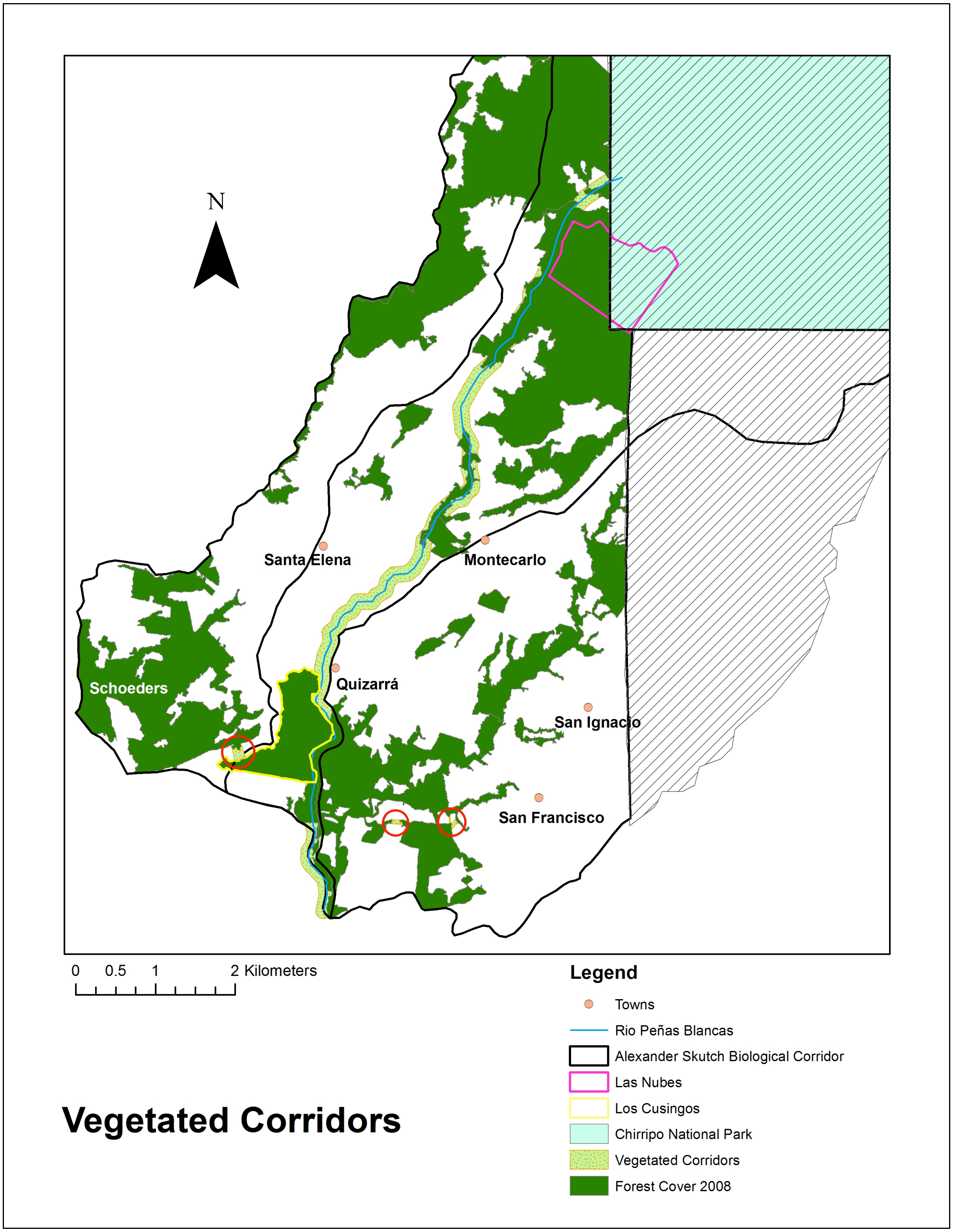
In the mid-1990s, the government of Costa Rica prohibited the removal of vegetation from lands within 10 meters of either side of a river on flat lands, and 50 meters in steeper areas (Schelhas, 1996). Additionally, the management plan for the ASBC has recommended a 100-metre buffer on either side of the RÍo Peñas Blancas (Jiménez-Salazar, personal communication). Ensuring the presence of these riparian corridors along major rivers is not only legally mandated, but it would also help to create a network of forested linkages throughout the entire corridor.



Map 5.1: Recommendation #1: Internal patches within forested areas in the ASBC should be ecologically restored in order to increase overall forest extent and core habitat area.



Map 5.2: Recommendation #2: Forest patches under 5 ha should be insulated from external stresses by allowing natural regeneration to occur along patch boundaries.



Map 5.3: Recommendation #3: Vegetated corridors could strengthen connectivity between forested patches in the southern part of the ASBC (highlighted with red circles). Maintaining the mandated 100-meter vegetated buffer along the Peñas Blancas can also increase habitat and connectivity on a north-south axis.

5.2 Incentives for Restoration and Sustainable Land Use Management

The sustainable management of lands outside forest patches is clearly an important operating goal of the ASBC if the corridor is to be managed under the concept of regional landscape conservation (Daugherty, 2005). Although this has not been a primary focus of the present study, the information obtained from land use mapping in 2008 will be central in planning and designing future restoration initiatives.

As Bennett (2003) and Schelhas and Greenberg (1996) emphasize, the conservation of off-reserve lands is strongly influenced by socio-economic factors, particularly how private landowners choose to use their land. Because ecological restoration initiatives within the ASBC will largely have to take place on privately owned lands, it is important to understand the values and motivations of rural landowners in the community. For example, in a survey of 50 residents in a community in southern Costa Rica, Jantzi et al. (1999) concluded that there are four major reasons why people value forest patches on their property: watershed protection, personal or economic uses, habitat preservation, and general environmental services. Participant responses from this study indicated that interest in forest conservation stemmed from past experiences with environmental degradation, religious values, community organizations and government policies or laws (Jantzi et al., 1999). This same study, and another that also took place in southern Costa Rica, both found that large land owners were more likely to have part of their property in forest because their lands did not have to be used as intensively as smaller land owners to generate income

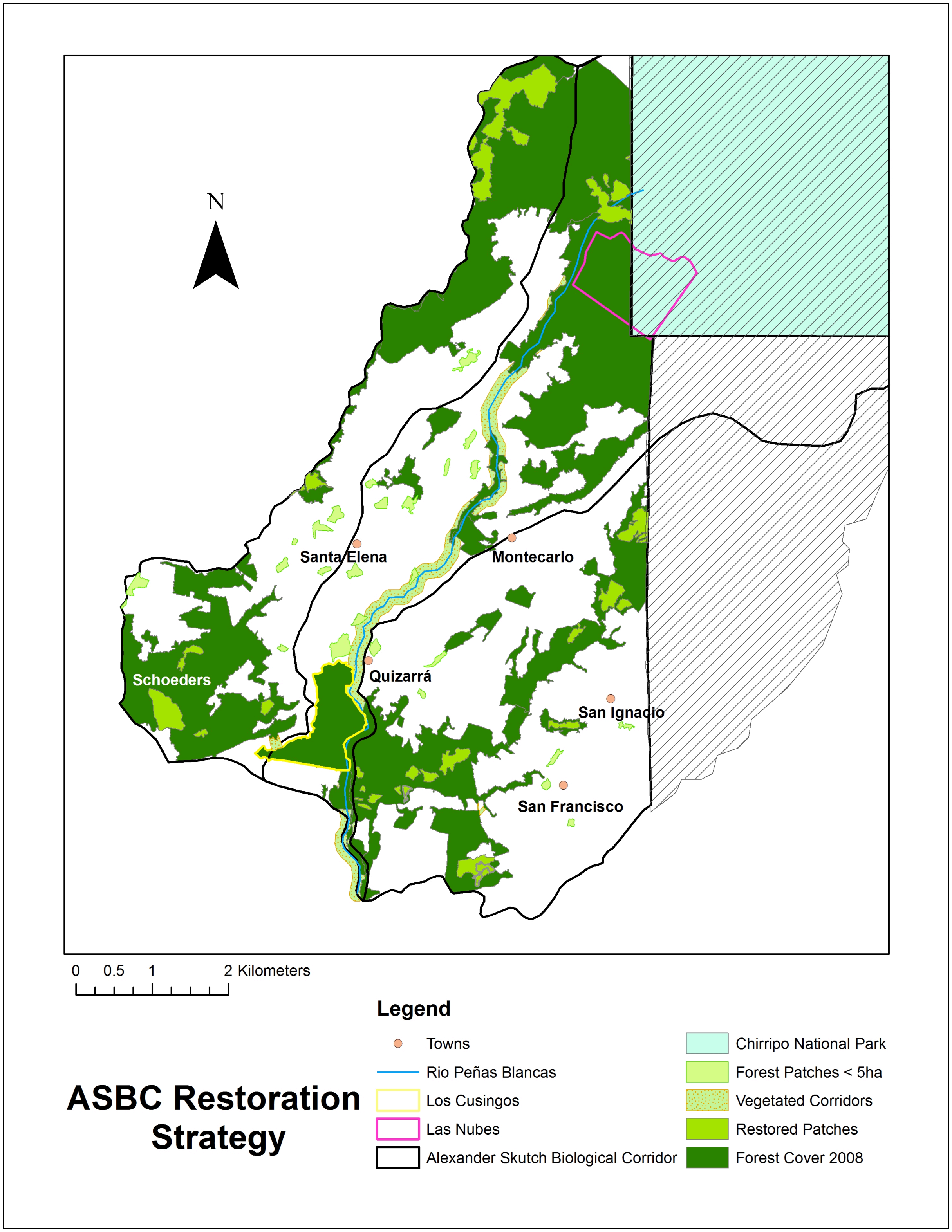
(Jantzi et al., 1999; Schelhas, 1996). It is no surprise that land-use decisions are heavily influenced by economics. As mentioned previously, this sentiment was reflected in the responses from interview participants in the ASBC, where nearly 35% claimed the motivation for converting coffee plantations to pasture lands was due to higher economic gains and lower labour inputs.

Recognizing the need to integrate attractive financial returns with the conservation of forested lands within Costa Rica, the national government established a Payment for Environmental Services (PES) program in 1997 (Sánchez-Azofeifa et al., 2007). This program aims to encourage the restoration of degraded lands, the sustainable use of forests, and the preservation of existing forest areas through financial compensation offered to program participants (Zbinden & Lee, 2005). It is this type of program that could enable restoration on private lands in the ASBC.

From interview results, it is clear that a large majority (82%) of respondents are aware of the PES program. Yet, of the respondents, only 2 out of 22 or roughly 9% actually participated in the program. Several factors may help to explain this large gap between knowledge and participation. First, participation in these market-based initiatives is often a function of education level and access to information (Zbinden & Lee, 2005). The ability to handle administrative tasks, enter into contracts, and deal with government bureaucracy was correlated with higher formal education of farmers in a census of PES participants in Costa Rica (Zbinden & Lee, 2005). Interview results from the present study show that over 60% of respondents had only an elementary-level education. Furthermore, 33% of respondents cited bureaucracy and contracts as an impediment to their participation in the program.

Secondly, a lack of adequate financial compensation from the government may also hinder participation. In a study by Sánchez-Azofeifa et al. (2007) of Costa Rica's PES program, average returns from the program ranged from (USD) $22-$42/ha/year, while returns from cattle-ranching ranged from $8-$125/ha/year. Larger landowners were also more likely to participate in the program because they could still generate income using another part of their property for agriculture (Sánchez-Azofeifa et al., 2007; Zbinden & Lee, 2005).

The application of the PES program in the corridor has the potential to prevent further losses of forest and to encourage the reforestation of key habitats in the area. Similar programs, such as reforestation-incentives offered by CoopeAgri, and the carbon off-set initiative currently being developed as the Las Nubes Carbon Fund also hold great potential for enhancing ecological conditions in the corridor. These initiatives could be used to encourage coffee and pasture farmers to increase the amount of tree cover on their properties, thereby making matrix conditions more ecologically favourable. In short, it is unrealistic to assume that restoration will occur on private-lands without attractive financial incentives, the right information, and appropriate facilitation and support. Map 5.4 illustrates the combined recommended restoration strategies for the ASBC.



Map 5.4: Recommendations for ecological restoration in the ASBC. Areas targeted for restoration include internal patches in forest areas, buffering small forest patches, and establishing vegetated corridors in key areas.

Conclusion

The ecological integrity of the Alexander Skutch Biological Corridor is in a precarious state. On one hand, a significant amount of forest area and interior forest habitat in the corridor has been lost since 1998. At the same time, whether it is through the conversion of coffee plantations to pasturelands or the expansion of pineapple farms, land uses in the ASBC appear to be less conducive to permitting species to use the matrix than in 1998.

On the other side of the coin, the corridor is supposed to be managed in a way that protects the ecological services and native biota in the region. Evidence from GIS-produced maps and FRAGSTATS statistics show that the corridor has experienced detrimental environmental changes since 1998; changes that have been linked to the loss of biodiversity and the impairment of ecological processes in many other studies examining the effects tropical forest fragmentation (Sekercioglu et al., 2007; Develey & Metzger, 2006; Daily et al., 2003; Laurance et al., 2002; Turner, 1996). Given the results from the present study, it appears that some of the operating goals of the corridor are not being fulfilled. Namely, the conversion of degraded pasturelands and sun-grown coffee plantations to more sustainable land uses; and, the reforestation of riparian corridors and abandoned lands (Daugherty, 2005). The underlying aim of these objectives is to increase the amount of forested area in the corridor so as to provide habitat, resources, or dispersal routes for native biota and to protect water and soil resources. As such, what the forest cover and land use data are suggesting is that the Alexander Skutch Biological Corridor is not mitigating the loss of forest and habitat degradation in an effective-enough manner.

That being said, many positive environmental changes have occurred in the corridor in the past decade. Although data results from the present study have revealed more negative ecological changes than positive, the forest cover maps (Maps 4.1, 4.2, and 4.9) show that a number of regenerating forest patches in the corridor have reduced the degree of isolation between some forest patches. Combined with the fact that these patches are linked to the large forested areas in the south, the increased connectivity in this part of the corridor may improve the probability of persistence for native species, particularly large-bodied mammals. In addition, small-scale reforestation efforts in the towns of Santa Elena and Quizarrá by community organizations such as Cocoforest and by student groups were noted by 52% of interview respondents. The World View 1 satellite imagery also shows that more shade trees have been planted on the El Grano Tico and Finca Bernina coffee plantations since 2005, when a set of aerial photos were taken of the corridor. Recent fundraising efforts by York University students have resulted in the purchase of a 2 ha protected area near Los Cusingos, intended to provide refuge for the endangered Central American Squirrel Monkey.

York University's involvement with the Fisher Fund for Neotropical Conservation has also enabled a great deal of diverse research projects and locally-based training to occur in the ASBC, including environmental education in schools, ecotourism surveys, water quality analyses, and baseline ecological and biodiversity data. The Las Nubes Carbon Fund, an avoided deforestation carbon offset program, is also currently being developed for the ASBC. Combined with the research produced from the three-year Avian Diversity Monitoring Program, the amount and breadth of research that has taken place in the ASBC has immense value for the sustainable management of the corridor. Essentially, many of the fundamental tools and knowledge exist to bring the ASBC closer to reaching its long-term goals.

Nevertheless, if current land use trends continue, further losses of forest, and by extension ecological integrity, can be expected. Considering that many species are already at the limits of their geographical and altitudinal ranges due to the three Holdridge life zones present in the corridor, further losses of forest may induce irrevocable changes in regional biodiversity levels. The increasing recognition that forested areas are essential for soil and watershed protection, erosion control, timber sources, and for ecotourism means that the social and economic sustainability of the corridor are intimately tied to the existence of substantial forest areas. Thus, it is imperative that appropriate action be taken to mitigate the degradation of ecological integrity in the ASBC and to restore key habitat areas.

Under this direction, locally-based organizations such as Cocoforest, COBAS (an association made up of representatives from the TSC, local communities, and Costa Rica's Ministry of Agriculture and Environment), Ture-COBAS (a community ecotourism association), and Asocuenca (a group devoted to the sustainable management of watershed lands) will be central in encouraging the wise use of lands within the corridor, and for disseminating educational information to communities to promote environmental awareness, including Costa Rica's PES program. Larger organizations such as the local farmer's cooperative, CoopeAgri and the TSC will also have key roles to play in the sustainable management of agricultural lands and protection of forests in the corridor. Finally, contributions from student or other researchers will be needed to strengthen the knowledge of ecological conditions in the ASBC as they relate to biodiversity monitoring and protection, as well as the further examination of opportunities for ecological restoration.

In short, any effort to protect, enhance, or restore ecological conditions in the ASBC cannot take place without the cooperation and communication of a diverse set of organizations and stakeholders.

There exists a responsibility for everyone involved, no matter race, creed, education level, wealth, occupation or age, to do what they can to help the Alexander Skutch Biological Corridor fulfill its namesake.

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Appendix A: Field Observations

Date:______________ Site Number:____________

Time:_______________ Weather: ________________

Location Information

GPS Coordinates: N Compass Bearing:

W

Elevation Error Range:

General Site Description:

Sketch of Site

Qualitative Information

Predominant land use (circle one)

Pasture Mixed Use Sugar cane

Pineapple Agriculture (other): ______________________

Approximate crop height (m): __________

Coffee: <10% shade cover

10-30% shade cover

> 30% shade cover

Forest: Regenerating (< 50 years; young forest class)

Secondary (< 50 years)

Primary/Mature (> 50 years)

Other: ____

Appendix B: Interview Questions

Date: ______________

Questionnaire: Land use in the Alexander
Skutch Biological Corridor

Please choose the option that applies to you:

Gender: Male/Female   Age: <25 25-35 35-45 45-55 55+

Education Level: Some elementary   Completed elementary

Some high school   Completed high school

Some college/university   College/university diploma

Annual income Level (USD): < $1,999 $2,000 – 3,999

$4,000 – 5,999 $6,000 – 7,999

$8,000 – 9,999 $10,000 – 15,000

> $15,000

Please take a few moments to answer the following questions:

1. How long have you lived in the Alexander Skutch Biological Corridor area?

2. In which town do you currently live? (please refer to the map if necessary).

   a) Have you always lived in the same town? If not, please list the town(s) you have lived in.

3. How many people currently live in your household?

4. What is your current occupation?

5. Who is the primary decision-maker in your household?

6. Do you or your family own or manage any land in the corridor? If so, how is the land currently used? (for example, cattle ranching, sugar cane, coffee etc.)

   a) Has the land always been used in this way? If not, how was the land used in the past?

   b) What are the future plans for the land, if any?

   c) Why has the land use changed? Or, if it has not changed, what is your motivation for maintaining the current use?

   d) Is the income you receive from your land as you expected?

7. How familiar are you with the types of land uses occurring in the corridor?
↑
   ____ Not familiar ____ Somewhat familiar ____ Very familiar

8. How would you describe how the land in the corridor has changed in over the past ten years?

   _____ Very little has changed

   ______ Some things have changed

   ______ A lot has changed

9. If you have noticed some changes, what kind of changes are they?

10. Why do you think these changes have occurred?

11. Do you know of any areas in the corridor that have been converted back to forest within the past 10 years? For example, newly reforested areas?

____ Yes ______ No ____ I'm not sure

12. If "yes" please describe, if you can, the general location of these lands. Use the map if necessary.

13. Do you know of any forest areas in the corridor that have been cut in the past 10 years?

____ Yes ____ No ____ I'm not sure

14. If "yes", please describe, if you can, the general location of these lands. Use the map if necessary.

15. Have you heard of the Payment for Environmental Services (PES) program offered by the government of Costa Rica?

    a) If so, do you think this program offers a good alternative to the types of land uses that are currently in the corridor?

    b) Do you foresee any challenges/obstacles to participating in this program?

Is there anything else you would like to add about land use changes or forested areas in the corridor?

Thank you very much for your participation!

Appendix C: FRAGSTATS Data Results

Class-Level Data

Metric	Forest Cover 1998	Forest Cover 2008
CA (ha)	2119.19	1716.19
NP	23	22
AREA_MN (ha)	92.14	78.01
TCA (ha)	1170.03	704.12
CPLAND	55.20%	40.00%
CORE_MN (ha)	50.87	32.01
CAI_MN	9.85%	6.63%
ENN_MN (m)	144.31	64.50
ENN_RA (m)	269.17	149.71
CLUMPY	-1.00	-0.16
CONNECT	39.13%	72.73%

(See section 3.7 in *Methods* for a description of each metric)