Road Impact on Deforestation and Jaguar Habitat Loss in the Mayan Forest

Conde, Dalia Amor

Publication date:
2008

Document version
Accepted author manuscript

Citation for pulished version (APA):
Conde, D. A. (2008). Road Impact on Deforestation and Jaguar Habitat Loss in the Mayan Forest.

ROAD IMPACT ON DEFORESTATION AND JAGUAR HABITAT LOSS IN THE

MAYAN FOREST

by

Dalia Amor Conde Ovando

University Program in Ecology
Duke University

Date:_______________________
Approved:

____
Norman L. Christensen, Supervisor

____
Alexander Pfaff

____
Dean L. Urban

____
Randall A. Kramer

Dissertation submitted in partial fulfillment of
the requirements for the degree of Doctor
of Philosophy in the University Program
in Ecology in the Graduate School
of Duke University

2008

ABSTRACT

ROAD IMPACT ON DEFORESTATION AND JAGUAR HABITAT LOSS IN THE MAYAN FOREST

by

Dalia Amor Conde Ovando

University Program in Ecology
Duke University

Date:_______________________
Approved:

____
Norman L. Christensen, Supervisor

____
Alexander Pfaff

____
Dean L. Urban

____
Randall A. Kramer

An abstract of a dissertation submitted in partial
fulfillment of the requirements for the degree
of Doctor of Philosophy in the University Program
in Ecology in the Graduate School
of Duke University

2008

Abstract

The construction of roads, either as an economic tool or as necessity for the implementation of other infrastructure projects is increasing in the tropical forest worldwide. However, roads are one of the main deforestation drivers in the tropics. In this study we analyzed the impact of road investments on both deforestation and jaguar habitat loss, in the Mayan Forest. As well we used these results to forecast the impact of two road investments planned in the region. Our results show that roads are the single deforestation driver in low developed areas, whether many other drivers play and important role in high developed areas. In the short term, the impact of a road in a low developed area is lower than in a road in a high developed area, which could be the result of the lag effect between road construction and forest colonization. This is consistent since roads resulted to be a significant deforestation driver for at least two decades. Roads significantly affect jaguar's habitat selection; however males showed a higher tolerance than females. From 1980 to 2000 female jaguars lost 36% of their habitat wile males lost 22%. Our forecasting of the impact of the proposed road, shows that it will promote the deforestation of approximately 16,851 has, and the jaguar habitat loss of 146,929, during the first decade; meanwhile the alternative route will have and impact of 2519 hectares and the habitat loss of 899 hectares.

Contents

List of Tables

List of Figures

Acknowledgments

Special thanks to Fernando Colchero for his support in all the different parts of this project. Song Quian for his important advice on statistical issues. Victor Hugo Ramos (CEMEC-WCS Petén) for his support on data acquisition for Guatemala and Belize. To Carlos Manterola for leading the Jaguar Conservation Program and the Mayan project for providing full support in al the faces of this project. Thanks to Cuahutemoc Chavez, Danae Azuara, Heliot Zarza, and Gerardo Ceballos for their support and important collaboration in the Jaguar section of this study. This project would not have been possible without the leadership in the field of Antonio Rivera and the expertise of Francisco Savala. Thanks to the Mexican Ministry of Transportation (SCT) for providing key data on dates of roads construction and pavement. The financial support for this project was provided by the Mesoamerican Biological Corridor Mexico (World Bank), Unidos para la Conservacion A. C., El Centro para el Conocimiento y Uso de la Biodiversidad (CONABIO, Gant #BJ006), Conservation International, programa: Semillas para la Conservacion Mexico (Grant 2005 (CI)), and Conservation Strategy Found (CSF). Probatura Península de Yucatán supported this Project with data and logistics, we are specially grateful to Maria Andrade and Gerardo Garcia. Thanks to Defensores de la Naturaleza, ProPeten, and Fundacion ARCAS for their support in the Guatemala Jaguar research, and to WCS-Guatemala for supporting us with equipment during the hard field seasons. Thanks to the members from Belize, Guatemala and

Mexico of the “Jaguar without Borders” initiative, for their comments that had made this project and applied one. Thanks to Tropico Verde and Parks watch for their help and support on the roads project. We would like to give special acknowledgements to the Wells family and to WINGS WORLDQUEST for their support.

1. Introduction

Tropical forest clearing accounts for roughly 20% of anthropogenic carbon emissions (IPCC 2007). However, they are a major target of infrastructure developments for oil exploitation, logging concessions or hydropower dam construction, among others, which inevitably conveys the expansion of the road network and the construction of roads in pristine areas. Roads have been found to be one of the most robust predictors of tropical deforestation (reviewed by Kaimowitz and Angelsen 1998), therefore the development of these infrastructure projects are of worldwide concern. Furthermore tropical forests support about two thirds of all know species and contain 65% of the world's 10,000 endangered species (National Research Council, 1980). They have been classified as biodiversity hotspots due to their high percentage of endemic species and the high threat of habitat loss (Myers 2000). Paradoxically to its key role as climate regulators and biodiversity hotspots, tropical forest has been seen as unproductive land, where governmental policies promote colonization as a way to alleviate the pressure from agrarian disputes, or as a rapid solution to allocate displaced communities due to environmental degradation or violence. Tropical forests are one of the last frontiers for the most vulnerable people worldwide in the search for subsistence land. Millions of people living in tropical forest exist on less than $1 dollar a day, and a third of a billion are estimated to be foreign settlers (Myers 1992). However, as the land degrades people are forced to migrate, exploring new forest frontiers and as a result deforestation increases. Whether supported or not by governmental programs, these settlers usually colonize the forest by using logging trails or new roads to access the forest for subsistence land (Amor 2007, Wilkie 2000).

Road construction is associated with an increase in forest impoverishment through logging and understory fire (Nepstad, et al. 2001). Roads increase the ability of people to reach desired locations in their search of subsistence land, promoting migration and the establishment of new villages in the forest frontiers (Nepstad, et al. 2001; Verburg, et al. 2004), which in many cases result in the subsequent displacement of small-scale farms by larger agricultural and livestock operations (Nepstad, et al. 2001). However, deforestation is just one of the impacts of roads on tropical forest. Road infrastructure and the access that it provides degrade forest ecosystems, affecting species habitat quality by increasing edge habitats (Gullison and Hardner 1993; Malcolm and Ray 2000). Roads as well provide access for hunting. In some cases road function as barriers for many species, subdividing populations and producing negative demographic effects (Forman and Alexander 1998).

Given these impacts of road development on tropical forest, it is essential to understand the impact of road investments on both tropical forest deforestation and degradation. In this document we analyzed the recent historical impact of road investments on both deforestation and forest degradation in the Mayan Forest.

The Mayan forest and the PPP

The Mayan forest is the second largest patch of tropical forest in the Americas after the Amazon, and it is at the intersection of three countries Mexico, Belize and Guatemala. The maintenance of this forest is of high concern since it is the focus of a major road development initiative (Amor et al. 2007). The expansion of the road network in the Mayan Forest is part of a regional development plan that covers Central America and the south of Mexico. This plan entails the construction of an international corridor of Mesoamerican Roads (RICAM), which exceeds more than 10,209 km, and sums and investment of around $5905 million dollars (SIEPAC, 2004). As well, this plan involves the development of a electric interconnection system that features a transmission line of 380 km from Panama to Guatemala, and the development of about 381 dams (hydro-electric) in the region (BID 2003; BID 2003; CFE 2004; Burgués 2005). Its main objective is to alleviate the high poverty levels, social inequality and economic underdevelopment of Mexico's south and Central America, by promoting trade between the involved countries and the USA (Economist 2008). However there has been a major concern on the impact of these projects on the local communities' economy, culture and natural resources (NFN 1997; HEED 2000; Miller, Chang et al. 2001). This mega infrastructure project originated from the Plan Puebla Panama (PPP), which was intended to extend from the Darien, in Panama, to Puebla in Mexico (SIECA 2004). However, nowadays it has been proposed to extend it to Colombia. The targeted region of the Plan Puebla

Panama has been along the Mesoamerican Hotspot which is one of the world's top Biodiversity Hotspots, harboring around 7% of the world's species, having lost up to 70% of its original area (Myers 2000).

Despite national and international conservation strategies in the PPP region, deforestation rates have steadily increased since the 1970s, giving Central America and Mexico the highest deforestation rates in the Western Hemisphere (FAO 2005). Moreover the development of the PPP projects will severely impact the remnant forest which are an important buffer against hurricanes and floods in the region and the principal source of goods and services for many local communities. Given the strong implications of the PPP projects on the region, we analyzed the impact of road investments located during the periods 1980-1990and 1990-2000 on Mayan Forest deforestation. As a proxy for the roads impact on the forest degradation we estimated the habitat loss of jaguar habitat for these same periods. We used the resulted models of both deforestation and jaguar habitat loss to forecast the impact of the "The Chetumal-Guatemala International Road", which is the largest road investment proposed in the region (Mena-Rivero 2004). We compared the impact this road construction with the impact of the improvement of the existing route. Using these data and analyses, we provide some recommendations for policy makers on the impact of two alternative road investments. To cover these three topics we organized this study in four following chapters.

Description of the chapters

Chapter II

In this chapter we analyzed the lag effect of road impact on deforestation and how the context of previous road investments change the effect of new road investments on deforestation. The lag effect of roads on deforestation provides an understanding of how the magnitude of deforestation temporally changes. For example if a road is placed by 1980, will it still have an impact on 1990 and on 2000 deforestation? And, if it does do we expect this impact to increase or decrease? One advantage of this type of analysis is that allows us to test whether road investments have a lag effect on the region's deforestation and how its magnitude varies across decades. The second advantage is that by analyzing these investments separately by periods, we can control for the time of the road placement and the time of deforestation. Otherwise, we could not determine what happened first, the deforestation or the road investment. In this chapter we also modeled how the development context of where roads are placed changes or not its impact on deforestation. In this sense, we wanted to know if in the short term a road placed in an already developed area is among many other drivers of deforestation, such as soil, markets distance, etc. Whether a road placed in area where roads have not been developed may be the only significant driver of deforestation, since access at this point is the main driver. As well, we compared how the magnitude of deforestation changes between a road constructed in a low or non-development area (i.e., far from existing roads) and a road constructed in an already-developed area. In this chapter we also assessed how the context of development and the impact of roads vary across countries, and how deforestation is associated with the early history of Mexico, Belize and Guatemala.

Chapter III

The main objective of this chapter is to assess the impact of road investments on jaguar habitat as a surrogate of forest degradation. We focused on this species since it fulfills most of the criteria of an umbrella, keystone, and flagship species for biodiversity conservation planning (Gomez et al. 2004). Jaguars can be considered as a keystone species since they have a significant influence on the ecosystem by regulating the population dynamics of a large number of prey species. Therefore, a reduction in their numbers will most likely affect the entire system (Mills et al. 1993). Jaguar habitat includes the habitat of a large number of plant and animal species habitats (Nunez, Miller et al. 2000). Therefore, by conserving jaguars habitat we are ensuring the conservation of many other species habitat, which makes the jaguar as one of the optimum umbrella species for large-scale conservation planning (Wikramanayake, Dinerstein et al. 1998; Coppolillo, Gomez et al. 2004). Due to their charismatic nature jaguars are considered a flagship species for conservation purposes.

In this chapter we modeled jaguar habitat in the Mayan forest to determine the impact of roads on jaguar habitat selection. We also determined whether road impacts varied with jaguar gender. We used the results of this model reconstruct the early history of jaguar habitat loss in the Mayan Forest, and we were able to give a proxy of the forest degradation in the region.

Chapter IV

In this chapter we develop we forecast the effects of the road investments on deforestation and forest degradation, testing the importance of including country variables as a surrogate of national policies that promote migration and settlement in the forest frontiers. Our analysis focuses on one of the major road development projects the Chetumal-Guatemala International (CGI) road, proposed by the PPP (RUCIA 2002) and the alternative existing route through Belize. We focused on the CGI road since is the biggest infrastructure project in the study are. This project is part of a commercial and tourism circuit that aims to connect the state of Quintana Roo in Mexico, with Guatemala and the rest of Central America. This road will facilitate the trade of goods between the Florida in the US and the Central America trough the Atlantic, while increasing the direct flow of tourists to the Mayan archeological sites. However, there is wide opposition to this road since it will bisect the Mayan Biosphere Reserve (MBR), which together with the protected areas of Calakmul and Balancan represents the largest patch of well preserved tropical forest of the Mesoamerican Hotspot.

We forecasted the impact of the two alternative road investments on both deforestation and jaguar habitat loss. Comparing both scenarios we determined which project will have the lowest effect on deforestation and jaguar habitat fragmentation and loss. With these results we analyzed the trade off that the policy planner faces in the region.

Chapter V

In this chapter we concluded the findings in each of the previous sections of this document and their importance in the context of conservation and development in the tropical forest, not only in the American continent but worldwide.

2. Road investments and deforestation of the Mayan forest

Introduction

Roads are widely described as one of the most important predictors of frontier expansion and deforestation in tropical forest regions, across a range of land dynamics. Though they are widely studied, careful documentation of the magnitude of roads' impacts is in fact relatively scarce. Not infrequently, a simple aerial or satellite snapshot is used to 'document' the impacts of reduced transport costs. More generally, average cross-sectional correlations of forest amount and road density are offered but are insufficient to demonstrate a causal link or to establish its magnitude. Further, even if causality and magnitudes were well established, it would be helpful to know how deforestation and related consequent impacts vary with the context in which the road investment is made. This is especially important within the design of development policies for which the balance of development and degradation outcomes is an imperative issue.

In this chapter we suggest, and demonstrate, that to address these questions it is exceptionally helpful to be able to track the sequence of road investments underlying the most recent available road map as well as the sequence of clearing underlying the latest forest map. Without knowing the timing of road investments and clearing, we can misrepresent the direction of causality between them. For example, an early unpaved road may increase access to the forest frontiers and, thereby, increase the deforestation rates. This, in turn, may affect further investments and decisions such as by local governments to provide various services, such as the pavement of unpaved roads and in areas were people has already settle and deforestation has already taken place. A decade or two later, cross-sectionally linking the paved roads as causes of nearby deforestation may misrepresent paving's impact.

Recent studies in the Brazilian Amazon have focused incorporating measures of road and forest change to avoid such errors. These studies, looking at locations receiving road investments (Pfaff, Walker et al. 2007) and also at their neighbors who do not receive road investments (Pfaff, Robalino et al. 2007) refute the suggestion in Andersen et al. 2002 that new roads will lower rates of deforestation in a county. The cited papers show that deforestation rises not only in census tracts which receive roads but also in nearby tracts in the same county without roads investments. They include fixed effects for the counties used in Andersen et al. (2002), since the census tract data they employ provides over 20 times (roughly 6000 vs. 3000) the observations.

Andersen et al. (2002) also appropriately regress deforestation, or forest change, on prior changes in roads. How, then, do they arrive at such a prediction given the above? The answer is the combination of a good idea with the limited data. The good idea was to analyze how road impact differs as a function of context, in particular prior clearing. Estimating an interaction, using the county data, suggested that more prior clearing led to lower road impact. Extrapolated, this suggested that new roads could lower clearing.

Pfaff, Walker et al. 2007 reevaluated these results by examining the much more numerous census tract observations in groups distinguished by level of prior clearing (0% , 1-50%, 51-75%, >75%). The dominant first two categories show strong increases in deforestation from roads investments. And while the last category is insignificant (with fewer observations), for 50%-75% prior clearing the increase in deforestation resulting from new roads investments is higher, not lower, than it is for the more pristine areas.

Such an outcome could arise because of the costs, and hence non-instantaneous pace of adjustment, on the frontier. When a road enters a previously less developed or pristine area, the labor and capital required to carry out all of the land-cover change that may suddenly be economically worthwhile are not present. Clearing primary forest is hard work, especially on ones own small land holdings. In contrast, in locations where some economic activity and forest clearing have already occurred, a rise in profitability due to a change in transport cost may more quickly be responded to and thereby may generate more additional deforestation in the first decade after the new road investment.

This would not mean that in the long-run new clearing versus baseline is higher. Yet based on these results alone, a decision maker could conclude that new roads into pristine areas will promote less additional deforestation than new roads following paths of prior development. Here, though, we see another value of observing the sequence of roads investment over time. Pfaff et al. 2006, for instance, show that new roads through a given site lead to follow-on investments in roads (such as paving of unpaved roads). Thus, it could easily be the case that entering the pristine area with a new road creates more additional deforestation over time than investment that follows upon past roads.

Such a perspective is relevant for current comment upon such famous ongoing policy initiatives as the "Avanca Brasil" program and the "Mesoamerican Road Interconnection Program (RICAM)" proposal. Each suggests the expansion of a road network within a region featuring a mix of developed and quite pristine areas.

To enhance understanding of road impacts across diverse landscape contexts, we take a similar approach to the Mayan forest (noting, given the importance of data resolution above, that here we use pixel data). The Selva Maya is an important tropical forest, the second biggest in the Americas after the Amazon and the largest continuous forest patch of the 'Mesoamerican hotspot' which contains around 7% of the world species. Located across Mexico, Belize and Guatemala, Selva Maya is subject to different policy, cultural and historical influences and to a grand road expansion program that will intersect its core. Given its biological importance and the environmental services it provides at a local and global scale, this region is a good case to consider road impacts.

We focus on four questions: 1) what are the short and medium term effects of paved and unpaved roads investments on deforestation?; 2) do these impacts differ when roads are placed in areas with existing pressure vs. in less developed locations?;

3) do the effect of non-road drivers also vary with development contexts? We might expect that roads in previously pristine areas a new road will be the dominant predictor; and 4) using a different measure of context, do road impacts vary across the countries? For example, countries with high subsidies in agriculture may experience more impact.

Methods

Study Area and Data Processing

The study area comprises the majority of the Petén of Guatemala, most of Belize with the exception of the Toledo district in the south, and a large portion of the states of Campeche and Quintana Roo in Mexico. The total area covers around 100,000 km2 which is delimited by four LANDSAT satellite images (Figure1). The region has tropical semi-deciduous forest, with an average of 1350 mm of annual rainfall and a pronounced dry season between February and June.

To remotely sense deforestation we used LANDSAT images in three time periods: (1) pre-1980, (2) 1980-1990, and (3) 1990-2000 (Table 1). For the base year (1980), we did a mosaic of four images from 1974 to 1980 to obtain a low cloud-free composite image of the study area. Likewise, the image dates that form the composite for the 1990 image ranged from 1988 to 1990. Sufficient cloud-free images from 2000 were available for that year's image mosaic. To equalize the resolution of the entire dataset to that of the coarsest data (MSS), TM and ETM+ images were resampled to a pixel resolution of 60x80-m.

Table 1. Path and row of satellite images used to estimate deforestation.

Path & Row	Receptor / Satellite	Date of Image
20-47	MSS / Landsat 3	January 1978
19-47	MSS / Landsat 3	February 1978
20-48	MSS / Landsat 3	February 1974
19-48	MSS / Landsat 3	December 1980
20-47	TM / Landsat 5	April 1988
20-48	TM / Landsat 5	November 1988
19-47	TM / Landsat 5	November 1990
19-48	TM / Landsat 5	December 1989
20-47	ETM / Landsat 7	March 2000
20-48	ETM / Landsat 7	March 2000
19-47	ETM / Landsat 7	April 2000
19-48	ETM /Landsat 7	September 2000

Two approaches were used to map deforestation in the three periods. For the pre-1980 period, we classified each image independently into forest and non-forest using the Maximum-Likelihood (supervised) classification algorithm and then combined the resulting forest/non-forest maps into a single mosaic. Training data for these classifications were collected from locations for which the land cover was known for 1980. To map deforestation for the 1980-1990 and 1990-2000 periods, we subtracted Normalized Difference Vegetation Index (NDVI) images of each period's beginning date from the period's end-date NDVI and identified deforestation from the histogram of differences (Yuang et al. 1998). To avoid interpreting phenological changes as deforestation, we classified the 2000 image into forest and non-forest types by Maximum-Likelihood and removed forest pixels from the change maps. The classification of the 2000 image, based on in situ training data collected in 2003, had a Kappa statistic of 0.8369. Clouds and water were also removed from each year by Maximum-Likelihood.

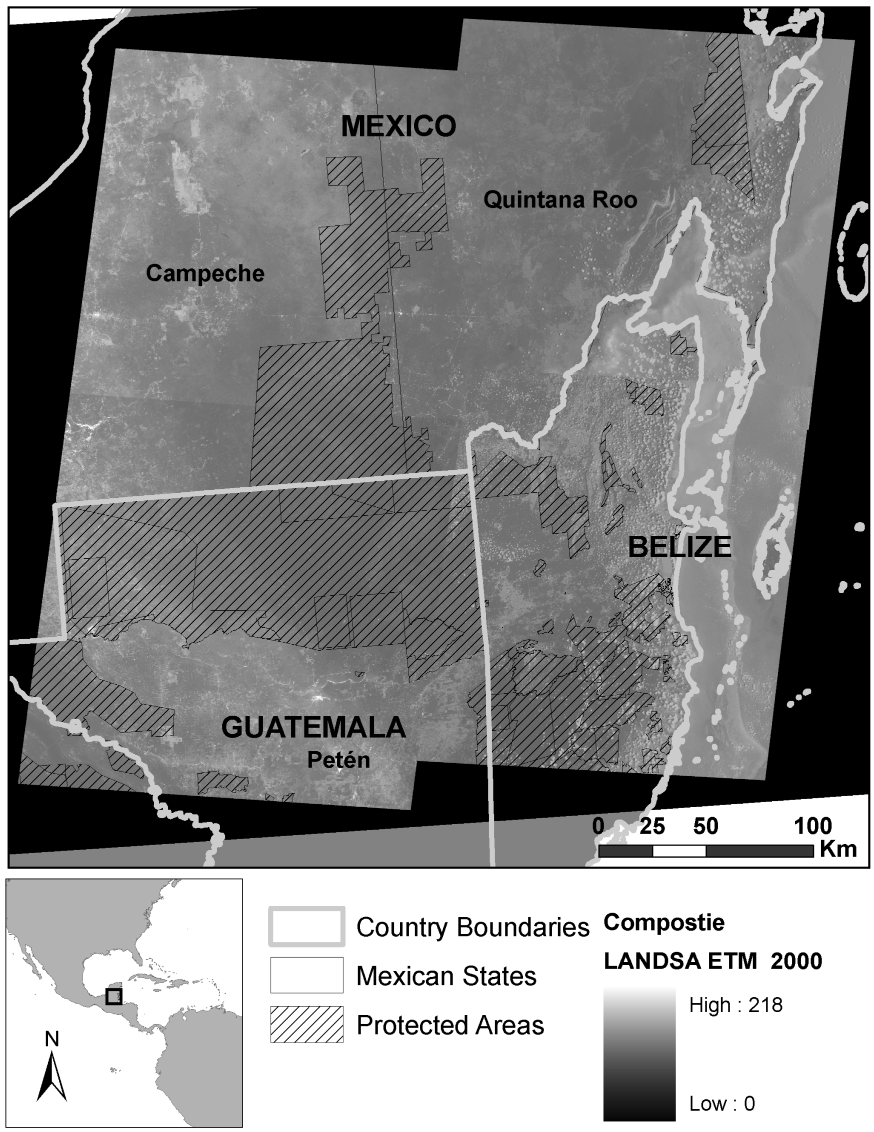

Figure 1 Study Area delimited by four LANDSAT images

The likelihood of forest clearing can be influenced by soil, elevation, slope, distance to previous deforestation and land tenure. We aggregated into four types a 24 types soil map classification from Garcia and Secaira (2006), based on soil characteristics and spatial continuity. For elevation, we used a 90m resolution digital elevation model . We calculated distance to deforestation for the pre-1980 and 1980-1990 periods, with the Euclidian distance algorithm of ARCmap ver. 9.1. We defined as "main markets" population centers that were present in the study area before 1980 and by 2000 had a population above eight thousand people. Small markets were defined as population centers that held between 2000-8000 people by the year 2000 and were present pre-1980. We computed the Euclidian distance to main and small markets. Protected areas boundaries for 1980-1990 and 1990-2000 were obtained from the CONABIO and Garcia and Secaira (2006) databases, no protected areas existed for the pre-1980 period. The country of Belize, the Petén of Guatemala and the states of Campeche and Quintana Roo in Mexico were defined as dummy variables. Mexico was divided in two states, due to their contrasting land use policy and history.

To track the evolution of road investments, we used a regional a road map for the year 2000 and we complemented it by digitalizing missing roads from the 2000 LANDSAT images. The García and Secaira (2006) roads layer has the attributes of each road as paved or unpaved. We obtained data on road pavement dates from CEMEC and Wildlife Conservation Society-Guatemala (WCS), as well as from the Mexican Ministry of Communication and Transportation (SCT). We digitize which segments were paved or unpaved for the pre-1980 and 1980-1990 periods by using the Landsat images, with these data we assigned to each road the paved or unpaved category. The two road types for three periods yielded to six variables (paved and unpaved per period: 80, 80-90 and 90-2000, Figure 2). For the six road variables, we computed the Euclidian distance. To analyze the impact of roads on deforestation in each time period, we assigned to each of the 15000 random sampling points, the distance to the closest road segment. This eliminated noise of considering roads that were really distant to the sampled point, since people access forests using the closest road to the target area; for example, if the target is within at 1km distance form road type A and in a 100km distance from road type B, people will more likely use road type A to access that forest parcel.

All the GIS layers were resampled to a pixel resolution of 10,000 square meters, which we defined as parcels; this resulted into a study area of 9,177,507 hectares. We defined this parcel unit based on the broader scale data layer a Digital Elevation Model (DEM, Shuttle Radar Topography Mission http://edc.usgs.gov/srtm/data/obtainingdata.html). We randomly sampled 15,000 parcels using Hawks' tools. We dropped the variables of transport cost distance to markets and roads density because it showed strong colinearity. The means and standard deviations of each variable can be seen in Table 2.

Table 2. Summary Statistics

Covariates	Mean or proportion for deforested parcels		
	By 1980	From 1980-1990	From 1990-2000
Distance to unpaved roads by 1980 (Km)	1.368	19.947	14.921
Distance to unpaved roads investments: 1980 to 1990 (Km)	–	2.99	4.107
Distance to unpaved roads investments: 1990 to 2000 (Km)	–	–	7.599
Distance to paved roads by 1980(Km)	8.687	22.544	32.911
Distance to paved roads investments: 1980 to 1990 (Km)	–	29.926	37.231
Distance to paved roads investments: 1990 to 2000 ,Km	–	–	15.36
Distance to Markets > 8000 people: type 1, Km	30.786	42.74	52.73
Distance to Markets > 2000 and <8000 people: type 2, Km	20.862	26.048	28.433
Elevation (m)	59	96	125
Distance to water sources (Km)	10.455	10.995	10.097
Protected areas in 1990 dummy	–	0	–
Protected areas in 2000 dummy	–	–	0.084
Distance to deforestation in 1980 (Km)	–	4.326	6.638
Distance to deforestation in 1990 (Km)	–	–	1.414
Campeche dummy	0.26	0.41	0.18
Quintana Roo dummy	0.25	0.31	0.27
Guatemala dummy	0.067	0.044	0.51
Belize dummy	0.43	0.13	0.038
Soil A dummy	0.62	0.54	0.64
Soil B dummy	0.19	0.31	0.16
Soil C dummy	0.003	0.044	0.1
Soil D dummy	0.0091	0	0.0026
Soil E dummy	0.0061	0	0
Soil F dummy	0.024	0.031	0.0013
Soil G dummy	0.024	0.019	0.01
Soil H dummy	0.024	0.0093	0.0064
Soil I dummy	0	0	0.0013
Soil J dummy	0.073	0.037	0.065
Soil K dummy	0.024	0.0031	0.0077
Total number of observations / No. of deforested	**15,432 / 332**	**15,139 / 324**	**14,917 / 790**

Model of Deforestation and Road Investments

According to economic theory the likelihood of a parcel of land being deforested will be higher if the profits of clearing a forest parcel are higher than the profits of leaving the land under forest cover. For this project we only focused on long-term land use change, therefore we did not quantify reforestation, only total forest loss. Based on previous research on deforestation drivers, we assumed that the likelihood of clearing a parcel will be influenced by distance to a road type investment (paved and unpaved) as well as the time of road placement. We assumed that the parcel characteristics that are likely to influence the profitability of deforestation are elevation, travel cost to the markets, soil type and protected areas status. We did not include slope since the area is a flat plateau with an average elevation of 300m.

To analyze temporal and individual effects of each type of road investment (paved/unpaved) we ran a separate model for each period that included the previous road investments, which resulted in six different models:

Impact of roads and other variables on 1980 deforestation

$y_{80} = \beta_0 + \beta R_{80} + \beta X$ …………… impact of 1980 paved roads (1)

$y_{80} = \beta_0 + \beta U_{80} + \beta X$ ……………..impact of 1980 unpaved roads (2)

Impact of roads and other variables on 1990 deforestation

$y_{90} = \beta_0 + \beta R_{80} + \beta R_{90} + \beta D_{80} + \beta Pa_{90} + \beta X$ …….impact of 1980, 80-90 paved roads (3)

$y_{90} = \beta_0 + \beta U_{80} + \beta U_{90} + \beta D_{80} + \beta Pa_{90} + \beta X$ …….impact of 1980, 80-90 unpaved roads (4)

Impact of roads and other variables on 2000 deforestation

$y_{00} = \beta_0 + \beta R_{80} + \beta R_{90} + \beta R_{00} + \beta D_{80} + \beta D_{90} + \beta Pa_{00} + \beta X$...impact of 1980, 80-90, 90-00 roads (5)

$y_{00} = \beta_0 + \beta U_{80} + \beta U_{90} + \beta U_{00} + \beta D_{80} + \beta D_{90} + \beta Pa_{00} + \beta X$..impact of 1980, 80-90, 90-00 roads (6)

Where *y* is the deforestation at time *t*; *R* is the distance to paved road investments placed at time *t*; *U* is the distance to unpaved roads investments placed at time *t*; *D* is the distance to deforestation that appeared at time *t*; *Pa* is a dummy variable for protected areas that were established at *t*; and ***X*** is a vector of the following covariates: elevation (m), dummy variable for country/state (Quintana Roo, Campeche, Guatemala and Belize); distance to main markets (markets 1); distance to small markets (markets 2); and a dummy variable for soil type.

As can be seen in the formulas (1-2), there is just one road type, that is, the roads that were present by 1980. Because these two models are for the first period analyzed (1980) we did not included the variable for distance to deforestation (*D*), since we do not have a previous period to measure the deforestation. The variable of Protected areas (*Pa*) is not included since no protected areas existed by 1980. However, in the models for

1990 deforestation (formulas 3-4) we have two road variables per model, one for the distance to roads present by 1980 ($R_{80,}$ U_{80}) and one for the distance to roads investments from 80-90 ($R_{90,}$ U_{90}). These models also included a variable of distance to deforestation, which controlled for the distance to the parcels deforested by 1980 (D_{80}). In addition, these two models included the variable for protected areas (PA_{90}). For the last period 1990-2000, we had three road variables per model (formulas 5-6): the first one for roads present by 1980, the second one for the distance to road investments from 1980-1990, and the third one for the road investments from 1990-2000. In this model we included two variables for distance to deforestation, one for 1980 (D_{80}) and the second for 1980-1990 (D_{90}) deforestation. This allowed differentiating the impact of each period of road investment on deforestation, which is essential to determine the temporal effect of each road type.

Since the road variable was decomposed by period, our model considered only the distance of the parcel (or sample point) to the closest road type. This avoided spurious conclusions on the relationship of each road variable type on deforestation. The advantage of using this method is that considers the mechanism or behavior of a peasant on accessing the land. For example, a farmer will not access a parcel from a road that it is a 100 km from the parcel if another road exists that is only at 1 km of distance. By only using the distance of the closest road to each parcel, we eliminated this source of error that can result in spurious coefficients. If we do not consider the closest road type then we can expect to obtain even a wrong relationship (sign) between the road and deforestation. Coming back to our example, the road (a) is 100km from a parcel and a road (B) is 1 km from the same parcel. The parcel is one hectare of deforestation that occurred from 1990-2000; road (a) is an investment that was placed by 1980, and road (b) is an investment from 1980-1990. If we consider both distances to model the deforestation in the year 2000, then road (a) will show a positive coefficient; implying that the farther you are from the road the likelihood of deforestation increases for a 1980 road, whether for the road (b) will show a negative value, since the likelihood of deforestation decreases as the distance to this road increases.

To understand the effects of roads given the landscape context, we divided our sample in two development categories: 1) high developed areas, and 2) low development or "pristine" areas. High developed areas included all the points located within 25Km of past road investments; low development areas included all the points situated beyond the 25Km distance. For this model we did not differentiate between paved and unpaved roads, since this considerable reduced our sample size. We modeled separately the impact of roads in high developed areas from low development areas. To verify the robustness of our results, we re-analyzed the data using distance to previous deforestation instead of distance for roads, and we varied the distance threshold from 10km to 50 Km.

We expect that the variables of road distance, markets distance and deforestation distance will have a negative coefficient. Since the likelihood of the deforestation is expected to decrease the farther the parcel is from a road, a market or an already deforested parcel. As well, elevation is expected to be negative, since higher elevations may have a lower probability of deforestation. The correlation with protected areas should also be negative, because the protected status may promote lower deforestation. In the case of country variable, we expect a different sign given the period. For 1980, Guatemala and the states of Mexico (Quintana Roo and Campeche) may show a negative sign, while Belize may have a positive one. This is highly likely since during the 70's Belize received subsidies for sugar cane production, wile in the region Guatemala and Mexico lacked colonization programs or subsidies. However, in from 1980-1990 and for 1990-2000 Belize may show a negative sign, since its economy shifted towards tourism in the coast and it initiated its program of protected areas. Meanwhile, Mexico and Guatemala may be positive, since Mexico started a colonization program in the region and subsidies for agriculture increased from the 1990 to 2000. In the case of Guatemala, the increase of migration in the Petén during this period is well documented.

We used a Generalized Linear Model (GLM) approach to understand the probability of deforestation in a parcel given the covariates mentioned above. The outcome variable is deforestation (y_i = 1 if a parcel i is deforested, y_i = 0 if a parcel is not deforested). For pre-1980 we considered all the deforested cells in the landscape present at that time, without knowing the period when the deforestation took place. For the second period: 1980-1990, we excluded all the deforested parcels present in 1980, therefore y_i = 1 only for parcels that were deforested during that decade. We followed this same protocol for the third period: 1990-2000.

We modeled deforestation at each point i as a Bernoulli process (y_i ~ Bernoulli (pi)) with probability p_i to be deforested; we linked this probability with relevant covariates through a logit-link function of the form:

$$\ln\left(\frac{p_i}{1-p_i}\right) = \mathbf{x}_i\boldsymbol{\beta} \tag{7}$$

where $\mathbf{x}_i$ is the vector of covariates (i.e distance to road, elevation, country, soil type, etc.) for point i and β is the vector of parameters linking $\mathbf{x}_i$ and p_i.

Although the covariates that we used had an inherent spatial structure, we tested for the effect of spatial autocorrelation by including in the GLM an autocovariate term. This means that the model takes the form:

$$\ln\left(\frac{p_i}{1-p_i}\right) = \mathbf{x}_i\boldsymbol{\beta} + \alpha A_i \qquad (8)$$

where α is a parameter for the autocovariate term A_i. This last is calculated as

$$A_i = \frac{\sum_{j=1}^{k_i} w_{i,j}\hat{p}_j}{\sum_{j=1}^{k_i} w_{i,j}} \qquad (9)$$

and is a weighted average of the number of points k_i in a radius of 5 km around point i, each with a weight $w_{i,j}$, which is equivalent to $w_{i,j} = 1/h_{i,j}$ where $h_{i,j}$ is the Euclidean distance between points i and j. The parameter $\hat{p}_j$ represents the estimated probability of deforestation for each point j. If there is a spatial effect that is not controlled by the covariates, it is expected that the distribution of parameter α will be significantly different from 0 (i.e. 0 will be outside the 95% credible intervals). Since the spatial autocorrelation term from this analysis was not significant, we did not include it our model.

Results and Discussion

Short and medium term effects of road investments on deforestation.

The results from the first six models (formulas 1-6) show that both paved and unpaved roads are significant deforestation drivers in the Mayan Forest. The only exception, was the coefficient for distance to paved investments in the 80-90 period, which were non significant for deforestation during the 80-90 period. However, for the following period (90-00), it was highly significant (Table 3). This lag effect of the 80-90 paved roads on subsequent deforestation is logical, since most of these paved roads were built by the end of the 1980s. As can be seen in Table 3, unpaved roads showed a consistent impact on deforestation that lasts for at least two decades. Unpaved roads built by 1980 were highly significant for the 1980s (with a coefficient of -0.37 and a standard error of 0.000) and for deforestation in the 1990s (with a coefficient of -0.24 and a standard error of 0.000). In this way, we show that even type of investment (paved and unpaved) in each period lowers transport costs on average in such a way as to increase rates of deforestation (with the caveat that for one period new paved investments have no significant impact). That sets the stage, then, for examining the context dependence of road impacts to shed light on the choices facing the policy planner.

Table 3. Log-regression coefficients of deforestation covariates resulted from the per road type, per period models.

	Unpaved by1980		Paved by 1980		Unpaved Investments 80-90		Paved Investments 80-90		Unpaved Investments 90-00		Paved Investments 90-00	
	Est.	**p-val**	**Est.**	**p-val**	**Est.**	**p-val**	**Est.**	**p-val**	**Est.**	**p-val**	**Est.**	**p-val**
DEFORESTATION present by 1980 (Total sampling units: 15,432 / Total deforested sampled units:332) 2.15% of deforestation initially												
Intercept	-0.37	0.07	-0.87	0.06								
Closest dist.	**-0.72**	**0**	**-0.87**	**0**								
Campeche	0.19	0.34	-0.41	0.47								
Quintana Roo	-0.55	0	-0.26	0.48								
Guatemala	-0.35	0.28	-1.24	0.14								
Elevation	0	0.01	0	0.5								
Dist to markets 1	-0.01	0.01	-0.01	0.06								
Dist to markets 2	-0.02	0	-0.01	0.06								
Soil dummy B	0.61	0.01	0.06	0.92								
Soil dummy C	-1.43	0.17	44.83	0.94								
Soil	0.03	0.85	0.2	0.65								

	Unpaved by1980		Paved by 1980		Unpaved Investments 80-90		Paved Investments 80-90		Unpaved Investments 90-00		Paved Investments 90-00	
	Est.	**p-val**	**Est.**	**p-val**	**Est.**	**p-val**	**Est.**	**p-val**	**Est.**	**p-val**	**Est.**	**p-val**
dummy D												
Sample size/def	*10937*	*275*	*4495*	*57*								
DEFORESTATION from 1980-1990 (Total sampling units:15,139 /deforested sampled units: 324) 2.15% of deforestation in a decade												
Intercept	-1.98	0	-1.71	0.05	-2.5	0	6.07	0.2				
Closest dist.	**-0.37**	**0**	**-0.11**	**0.6**	**-0.38**	**0**	**0.16**	**0.63**				
Campeche	0.7	0.03	1.36	0.09	2.71	0	-6.32	0.1				
Quintana Roo	-0.22	0.47	0.94	0.21	0.61	0.11	-7.65	0.07				
Guatemala	0	1	-16.1	0.99	1.8	0	-	-				
Protected Areas	-12.39	0.98	-14.07	1	-11.21	0.97	-	-				
Elevation	0	0.67	0.01	0.13	0	0.04	-0.06	0.04				
Dist. to def 80	0.01	0.86	-1.89	0.01	0	0.9	-0.74	0.19				
Dist to markets 1	-0.01	0.09	-0.03	0.01	-0.02	0	-0.01	0.84				
Dist to markets 2	-0.01	0.06	-0.04	0.04	-0.01	0.03	0.05	0.17				
Soil dummy	0.34	0.37	0.32	0.7	0.5	0.12	-	-				

	Unpaved by1980		Paved by 1980		Unpaved Investments 80-90		Paved Investments 80-90		Unpaved Investments 90-00		Paved Investments 90-00	
	Est.	**p-val**	**Est.**	**p-val**	**Est.**	**p-val**	**Est.**	**p-val**	**Est.**	**p-val**	**Est.**	**p-val**
B Soil dummy C	1.22	0.03	-	-	0.67	0.18	-0.04	0.98				
Soil dummy D	-0.09	0.76	-0.07	0.92	-0.12	0.7	0.66	0.64				
Sample size/def	*3296*	*97*	*1856*	*19*	*9597*	*200*	*390*	*8*				
DEFORESTATION 1990-2000 (Total sampling units:14,917 / deforested sampled unitis:790) 5.3 % of deforestation in a decade												
Intercept	-2.72	0	-1.74	0.01	-2.69	0	2.77	0.16	-4.49	0	-16.31	0.98
Closest dist.	**-0.24**	**0**	**-0.24**	**0.17**	**-0.17**	**0**	**-0.69**	**0.08**	**-0.25**	**0**	**-0.24**	**0**
Campeche	1.13	0	1.33	0.12	0.62	0.13	-3.66	0.02	2.4	0	14.59	0.99
Quintana Roo	1.15	0	1.23	0.06	0.91	0.01	-3.99	0.01	2	0	14.3	0.99
Guatemala	3.78	0	-14.27	0.99	2.4	0	-	-	3.85	0	16.23	0.98
Protected Areas	-15.54	0.98	-14.51	0.99	-0.84	0	-12.08	0.99	-1.34	0	-0.37	0.52
Elevation	-0.01	0	-0.01	0.13	0	0	-0.02	0.02	-0.01	0	-0.01	0.02
Dist. to def 80	0.08	0.15	-0.04	0.83	-0.03	0.02	-0.24	0.63	-0.02	0.16	-0.02	0.51
Dist. to def 80-	-0.4	0	-0.17	0.31	-0.15	0	0.04	0.89	-0.02	0.57	-0.2	0.1

	Unpaved by1980		Paved by 1980		Unpaved Investments 80-90		Paved Investments 80-90		Unpaved Investments 90-00		Paved Investments 90-00	
	Est.	p-val	Est.	p-val	Est.	p-val	Est.	p-val	Est.	p-val	Est.	p-val
90 Dist to markets 1	0.02	0	0.01	0.44	0.01	0	0.04	0.04	0.01	0.09	0.01	0.1
Dist to markets 2	-0.01	0.15	-0.02	0.03	-0.01	0	-0.02	0.31	0	0.43	0	0.65
Soil dummy B	-0.12	0.71	-0.79	0.17	-0.05	0.85	-0.68	0.67	-0.02	0.97	-0.31	0.54
Soil dummy C	0.13	0.81	-	-	1.11	0	-	-	1.41	0.01	-0.3	0.81
Soil dummy D	-0.04	0.88	-1.08	0.01	-0.05	0.83	-0.34	0.77	0.43	0.38	-0.15	0.72
Sample size/def	*1953*	*157*	*1284*	*38*	*5303*	*324*	*317*	*20*	*5083*	*180*	*977*	*71*

Differences of road impacts given the landscape context.

Our results show that the impact of roads investments on deforestation is highly dependent on the landscape context (Tables 4 & 5). In the short term, both distance to previous roads and to previous deforestation are important elements that directly affect the magnitude of deforestation promoted by new investments. Road impact was higher where prior development and clearing are likely to have occurred. Conversely, impact was lower in pristine areas when the prior distance to the closest road was relatively high (with a coefficient of -0.4 vs -0.23 for 1980s deforestation, Tables 4 & 5). These results were consistent even when we varied the cutoff dividing the samples to see how the 'developed vs. pristine' definition affects results.

Two results stand out from this examination. First roads are a central feature in the process of deforestation, since in low development areas the only deforestation driver are roads. Second, in the case of 1990-2000 roads placed in highly developed areas have a higher impact on deforestation than when roads are placed in previously less accessible areas (or pristine areas). This supports previous findings on roads impact in the Brazilian Amazon (Pfaff et al. 2007a & 2007b). However, for the 1980-1990 deforestation there was not a significant difference between the impact of roads placed in high developed areas versus low development areas (Figure 2).

Table 4. Results from the logit-models of 80-90 deforestation in high and low developed.

Output variable Deforestation from 1980 to 1990 (Y=1)						
Sample = points which in 1990 were closest to 80-90 unpaved investments						
"close" = <=25km	**High Development CLOSE from 1980 roads**			**Low Development FAR from 1980 roads**		
Covariates	**Coeff**	**SE**	**Sig**	**Coeff**	**SE**	**Sig**
(Intercept)	-2.54876	0.449714	***	-11.38	0.1074	
Unpaved roads 90-80	-0.44899	0.049386	***	-0.23	0.06	***
Campeche, Mexico	2.501916	0.381564	***	11.59	1.00	
Quintana Roo, Mexico	0.477885	0.388239		9.63	0.100	
Petén, Guatemala	2.065306	0.404955	***	9.51	0.020	
Protected Areas 80-90	-11.2285	0.33166		NA	NA	
Elevation (mts)	-0.00366	0.001459	*	-0.01	0.00	
Distance to def. in 1980 (Km)	-0.00174	0.019976		0.06	0.03	.
Distance to main markets	-0.01424	0.005142	**	-0.01	0.01	
Distance to small markets	-0.00841	0.006515		-0.02	0.02	
Soil B dummy	0.636963	0.329528	.	-1.20	1.25	
Soil C dummy	0.512129	0.543185		-0.18	1.49	
Soil D dummy	-0.11371	0.319694		-0.24	1.10	

Table 5. Results of the logit-models of 90-00 deforestation in high and low-developed areas.

Output variable Deforestation from 1990 to 2000 (Y=1)						
Sample = points which are closest to 90-00 unpaved investments						
"close" = <=25km	**High Development CLOSE from 80-90 roads**			**Low Development FAR from 80-90 roads**		
Covariates	**Coeff**	**SE**	**Sig**	**Coeff**	**SE**	**Sig**
(Intercept)	-4.485961	0.631553	***	-27.490	0.180	
Unpaved roads 90-00	-0.24483	0.039156	***	-0.189	0.048	***
Campeche, Mexico	2.39708	0.548414	***	9.451	0.680	
Quintana Roo, Mexico	1.996857	0.530345	***	8.988	0.680	
Petén, Guatemala	3.84759	0.535137	***	11.090	0.500	
Protected Areas 90-2000	-1.344767	0.28346	***	-0.926	0.378	*
Elevation (mts)	-0.005437	0.001655	**	-0.009	0.003	*
Distance to def. in 1980 (Km)	-0.017385	0.012517		-0.019	0.022	
Distance to def.80-90 (Km)	-0.017416	0.031051		-0.127	0.074	.
Distance to main markets (Km)	0.006851	0.003995	.	0.013	0.009	
Distance to small markets (Km)	-0.004189	0.005256		0.009	0.010	
Soil B dummy	-0.022048	0.530106		14.770	0.140	

Output variable Deforestation from 1990 to 2000 (Y=1)					
Sample = points which are closest to 90-00 unpaved investments					
"close" = <=25km	**High Development CLOSE from 80-90 roads**			**Low Development FAR from 80-90 roads**	
Soil C dummy	1.408547	0.545566	**	16.960	0.130
Soil D dummy	0.43429	0.490547		15.630	0.20

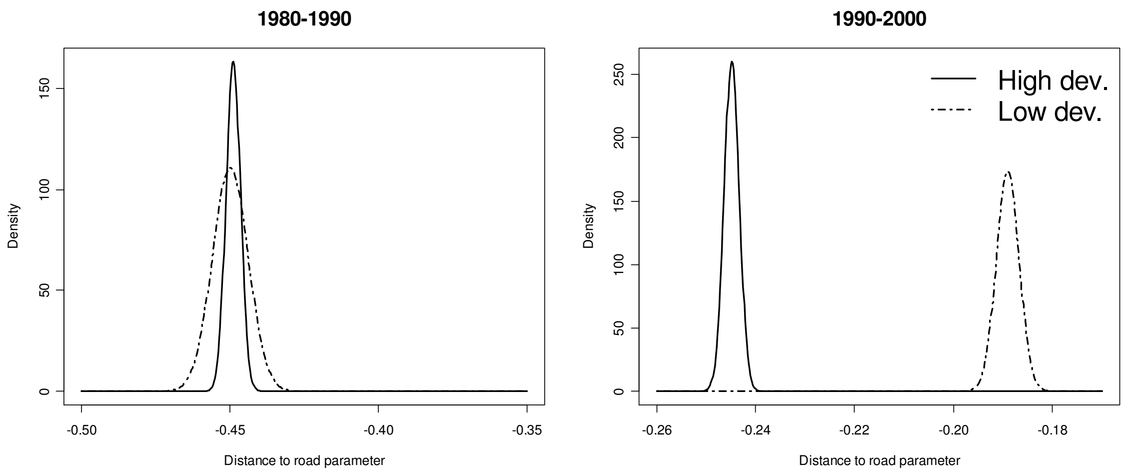

Figure 2 Density distributions of the distance to road parameter, for hig and low development areas, for both periods.

Differences of deforestation drivers impacts, other than roads given the landscape context.

Our results show that the only significant driver of deforestation is roads in areas with low development. Previous studies show that the benefits of clearing land for agriculture or cattle depends on the access to markets, the distance to roads, as well as on the biophysical conditions of the land such as soil quality and elevation. However, our results show the important role of roads on deforestation at the forest frontiers. In areas far from existing roads in 1980, the only high significant predictor of 1980 to 1990 deforestation was the distance to the roads built during this period (-0.23, Standard Error SE 0.06). The distance to cleared land in 1980 was correlated with low significance (0.06, SE 0.03, Tables 4 & 5), and all the other variables were non significant. For deforestation from 1990 to 2000, roads built in this period were the only highly significant variable (-0.189, SE 00.4). Protected areas and elevation were correlated with low significance (0.93, SE 0.38 and 0.009, SE 0.003). On the other hand, when development has already occurred in an area (existing roads and cleared land exist close to the new investments), not only the distance to the closet road but also a number of others factors thought to affect net benefits of land uses are highly significant predictors of deforestation rates; such as the country or state, protected areas, soil type and elevation (Tables 4 & 5). Our results stress the important role of roads as main deforestation drivers in the forest frontiers. Even if its immediate impact is lower than for the roads placed in already developed areas, these new roads investments are the ones that shape the future clearing and the development patterns in the forest. In the long term, we can expect that the impact of roads in these pristine areas will increase by promoting further development in the region.

The role of country/state on deforestation.

Our analysis shows the role of countries or states on deforestation. When roads are placed far from previous development the country/state covariates were not significant deforestation drivers for both 80-90 and 90-00 deforestation (Tables 4 & 5). However, when roads investments were placed in a developed area (close to other roads or previous deforestation) they were highly significant covariates. For 1980 to 1990 deforestation we can see that Campeche (2.5 with a S.E 0.39) and Petén (2.0 with a S.E 0.42, table 3) were significant and positive deforestation drivers, Campeche being slightly more significant than Petén. However, for 1990 to 2000 deforestation not only were Campeche (2.39 with a S.E 0.5) and Petén (3.84 with a S.E 0.53), significant, but also Quintana Roo (1.9 with a S.E 0.53, Table 5).

These results are consistent with what we would expect to be the indirect effects of national and state policies on deforestation. Since 1980, the discovery of oil in the state of Campeche promoted high immigration mainly to the coast; however, the rise in income in the state supported the conversion of tropical forest for the production of sugar cane and rice. During the same period, Guatemala suffered from the bust of cotton prices in the South Coast, which promoted the migration toward other countries, although while many of the migrants used the Petén as a transit area, some subsistence farms where established in the Petén. This was mainly when the FAR armed forces that were settled in the Petén during the conflict started their exile to Mexico in 1985. From 1985 to 1989 forest conversion for the cultivation of cannabis drastically increased from at least 225 to 1,220 hectares. This was in addition to the expansion of subsistence farms. On the other hand, Quintana Roo and Belize were not significant covariates for the 80-90 deforestation. Most of the investments in Quintana Roo were focused for the tourism industry in the Caribbean coast; and the subsidies in the forest were mainly focused on forestry management. In 1983 most of the ejidos, which are communal lands, that owned land in the forest, formed part of the Forestry Pilot Program. Their goal was to introduce a participative management of the forests with a sustainable harvest for timber and non-timber products. This big initiative from the Mexican and German government was able to support the conversion of 500,000 has of tropical forest into forestry ejido, belonging to five forestry societies. In the case of Belize from 1980 to 1990, there were few investments for agriculture or other land uses in the study area. Most of the land conversion was done for sugar cane production in the 1970s, which was mainly subsidized by England and exported to England (Bolland 1985). However, during the 80s, we can observe Belize was in the early years of its independence, and did not have strong policies for forest conversion, with the exception in 1986, when the government provided more than 15,000 acres to Belizean families and Salvadorian refuges, near to the capitol Belmopan. Most of the investments were focus on the coast for tourism (Anne 1998).

The country and state covariates for 1990 to 2000 deforestation as well reflect the effects of the country and state policies during that decade. This time, not only were Campeche and Petén significant predictors of deforestation, but also Quintana Roo (Table 5). In this case, the Petén showed the highest coefficient (3.84) and from our analysis on deforestation we can see that Petén had the highest proportion of deforestation (Figure 3). Although in 1990 the Mayan Biosphere Reserve (RBM) was created, during this decade different factors made the Petén the one of the main destines for migrants. In 1994, the repatriation process of the refuges from the armed conflict began and the government gave them land in the Petén. At the same time, the finding of oil in the north west of the RBM, promoted the investment of roads construction. Peasants from the South Coast continued to migrate to this region since the crash in cotton prices and other products left them without jobs, and those that could not migrate to the US were in search of subsistence land in the Petén (Grandia 1992). However, the land where peasants had established has subsequently been occupied by big land owners mainly for the development of cattle ranges. By 1999, around 50% of the owners of parcels had 92% of the land; this accelerated the invasion of landless peasants to the protected areas. By 1996, there were 41 illegal communities in the RBM which increased to 80 by 1990 (Clark 2000). This shows the indirect effects of national policies, since the Petén's coefficient raised from 2.0 to 3.8 in one decade, making it the state with the highest impact on deforestation for the region. Campeche continued with a similar impact than in the previous decade, the main investments targeted the agriculture and cattle ranging. In the case of Quintana Roo, during this decade, most of the support from the federal government to the Forestry Ejidos stopped, and subsidies to support Chile plantations started. It could be that the lack of support to forestry ejido was one of the triggers for forest conversion, during the 90s Quintana Roo is a significant driver of deforestation, even if it is lower than Campeche and the Guatemalan state of Petén. At the same time Belizean economy relayed in the ecotourism, not just for its beaches but as well in the rainforest, only in the Toledo District, was promoted the agriculture for the production of critics, however this district was not included in our study area.

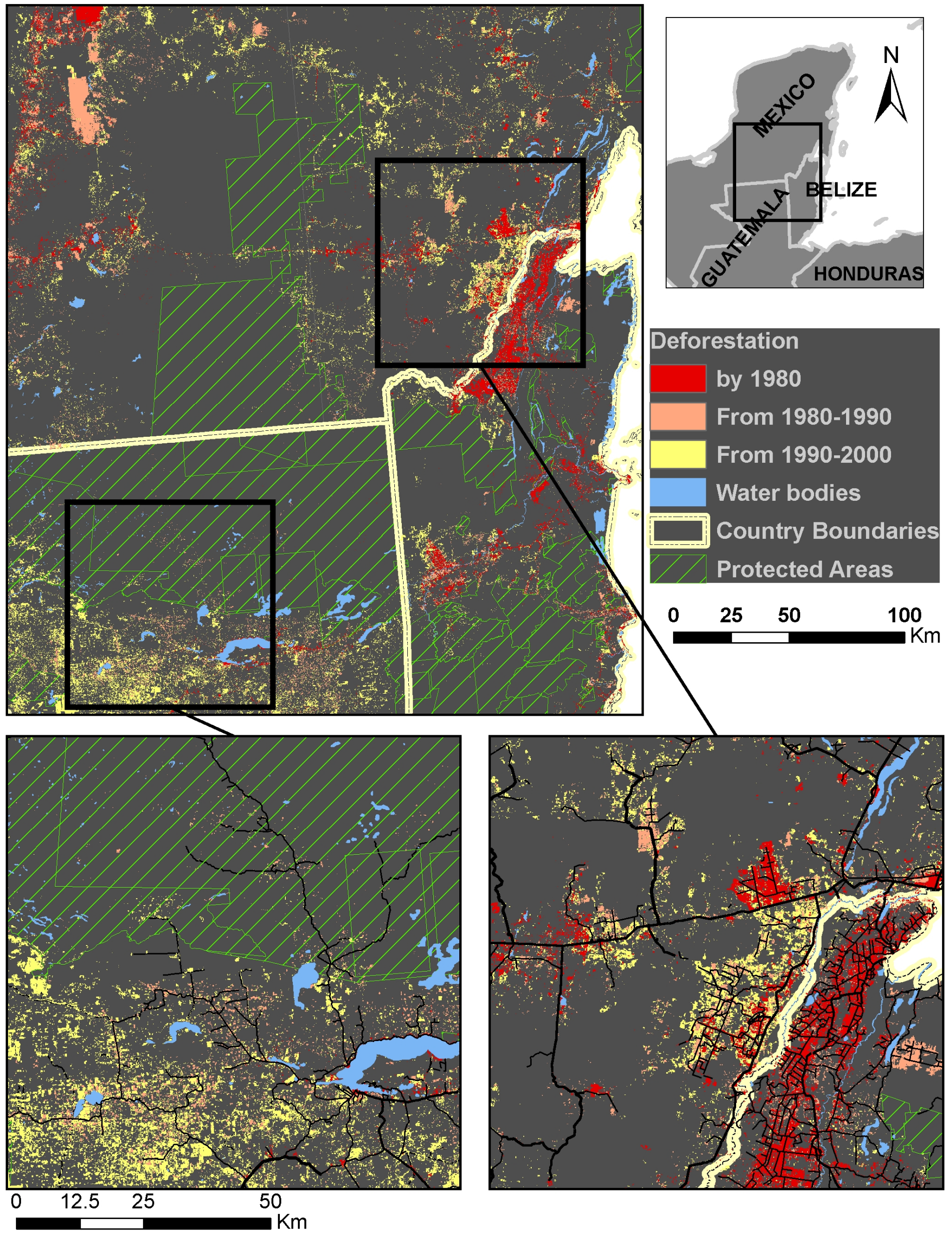

Figure 3 Deforestation in the study area by 1980, from 1980 to 1990 and from 1990 to 2000

What are the implications of these results for decision makers?

Understanding the impact of road investments on forest clearing is crucial for the design of development policies in tropical forests. The combination of our five results reflects the spatial and temporal tradeoffs facing a policy planner. A new road into a previously undeveloped area will be the determinant of the long-term future path of development and deforestation by shaping the new forest frontier, even if, in the short term, its magnitude is lower than a road placed in an already developed area. Therefore, the area affected is not cleared in the first decade at the same rate as paths of new roads located in the development trajectory where activity is already ongoing. Nevertheless, in the long term, we can expect that the impact of roads in the pristine areas will increase by promoting development in the region, which as a result will promote new roads by providing political and economic incentives for further investments. Consequently, because the roads into undeveloped areas very clearly determine that new paths of clearing arise that are likely to be followed by even more investment and deforestation. As well it is important to consider that the impacts will be different given the country or state of investment, however, we found that for the Selva Maya, when a road is placed in a low developed or pristine area the country or state effects seems to do not be significant. But in the long term it will definitely shape the expansion of the agriculture frontier.

A development planner that contemplates the conservation and management of tropical forests needs as well to consider road effects beyond its impacts on deforestation. Roads impact on habitat quality and fragmentation may play a key role as indicators of were to place a road. Although, as our results show that in the short term, a road may promote less deforestation in a pristine area than in a developing area, its impact on fragmentation of certain species may be higher. Further studies that include this type of analysis will be an important contribution to the literature on roads impact on tropical forests. To pursue this type of analysis and to understand the long-term effects of roads on deforestation will be essential for the proper long term management of tropical forests.

3. Using a jaguar habitat model to understand the early history of road impact on the Mayan forest degradation

Introduction

Road investments are a primary driver of deforestation responsible for much species habitat loss in tropical regions. However, deforestation is just one of the impacts or roads on tropical forest. Roads affect species habitat quality and increase habitat fragmentation (Spellerberg). Because satellite images provide and ideal tool to monitor deforestation, most of the studies of road impact on tropical forest have focused on deforestation alone (Chomitz and Gray 1996; Cropper, et al. 1999; Cropper, et al. 2001). Nevertheless, if we aim to conserve and manage tropical ecosystems it is important to understand in conjunction the implications of infrastructure investments in both tropical deforestation and forest degradation (Pfaff, et al. 2008). To assess the impact of road investments on the Mayan forest degradation we modeled the habitat the jaguar (*Panthera onca*), since it is an optimal umbrella species for conservation planning (Gomez et al. 2004). With the results of this model, we were able to reconstruct the early history of jaguar habitat loss in the Mayan Forest to understand the impact of road invested on the Mayan Forest degradation.

Road investments are usually proposed as an instrument to promote economic growth in rural areas by creating access to markets and improving accessibility for the extraction and exportation of natural resources (Riverson et al. 1991), although those investments are not always economically rentable and represent further economic loss (Amor et al. 2007). Roads have been described as the single most robust predictor of frontier expansion and subsequent deforestation in tropical forest (reviewed by Kaimowitz and Angelsen 1998). Given the impact of tropical deforestation on global warming (Silver et al. 2000) and biodiversity loss (Myers, Mittermeier et al. 2000) there is an array of studies that analyze different aspects of road impact on deforestation (Cropper et al. 2001; Verburg et al. 2004; Pfaff, et al. 2007a; Pfaff, et al. 2007b; Pfaff, et al. 2008). However, it is well known that road effects in tropical ecosystems extend beyond deforestation alone. Roads increase forest impoverishment through logging and understory fire (Nepstad, et al. 2001), and they affect species habitat quality by increasing edge habitats (Gullison and Hardner 1993; Malcolm and Ray 2000). Roads are barriers for many species, subdividing populations and producing negative demographic with probably negative genetic consequences (Forman and Alexander 1998). The access that roads provide to the forest fringes increases resource exploitation, such as illegal logging and hunting for bush meat (Wilkie, et al. 2000).

It is essential to understand the impact of road investments on both deforestation and on species habitat, especially for the design of development and ecosystem management policies if we aim to conserve the remnant tropical forest. In this chapter we modeled jaguar habitat to understand the impact of roads, base of this results we were able to estimate the jaguar habitat changes from 1980 to 1990 and from 1990 to 2000. The objectives of this chapter are:

1. Model jaguar habitat in the Mayan forest using Jaguar GPS locations.
2. Understand the role of roads on jaguar habitat selection.
3. To assess if roads have a different impact between males and females and during day and night.
4. To estimate the percentage of habitat loss from 1980-1990 and from 1990-2000 in the Mayan Forest.

Methods

Species

I focused on the impact of road investments on jaguar habitat loss and fragmentation, since this species fulfills most of the criteria of an umbrella, keystone, and flagship species for biodiversity conservation planning (Gomez et al. 2004). Jaguar habitat includes the habitat of a large number of plant and animal species habitats (Nunez et al. 2000) making it one of the optimal umbrella species for large-scale conservation planning (Wikramanayake, et al. 1998; Coppolillo, et al. 2004). Jaguars can be considered as a keystone species since they have a significant influence on the ecosystem by regulating the population dynamics of a large number of prey species. Therefore, a reduction in their numbers will most likely affect the entire system (Mills, Soulé et al. 1993). Moreover, jaguars are as well a flagship species, since are emblematic and people relates to it.

The jaguar is the largest felid in the western hemisphere and the third largest felid worldwide. It is a species of conservation concern throughout its range (Novack, et al. 2005). Its actual distribution ranges from Mexico to Argentina and inhabits areas in the arid scrublands of northern Mexico, the moist tropical forests of Central and South America, and the grasslands of the Pantanal in Brazil, which represent only the 54% of its former habitat (Kinnaird, et al. 2003). Jaguar habitat loss is mainly due to land use changes towards agriculture, cattle ranching and human settlement, which enhances conflict between jaguars and humans (Polisar, et al. 2003). Of the factors that threaten jaguar, habitat loss and fragmentation are considered to be the most critical (Medellin, et al. 2000). This problem is especially severe in areas such as the Mayan Forest were deforestation rates are increasing, especially in the Guatemalan region (see Chapter II).

The Mayan forest is the biggest patch of tropical forest and one of the priority jaguar conservation units (JUC) defined by the Jaguar Conservation Program (Medellin, et al. 2000). The JUC are areas where the population of resident jaguars is potentially self-sustaining over the next 100 years (Kinnaird, et al. 2003). Our study area is part of this JUC and as well includes the priority sites for the species conservation, which was assessed by the tri-national initiative “Jaguars without Borders”. This is an international effort to conserve the Mayan Forest between different sectors: governments, academia and non governmental organization of Guatemala, Belize and Mexico using the jaguar as a flagship species. On the other hand, the Mayan forest is a key region to develop a network of roads to connect the archeological places to promote tourism (BID 2005) as well and to develop highways to provide a direct route for commerce between the USA, Mexico and Guatemala (Mena-Rivero, et al. 2004).

Most of the jaguar studies have focused on feeding ecology, activity patterns, spatial organization and estimations of population density. However, few studies have modeled jaguar habitat (Zarza 2006). Previous research suggests that in the tropical forest of southern Mexico Jaguar density is of one individual per 40-60 Km^2 in the tropical forest of southern Mexico, and one jaguar per 60-100 km^2 in other regions of Mexico (Aranda 1990). In Belize around of one individual per 40 km^2 is estimated to be present in the Tuichi/Hondo to a high of one per 11 km^2 in Cockscomb Basin Wildlife Sanctuary (Silver, Ostertag et al. 2000). In the Brazilian Pantanal the jaguar density estimated is higher; between of 6.5–6.7 individuals/100 km^2 (Soisalo and Cavalcanti 2006). Jaguar density highly depends on prey availability and territorial disputes, which as well defines individual’s home ranges. Previous studies suggest that one male jaguar home range could contain the ranges of two or more females (Maffei, et al. 2004). Rabinowitz & Nottingham (1986) found evidence of range overlaps among females as well as among males. The estimates of jaguar home ranges from several studies range from 14.1 – 116.5 Km^2 (Table 6). The home ranges will vary as a result of the relative abundance of prey species, which may influence the home range sizes as well as the prey selection patterns by jaguars (Schaller and Crawshaw 1980; Crawshaw and Quigley 1991; de Azevedo and Murray 2007). The availability of a prey species for a predator is set by the prey species productivity and the intensity of its use by other predators (Hespenheide, 1975). The prey species activity patterns probably determine as well jaguar crepuscular activity patterns (Emmons 1986; Rabinowitz and Nottingham 1986; Maffei, et al. 2004). The strong dependence of jaguars on their species abundance and activity makes them an ideal species to assess the impact of roads investments on the Mayan Forest. Therefore, they could act as a proxy of forest degradation.

In this study we considered as well gender differences, since there is a clear difference between females and males home range sizes. Models that do not test for this difference can under-represent the impact of roads on females which play a key role for the population viability, especially while raising their cubs.

Table 6 Estimation of jaguar home ranges reported in the literature.

Home Range Km^2	Site characteristics	Gender	Type of study	Source
63	Mato Groso, Pantanal	M	Radio Telemetry	Average estimated from: (Schaller and Crawshaw 1980)
31.5	Mato Groso, Pantanal	F	Radio Telemetry	Average estimated from: (Schaller and Crawshaw 1980)
33.4	Cockscomb Basin, Belize	-	Radio Telemetry	(Rabinowitz and Nottingham 1986)
14.1	Mixture Inundated grassland	-	Radio Telemetry	(Crawshaw and Quigley 1991)

Home Range Km²	Site characteristics	Gender	Type of study	Source
	and woodlands pantanal Brazil			
81.4	Venezuela Llanos, dry season	F	Radio Telemetry	Average estimated from: (Scognamillo, Maxit et al. 2003)
53.75	Venezuela Llanos, rainy season	F	Radio Telemetry	Average estimated from: (Scognamillo, Maxit et al. 2003)
100	Venezuela Llanos, dry season	F	Radio Telemetry	Average estimated from: (Scognamillo, Maxit et al. 2003)
24	Venezuela Llanos, rainy season	F	Radio Telemetry	(Scognamillo, Maxit et al. 2003)
65	Bolivia's Chaco, dry forest	M	Camera trapping	(Maffei, Cuellar et al. 2004)
29	Bolivia's Chaco, dry forest	F	Camera trapping	(Maffei, Cuellar et al. 2004)
116.5	Brazilian Pantanal	M	GPS Telemetry	(Soisalo and Cavalcanti 2006)
58.55	Brazilian Pantanal	F	GPS Telemetry	Average estimated from: (Soisalo and Cavalcanti 2006)
63.38	Brazil, Flood plain forest	M	Radio Telemetry	(de Azevedo and Murray 2007)
38.20	Brazil, Flood plain forest	F	Radio Telemetry	(de Azevedo and Murray 2007)

Study area

The study area includes the majority of the Petén of Guatemala, most of Belize with the exception of the Toledo district in the south, and a large portion of the states of Campeche and Quintana Roo in Mexico. It covers most of the Mayan Forest and includes the protected areas of Calakmul in Mexico, the Mayan Biosphere Reserve (RBM) and part of the protected area of the Sierra del Lacandon in Guatemala, as well as the protected areas of Chiquibul, Rio Bravo, Manatee in Belize. The total area, delimited by four Landsat satellite images, covers around 100,000 km². This region is predominantly tropical semi-deciduous forest, with an average annual rainfall of 1350 mm and a pronounced dry season between February and June (Pennington 1968; Holdridge 1971). However, the rainy season in the past years has been from July to October. The Mayan Forest is almost completely covered by mature forest, classified as subtropical moist (Holdridge et al., 1971). This type of forest can be classified in 3 categories; upland forest, bajo or lowland forest, and swamps. Upland forest is found in areas of greater relief, and is characterized by a high, closed tree canopy. Bajo forest is mostly dry deciduous lower & midsize forest and lowland alluvial forests. It has a low, somewhat open canopy, thick underbrush, and is seasonally inundated (Novack, et al. 2005). Swamps are classified as Mesoamerican palustrine vegetation (Fa, et al. 2000).

The Mayan Forest is the second largest contiguous patch of tropical forest in the Americas and the biggest forest of the Mesoamerican Hotspot (Myers, et al. 2000), which is one of the most biodiverse regions in the world (Kaiser 2001). The Mesoamerican Hotspot holds 7% of the world's species and is home to approximately 2,318 endemic vertebrates (5.7 % worldwide), however, only 25% of its forests remain (Myers, et al. 2000).

Data collection

Jaguar data

From 2001-2007, researchers and staff from the National University of Mexico, Jaguar Conservancy, ECOSAFARIS and Unidos Para la Conservacion, and Duke University captured jaguars during the dry season (from January to May) in the Biosphere Reserve of Calakmul and the forestry ejido of Caobas in Mexico. The same institutions together with Defensores de la Naturaleza y Fundacion ARCAS captured during two seasons (2006 and 2007) in the Mayan Biosphere Reserve in Guatemala. The captured jaguars were chemically anaesthetized using a projectile dart. A dose of 11 to 16 mg/kg of ketamine was used to immobilize the animals (Ceballos, et al. 1999). Immobilized animals were examined for general body condition, measured, weighed, and fitted with Lotek GPS collars (Lotek engineering, Newmarket, ON, Canada; see URL http://www.lotek.com). The GPS collars were programmed to fix the geographic coordinates of the individual's position at intervals between two and four hours. The GPS recorded the location from between 2 to 12 months, this variation is because some of the GPS collars batteries were damaged. Most of the individuals were recaptured the following year. During the recapture, their GPS coordinates were downloaded to a computer (laptop) and the collar's GPS batteries were changed. The individual was re-collared and released; some individuals were again recaptured in the following years. For this study we used the data GPS points of 3 female individuals found in Mexico outside of a protected area, one male found in the Calackmul biosphere reserve in Mexico, and two males found in the Mayan Biosphere Reserve in Guatemala (Figure 1).

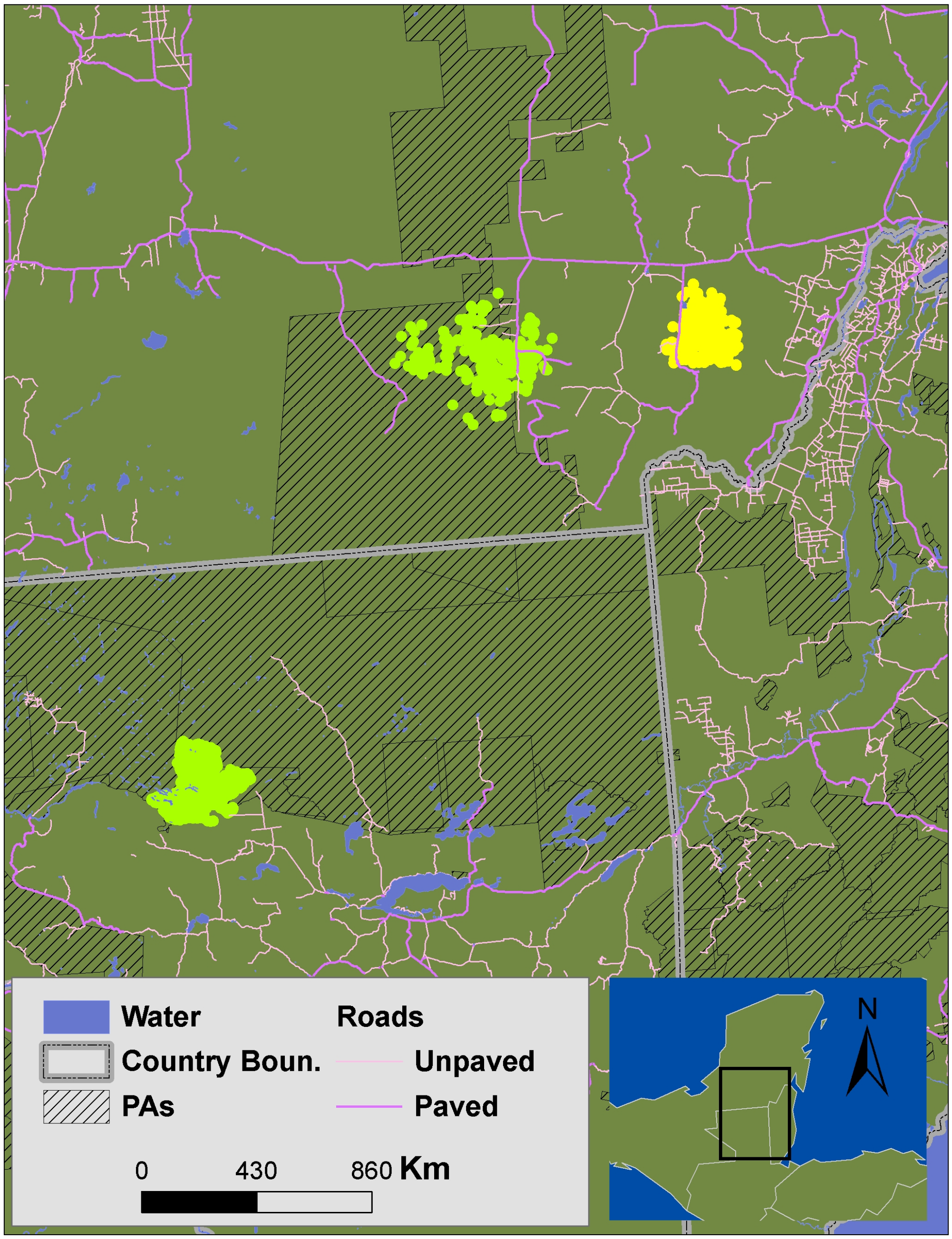

Figure 4 Study Area and jaguar GPS points

In Yellow are the female data points (3 individuals) and in green are the male data points (3 individuals).

Landscape Data

We used the Selva Maya Zoque y Olmeca Vegetation and Population Center's data layers for the year 2000 (Garcia and Secaira 2006). To analyze changes in habitat quality and fragmentation from 1980-1990 and 1990-2000 we used the deforestation and road investments data developed in chapter II. To determine vegetation types previous to the 1990 and 2000 period, we used the Selva Maya Zoque y Olmeca potential vegetation map (Garcia and Secaira, 2006).

In order to model jaguar habitat we focused on nine land use variables with likely importance for jaguar habitat selection, including: vegetation type, distance to roads, density of population centers (i.e., cities and towns), distance to water, elevation, human population density. To determine whether habitat use and road avoidance varies among individuals, season and during the day and night, we created a dummy variable for: gender, season (dry/rainy), and time of day (day/night). (Elevation was taken from the Shuttle Radar Topography Mission dataset http://edc.usgs.gov/srtm/data/obtainingdata.html). The dummy variable for "season" separated the rainy from the dry season, and we created as well a binary "day/night" variable to describe the time at which GPS points were acquired (Table 7).

To understand the spatial organization of jaguars given the land cover, and to estimate the size of their home ranges we approximated individual home ranges by their minimum convex polygons (MPC) and estimated percentages of vegetation types and mean values of habitat variables in each home range using ArcView 3.2 (ESRI, Inc, Redlands, CA, US).

Jaguar Habitat Model

To model the species' potential habitat in the Mayan Forest we used a habitat availability approach which assumes that observed occurrences are a sub-sample of available sites that inform the animal's preferences (Manly et al. 1993). This approach has the advantage that differs from the presence-absence models that assume that certain areas are never use (Boyce, et al. 2002; Pearce 2002). To sample the domain of available habitat, we distributed random points in a 10-km buffer around each individual's home range; we set the number of random locations equal to the number of GPS locations per individual after data filtering. At each random pseudo-absence location, we collected spatially coincident values of the habitat and gender variables and randomly assigned season and day/night values.

To reduce temporal autocorrelation, other studies of habitat use by carnivores have included only points separated by a number of hours chosen somewhat arbitrarily (Klar, et al. 2008). Here we calculated temporal autocorrelation in both latitude and longitude between the locations x_i, t for each jaguar i at times t = 0 (in hours) with respect to their position at times $t = \{t + 1, \dots , t + k\}$, k being the maximum lag we explored (up to 7 days = 168 hours). We calculated empirical variances for the initial series x_0 and that at a lag t (x_t) as

$$\mathrm{var}[x_t] = \frac{1}{n_t} \sum_{i: x_{t,i} \neq NA} (x_t - \bar{x}_t)^2 \tag{4}$$

where n_t is the number of known observations for series x_t, and $\bar{x}_t$ is the mean for the series. We then calculated the covariance between x_0 and x_t as

$$\mathrm{cov}[x_0, x_t] = \frac{1}{n_t} \sum_{i: x_{t,i} \neq NA} (x_0 - \bar{x}_0)(x_t - \bar{x}_t) \tag{5}$$

We obtain autocorrelation values between x_0 and all lagged series as

$$r_{0,t} = \frac{\mathrm{cov}[x_0, x_t]}{\left(\mathrm{var}[x_0]\,\mathrm{var}[x_t]\right)^{1/2}} \tag{6}$$

where $r_{0,t}$ is known as the crosscorrelation between series x_0 and x_t, with values ranging from -1 to 1 (REF). A low crosscorrelation will tend to zero. Then we found the time interval after which the crosscorrelation was lower than 0.3 for all individuals in either of both dimensions (i.e., latitude and longitude). A time interval between observations for which the crosscorrelation is 0.3 or less implies that these can be considered independent; thus further analysis will not require incorporating explicitly temporal autocorrelation into the models.

We used a Generalized Linear Model (GLM) to analyze jaguar habitat preferences, focusing on the impact of roads. GLMs are commonly used to assess species habitat and to select key areas for species conservation (Pearce 2002; Westphal and

Possingham 2003; Rhodes, Wiegand et al. 2006; Klar, Fernandez et al. 2008). These models usually include explanatory variables representing both natural and anthropogenic factors; however they have rarely been used to identify gender habitat preferences. We included in the model interaction term of gender and land-cover, and to test for differences of jaguar's gender avoidance toward roads we included in the model an interaction term of gender and road distance. A significant coefficient on the interaction term indicates that the relationship between jaguar presence and distance to roads varies by gender. Also, a significant coefficient of the interaction term gender and land-cover indicates that jaguar preferences toward land-cover changes by gender. In table 7 we show the variables that we used to model jaguar habitat.

Table 7. Variables used to model the impact of roads and jaguar habitat use.

Variable	Description	Source
Distance to roads	Euclidian distance to roads (Km)	Chapter II
Elevation (m)	From Digital Elevation Model (90m resolution)	SRTM-DSM
Human population density	Estimated as a Kernel density	From Chapter II
Distance to water	Estimated Euclidian distance (from the vegetation layer)	(Garcia and Secaira 2007)
Forest (dummy)	Seasonal ever green forest	(Garcia and Secaira 2007)
Lowland forest (dummy)	Dry deciduous lower & midsize forest; lowland alluvial forests	(Garcia and Secaira 2007)
Swamps (dummy)	Mesoamerican palustrine vegetation	(Garcia and Secaira 2007)
LUC (dummy)	Agriculture, secondary growth, cattle range, urban	(Garcia and Secaira 2007)
Season (dummy)	From July to October: rainy and form November to June: dry	Data points (GPS collar)
Night (dummy)	Night: from 8pm to 5pm = 1 Day: From 6am to 6pm = 0	Data points (GPS collar)
Gender (dummy)	Male (marroco & tony) = 1; Female (dalia, paola & eugenia) = 0	Data points (GPS collar)

We modeled the likelihood of jaguar presence at each site *i* as a Bernoulli process (y_i ~ Bernoulli(p_i), with probability p_i); we linked this probability with relevant covariates (Table 7) through a logit-link function of the form:

$$\ln\left(\frac{p_i}{1-p_i}\right) = \mathbf{x}_i\boldsymbol{\beta} \tag{7}$$

where $\mathbf{x}_i$ is the vector of covariates (Table 7) for point *i* and $\boldsymbol{\beta}$ is the vector of parameters linking $\mathbf{x}_i$ and $\boldsymbol{p}_i$. The fitted models were compared using the Akaike Information Criterion (*AIC*), which penalizes the maximum likelihood according to the number of model parameters (Akaike 1974; Achard, Eva et al. 2002), and the model with the lowest AIC was selected for further analysis. The AIC of a model is calculated as:

$$AIC = -2\log(L) + 2K \tag{8}$$

where *L* is the marginal likelihood of the model and *K* is the number of parameters in the model. The AIC of a model is the relative likelihood of the model compared with all other models possible from a set of covariates (Boyce, Vernier et al. 2002).

To avoid spurious results in the parameter estimation due to a biased random selection of pseudo-absence points, we ran the analysis with 2000 different sets of pseudo-absence points and stored the resulting estimates. For each jaguar, we generated 1000 random points and randomly selected in each set a number of pseudo-absences equal to the number of observation per jaguar. This method could be considered as a "semi" non-parametric bootstrap (Clark 2007), since it is only applied to the random points, and not to the entire set. This allowed us to calculate mean values, 95% confidence intervals for each parameter and *p* values as the proportion of estimates larger than zero for parameters with positive effects, and lower than zero for those with negative effects.

Mapping the early history of jaguar habitat

To evaluate the impact of roads on jaguar habitat from 1980-1990 and 1990-2000, we translated a map of potential vegetation in 1980 into a map of jaguar habitat. We used our deforestation maps for 1980 and 1990 (Chapter II) and the Maya Zoque y Olmeca vegetation map for 2000 (Fa et al. 2000) to locate disturbed areas in each period. We used our model to generate a probabilistic landscape of jaguar habitat based on the vegetation map and the roads and population centers present in each period. To convert the probabilistic landscape into a habitat-non-habitat landscape, defined a threshold of habitat suitability for each time period based on ROC analysis. Finally, we used the software FRAGSTATS (McGarigal, et al. 2002), to determine the number and size of habitat patches in each period.

Results

Descriptive Statistics

We found that female's smallest home range (delimited by the MCP) was of 122 Km^2 and the maximum was of 293 Km^2 (Table 3). For males the smallest home range was around 280 Km^2 and the maximum was close to 970 Km^2. However, these results could be biased since we did not have the same number of observations per jaguar. The average distance to roads in the home ranges was higher for males than for females, which is expected since the males were captured inside protected areas. However, even if the males were mainly inside of protected areas we found that the highest average for human population density was in one of the males home ranges (Table 3). The percentage of forest within each home range (including both types: evergreen and lowland) ranged from 78-90% with the exception of Marroco, which consisted of 68% of forest and 29% of swamps.

Table 8 Mean values of continuous variables per individual's MCP

Variables/ jaguar name	Dalia	Eugenia	Marroco	Paola	Tony
Distance to roads (m)	4,005	3,436	8,863	3,970	5,741
Elevation (m)	155	149	78	153	287
Human Population density	18	26	14	25	80
Total Km^2	12,167	29,328	28,315	12,574	97,005

Table 9 Percentages of vegetation types found per individual's MCP.

Land cover/ jaguar name	Dalia	Eugenia	Marroco**	Paola	Tony**
Jungle	69.15	58.91	57.70	60.73	72.58
Jungle lowlands	25.88	28.11	10.95	31.18	6.35
Swamp	0.00	0.00	29.16	0.00	0.00
Secondary*	0.01	4.16	0.00	0.13	4.87
Agriculture and cattle*	0.00	6.15	0.02	3.16	16.20
Bare ground/grassland*	4.96	2.67	2.17	4.80	0.00
Home Range inside of Protected Areas	0.00	0.00	83.60	0.00	76.00

*these categories were classified as one for the statistical analysis. ** Male jaguars.

Our analysis of autocorrelation showed that temporal crosscorrelation is minimized by using only observations separated by at least 3 days. This temporal threshold between observations showed a correlation coefficient lower than 0.3. As can be seen in Figure 4, the temporal crosscorrelation for all the individuals decays at a three day threshold, being clearer for those individuals with larger data sets such as Marroco, Paola and Tavo. This resulted in a total sample size of 272 jaguar observations (presence).

Jaguar's habitat model and road impact

Our analysis of autocorrelation showed that temporal crosscorrelation is minimized by using only observations separated by at least 3 days. This temporal threshold between observations showed a correlation coefficient lower than 0.3. As can be seen in Figure 4, the temporal crosscorrelation for all the individuals decays at a three day threshold, being clearer for those individuals with larger data sets such as Marroco,

Paola and Tavo. This resulted in a total sample size of 272 jaguar observations (presence).

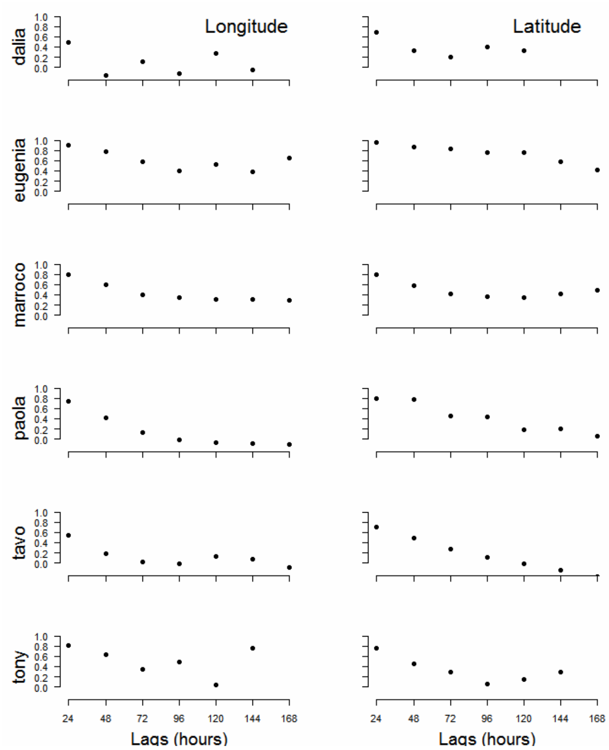

Figure 5. Temporal crosscorrelation in both dimensions (i.e. latitude and longitude) for all jaguars.

The most parsimonious model (AIC = 243.7) included: forest, and disturbed land-cover types. It included as well gender, road distance and the interaction term of road distance and gender. Parameter estimates from this model are shown in Table 10. This resulted in a model that dropped variables such as: season, day/night, and the elevation. Jaguar habitat preferences are strong for forest and negative for disturbed areas (Table

10). Also, our results suggest that road effects on jaguars habitat selection are not the same for males and females.

Table 10. Model coefficients and standard error of the model resulted from the bootstrap.

Response variable jaguar observations				
Parameter	**Mean**	**2.50%**	**97.50%**	**pval**
Intercept	-2.13	-2.55	-1.74	< 0.0001
Roads distance (Km)	0. 444	0. 333	0. 570	< 0.0001
Gender = male	1.52	1.09	1.94	< 0.0001
Forest	0.759	0.401	1.10	< 0.0001
Lowland forest	0.0886	-0.327	0.495	0.325
Interaction = male: road dist	-0. 44	-0. 56	-0. 32	< 0.0001
Residual deviance			2589.6	
Number observations: presence (GPS points)			929	
Total obs. (GPS + Random data points)			1867	

As can be seen in Figure 5, for males and females the intercept is defined by the land cover types and the slopes show the effect of the distance to roads for each of the vegetation types. The slope shows little change for males, indicating that roads distance has only a small effect on their habitat selection. However, for females it is clear that the probability of finding a female is significantly higher as the distance to a road increases (Figure 3). In contrast with the males, females clearly select those areas that are far from roads, even when all the males were found inside of protected areas that have enough space to allow them to avoid roads (Figure 3).

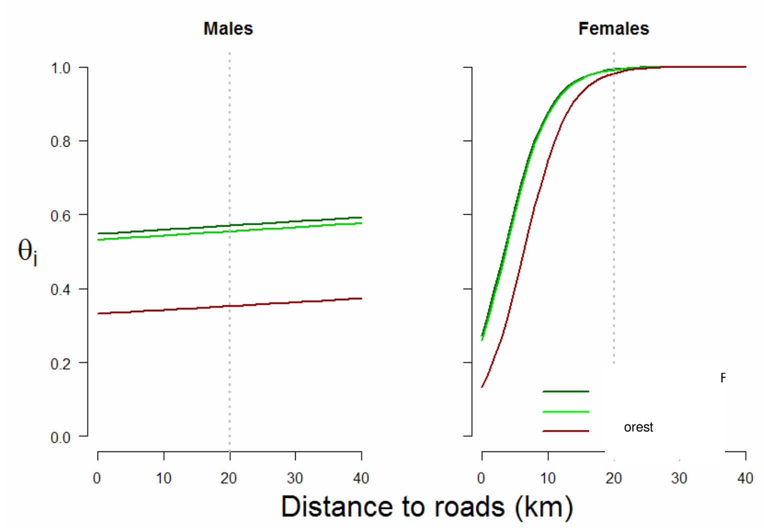

Figure 6. Graph of probability of occurrence of male and female jaguars based on type of vegetation and distance to roads

Roads impact on the Mayan Forest from 1980 to 2000: a jaguar's perspective.

The cross-validation area under the ROC curve for females was 0.74 for true positive and 0.46 for false positive, indicating reasonable discrimination ability (Pearce and Ferrier 2000, Figure 4). Therefore, we concluded that the structure of the model was appropriate, and we used the 0.74 threshold to generate a habitat-non-habitat map to quantify habitat loss and fragmentation for female jaguars for the three periods. We did not estimate the area under the ROC curve for males. This is because the GLM model results show that males' probabilistic landscapes are not significantly affected by the distance to roads distance. Therefore, the only determining factor is the land cover, which is a categorical.

Our results indicate that 1980 to 2000 females lost approximately 36% and males lost approximately 22% of their habitat (Table 6). From 1980 to 1990 females lost 14,091 km2 of habitat and from 1990-2000 they lost 10,802 Km2. Males, in comparison, lost around 1,782 Km2 from 1980-1990, which is an order of magnitude less than the 17,262 Km2 of habitat loss in the following period (1990-2000).

Females' habitat fragmentation increased during these two periods. This is clear by the creation 157 new patches from 1980 to 1990 and of 577 patches from 1990 to 2000. During this last period, the generation of small new patches almost doubled (Table 6). Males showed a similar trend to females in fragmentation. The highest habitat fragmentation occurred during the last period. From 1980 to 2000, the size of the biggest habitat patch for females was reduced in a 53% (from 37,994 to 20,410 Km^2) and for males in a 64.93% (from 85,731 to 55,672 Km^2), which is reflected in both habitat loss and fragmentation.

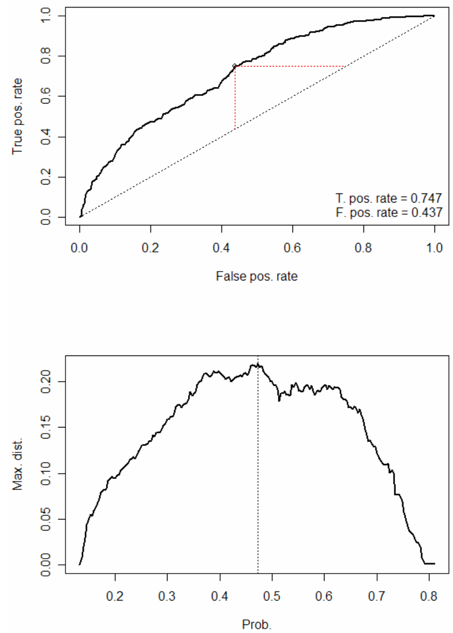

Figure 7. Result from the Receiver operating characteristic (ROC) analysis for the probability of jaguar occurrence.

Top: Relationship between the rate of true positive versus false positive based on our model; bottom: Probability at which the ratio between true positive and false positive rates is maximized

Our results indicate that 1980 to 2000 females lost approximately 36% and males lost approximately 22% of their habitat (Table 6). From 1980 to 1990 females lost 14,091 km2 of habitat and from 1990-2000 they lost 10,802 Km2. Males, in comparison, lost around 1,782 Km2 from 1980-1990, which is an order of magnitude less than the 17,262 Km2 of habitat loss in the following period (1990-2000).

Females' habitat fragmentation increased during these two periods. This is clear by the creation 157 new patches from 1980 to 1990 and of 577 patches from 1990 to 2000. During this last period, the generation of small new patches almost doubled (Table 6).

Males showed a similar trend to females in fragmentation. The highest habitat fragmentation occurred during the last period. From 1980 to 2000, the size of the biggest habitat patch for females was reduced in a 53% (from 37,994 to 20,410 Km^2) and for males in a 64.93% (from 85,731 to 55,672 Km^2), which is reflected in both habitat loss and fragmentation.

Table 11 Results of habitat change from 1980–1990–2000 for female and male habitat availability.

The size of the biggest available patch is under Max patch column, the mean patch size is referred as mean patch size, and the patch size standard deviation is Sd. The number of patches by size ranges shows the number of patches within each size category.

Year	Number of Patches	General habitat metrics (Km^2)				Patch numbers by size category (Km^2)			
		Tot habitat	Max. Patch size	Mean patch size	Sd.	< 100	100 to 1,000	1,000 to 10,000	10,000 to 100,000
Females									
1980	465	74,170	37,994	160	2,013	73	4	6	2
1990	622	60,079	29,317	97	1,261	101	23	6	1
2000	1,199	49,277	20,410	41	638	213	22	8	1
Males									
1980	3,812	87,859	85,731	23	1,389	121	3	0	1
1990	6,871	86,111	83,886	13	1,012	133	3	0	1
2000	2,539	68,849	55,672	27	1,113	241	12	1	1

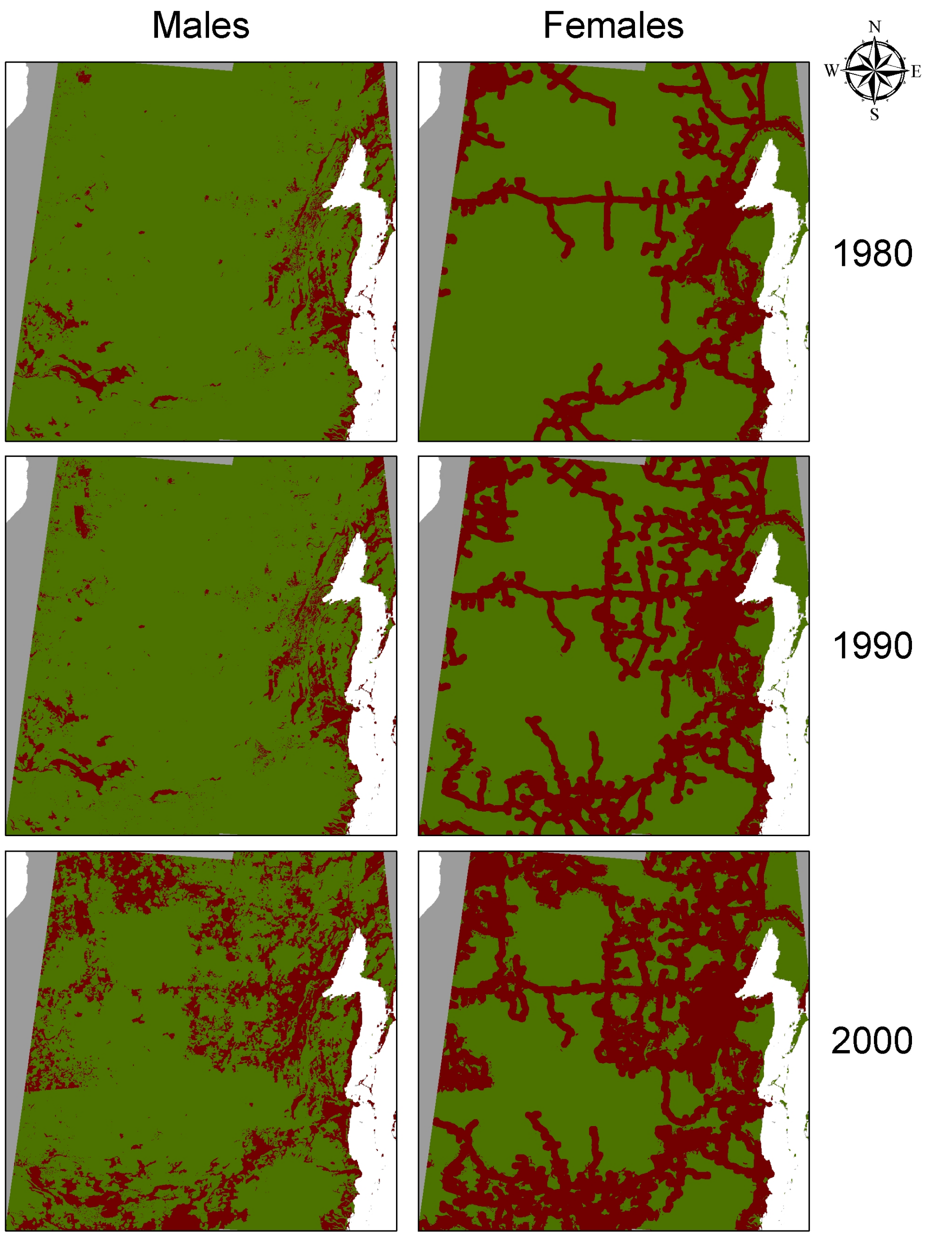

Figure 8. Maps of habitat availability for males and females in 1980, 1990, and in 2000.

In green we show potential habitat and in red non-habitat

Discussion

Jaguars are sensitive to the presence of roads and the infrastructure that they convey. Moreover, we found that the effects of roads differ by gender; females are considerably more vulnerable to the proximity of roads. The differences between males and females are consistent with other studies were they found that female jaguars show much more restrictive movements than males (Schaller and Crawshaw 1980). As well, this gender difference could be similar to that found in other big carnivores such as tigers. Goodrich et al. (2002) found that large tiger females' effectiveness to raise their cubs is highly limited by the proximity of roads. Our study suggests that this level of vulnerability could be associated not only with the restrictive movement of jaguar females but it could be as well associated to their ability to raise their cubes in close proximity to roads. However, further studies will be needed understand what drives the different tolerance between male and female jaguars toward roads. It is important to add that the home ranges of the studied animals were only in areas with dirt and two lane paved roads. It is likely that interstate routes and four lane roads have a greater effect on both males and females. Further studies need to analyze this issue, especially with the extension of the Escarcega-Ixpujl road in the Mayan Forest, which crosses the protected areas of Balanku and Calakmul.

We also found the largest home range for jaguars reported in the literature, ranging from 122 to 970 Km^2. This is considerably higher to the ones described by previous findings (14.1 to 116.5 Km^2; Table 1). However, we can not generate conclusions by this comparison, since our results have a higher spatial resolution, since they were obtained with direct GPS readings from the collar, while previous studies used different methods such as camera trapping, radio telemetry, and GPS radio telemetry. In this sense our data are much more reliable to assess the extent of jaguar home ranges. Nevertheless, our findings confirm and enhance knowledge of the large extent of jaguars' home ranges, which makes the species one of the optimum umbrella species for large-scale conservation planning (Wikramanayake, et al. 1998; Coppolillo, et al. 2004).

Our reconstruction of the species habitat in 1980's, 1990's and early 2000's show the dramatic impact of roads and subsequent deforestation on jaguar's habitat in the Mayan Forest. From 1980 to 2000, 34% of female and 22% of male habitat was lost while fragmentation doubled. Our results stress the importance of considering the impact of roads not only on deforestation but as well on species habitat. If we only account for deforestation we are not able to understand the impact of infrastructure projects beyond the canopy loss. However, if we analyze species' responses to roads investments we can infer its impact beyond deforestation. Roads directly or indirectly affect the species habitat due to poaching, vehicle collisions, and the reduction on prey availability for carnivores due to the increase on access for hunters. Previous studies on roads impact on large carnivores show that one of the main problems is the reduction of prey due to human hunting for bush meat (Thiel 1985; McLellan and Shackleton 1988; Noss, et al. 1996; Kerley, et al. 2002). The increase in bush meat hunting can severely affect jaguar densities, survival and reproductions due to reduction in prey availability (Fuller and Sievert 2001; Novack, et al. 2005). Therefore, road impact on jaguar habitat could be as well a representation of road indirect or direct impact on prey species such as the collared peccary, coati, armadillos and tapir, among others (Taber, et al. 1997; Novack, et al. 2005; Weckel, et al. 2006).

The dramatic loss of female and male jaguar habitat during the last two decades shows the high impact of land cover change and road investments in the Mayan Forest. Development plans such as the Mundo Maya that propose the interconnection of Mayan archeological sites, as well as the road development by the Plan Puebla Panama, could have severe impacts on the ecological functionally of the last large intact tropical forest in Central America. Studies that analyze the impact of the proposed road investments on deforestation and jaguar habitat will be essential to find alternative sites for road development that will prevent further deforestation and habitat loss.

4. Impact of a road investment in the Mayan Forest: forecasting and policy alternatives

Introduction

The Mayan forest of Mexico, Belize and Guatemala is one of the largest remnants of tropical rainforest in the Americas, second only to the Amazon (Garcia and Secaira 2006). However, expansion of the road network in the region to promote the commerce between Mexico, Guatemala and the USA is proposed as a consequence of the new free trade agreement between Central America and the USA. This expansion of the road network is part of a regional development plan for southern Mexico and Central America. The plan proposes the development of more than 10,000 Km of roads, around 321 dams, oil and gas pipe lines, as well as the expansion of electric power lines (Burgues 2006). The project originated from the Plan Puebla Panama (PPP), which was intended to extend from the Darien, in Panama, to Puebla, Mexico (SIECA 2004). Today, this development project extends well beyond this range. The targeted region of the Plan Puebla Panama has been catalogued as one of the world's primary Biodiversity Hotspots, harboring 7% of the world's species, having lost up to 70% of its original area (Myers, et al. 2000). Despite national and international conservation strategies, such as the Mesoamerican Biological Corridor (MBC), deforestation rates have steadily increased since the 1970s (Chapter II), placing Central America and Mexico as the region with the highest deforestation rate in the Western Hemisphere (FAO 2005). Moreover the development of the PPP projects will severely impact the remnant forests and their connectivity.

The aim of this Chapter is to forecast the effects of the road investments proposed as part of the project, together with other environmental and anthropogenic variables, on the deforestation rate of the Mayan forest. Also, we evaluated habitat loss and fragmentation for the largest carnivore of the region, the jaguar (*Panthera onca*) as a way to estimate the impact of this project on the forest degradation. Our analysis focuses on the Chetumal-Guatemala International road proposed by the PPP (RUCIA 2002) and the alternative route through Belize.

The Chetumal-Guatemala International road is part of a commercial and tourism circuit that aims to connect the state of Quintana Roo in Mexico, with Guatemala and the rest of Central America. This road will facilitate the trade between the US and the countries of the region trough the Atlantic, while increasing the direct flow of tourists. This project is of high relevance, since the Central American Free Trade Agreement (CAFTA) has been signed by Guatemala, El Salvador, Honduras, Nicaragua and Costa Rica (CIC 2008). However, this road will bisect the Mayan Biosphere Reserve (MBR) which together with the protected areas of Calakmul and Balam-kú represents the largest patch of intact tropical forest of the Mesoamerican Hotspot, and second largest tropical forest in the Americas after the Amazon. The development of this road is of special concern, since road construction is one of the most robust predictors of frontier expansion and subsequent deforestation in tropical forests (reviewed by Kaimowitz and Angelsen 1998).

The conservation of the Mayan Forest is essential to maintain important environmental services, such as local climate regulation, water capture, natural crop pollinators, and global CO_2 sequestration. The local environmental services are essential for the local communities' survival and they are a buffer against hurricanes and floods. Therefore, it is important to assess the impact of the CGI road, and compare it to alternative investments. Based on our results from chapter II, we expected that a new investment will have a relative higher impact if it is placed in areas that were already developed than if placed in pristine areas (Pfaff, et al. 2007). Therefore, we would expect that in the short term the CGI that crosses the MBR will have a lower impact on deforestation than the improvement of the existing road through Belize. However, the impact of the CGI it is expected to increase since the colonization of the region will demand further investments and eventually the development of new roads. It is also important to consider how the impact of the alternative investments will differ given the country and the level of protection of the area. We expect that the interaction of the road investments with the country variable could be a key element to incorporate in a forecasting model, since it could be seen as an indicator of the effects of macro level and policy instrument variables on deforestation (Kaimowitz and Angelsen 1998). To assess these issues and forecast the impacts of these alternative routes, we developed a deforestation model based on Chapter II findings. Because road impacts go beyond deforestation, we extended our analysis to forecast the effects of the two alternative investments on the loss and fragmentation of jaguar habitat.

Methods

Deforestation model

Based on the previous analysis in the study area (Chapter II), we assumed that the probability of clearing a parcel (100 x 100m pixel) in the Mayan forest will be influenced by distance to an existing road investment, soil type, distance to previous deforestation, elevation, distance to markets, country, protected areas, and the level of development (Table 12). For the purpose of the forecasting we incorporated in the same model all these variables in contrast with the analysis in Chapter II where we separated our sample in two areas with different road development context. In Table 12 we show the covariates that were included in this analysis.

Table 12 Model covariates

Covariates	Description	Source of data layer
Country	Dummy for country: Mexico, Belize or Guatemala.	(Garcia and Secaira 2006)
Elevation (m)	Digital Elevation Model	(Garcia and Secaira 2006)
Soil	Four soil classes (see chapter II)	(Garcia and Secaira 2006)
Distance to old deforestation (Km)	Euclidian distance to deforestation until 1990	Chapter II
Distance to roads old (Km)	Euclidian distance to roads present by 1990	Chapter II
Roads new (Km)	Euclidian distance to new road investments placed in 2000.	Chapter II

Covariates	Description	Source of data layer
Level of Development	Dummy for a high development area = close to old roads (distance ≤25km) and low development area = far from old roads (distance > 25Km).	Chapter II
Protected Areas	Dummy for protected areas.	(Garcia and Secaira 2006)
Distance to main markets	Euclidian distance to population centers >8000 people in 2000.	Estimated from (Garcia and Secaira 2006)
Distance to small makes	Euclidian distance to population centers > 2000 people and < 8000 people.	Estimated from (Garcia and Secaira 2006)
Protected Areas : Country	Interaction of protected areas and country.	
Country : Road distance	Interaction of country and distance to a old road.	
Roads 2000 : High or low development	Interaction of Euclidian distance to roads investments in 2000 with the dummy variable for High/low development.	

We used a Generalized Linear Model (GLM) to understand the probability of deforestation in a parcel given the covariates mentioned above. The response variable is deforestation, represented by the indicator y_i, equal to 1 if a parcel i was deforested between 1990 and 2000 and zero otherwise; those deforested previous to that period were not included in the analysis. We modeled deforestation as a Bernoulli process ($y_i \sim$ Bernoulli (p_i)) where p_i is the probability that parcel i is deforested. We linked this probability with relevant covariates (Table 12) through a logit-link function of the form:

$$\ln\left(\frac{p_i}{1-p_i}\right) = \mathbf{x}_i\boldsymbol{\beta} \qquad (9)$$

where $\mathbf{x}_i$ is the vector of covariates (i.e distance to road, elevation, country, soil type, etc.) for point i and $\boldsymbol{\beta}$ is the vector of parameters linking $\mathbf{x}_i$ and p_i.

For model selection we explored different combinations of covariates with the R function "step" from the package "stats" (R development core 2008). This is a maximum likelihood based approach from which Akaike Information Criterion (AIC) (Akaike

1974; Achard, Eva et al. 2002) values are calculated for each model. These AIC values are a measure of goodness of fit based on the log-likelihood, and a penalization term according to the number of parameters (Akaike 1974; Achard, Eva et al. 2002). The AIC of a model is calculated as:

$$AIC = -2\log(L) + 2K \tag{9}$$

were L is the likelihood of the model and K is the number of parameters in the model. Thus, the model with lowest AIC is chosen. The AIC of a model is the relative likelihood of the model compared with all other models possible from a set of covariates.

Forecasting the probability of deforestation

We used the coefficients obtained in the deforestation analysis to forecast the impact of two projected road investments: 1) the "Caobas-Flores Mexico, Guatemala" segment; and 2) the existing route from Chetumal to Flores that runs through Belize (Figure 8). The Caobas-Flores segment was proposed by the Mexican Ministry of Transport and Communications (SCT) as part of the Chetumal-Guatemala International Road, which includes: a) the modernization of the route from Caobas to Arrollo Negro (86 Km) which has been completed, b) the construction of the road from Arrollo Negro, to Uaxactún, and c) the modernization of the segment from Uaxactún to Flores (Figure 8). The alternative project 2) consists on the modernization of the existing route from

Chetumal to Flores trough Belize. Thus, for our purposes a road investment could be new construction or the improvement of an existing road.

To assess the importance of the country variables and protected areas we ran two additional scenarios. The first scenario did not include the dummy variables for protected areas and country. The second scenario included the protected areas variable but not the country variable. The third scenario was the model produced by the lowest AIC value. We ran these three scenarios for the selected model and applied the resulting coefficients to the data layers in 2000 to forecast the effect of the new road investments. Since our models were based on data up to the year 2000, this forecast was initiated at that year. Thus, we used distance to previous deforestation and to old roads as those produced up to 2000. The dummy variable for level of development included as "high development" new roads less than 25 km from existing roads and "low development" new roads over 25 Km from existing roads. Using the latest data layers (year 2000) we generated a probabilistic forecast of deforestation on this landscape by using the Map Algebra tool from Arc View version 3.2 (ESRI, Inc, Redlands, CA, US).

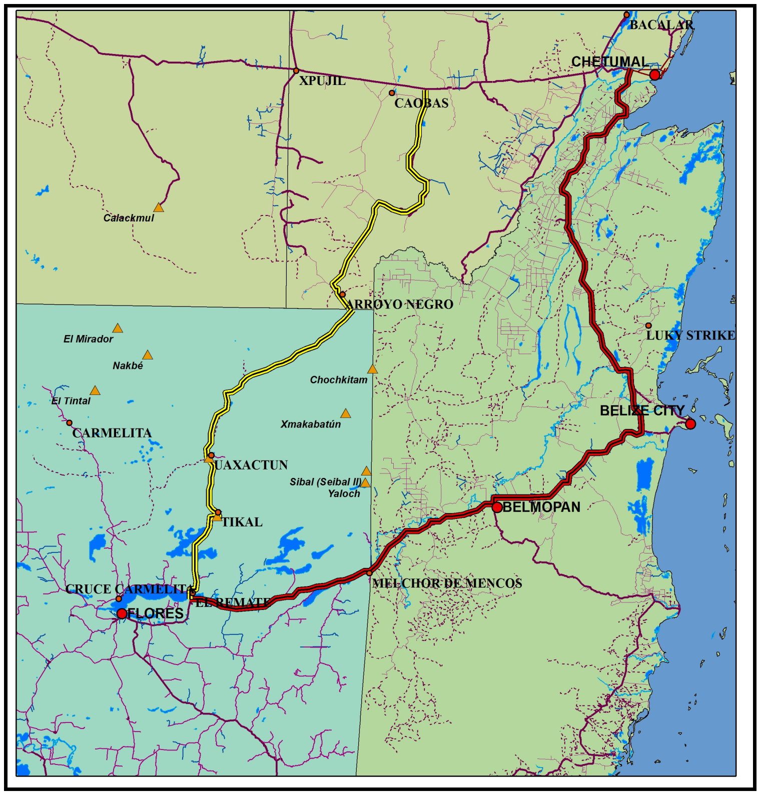

Figure 9. Roads analyzed. In yellow is the CGI road and in red the actual route through Belize.

Estimating deforestation

To convert from a probability-of-deforestation landscape to a forest/non-forest landscape we used the area under the receiver operating characteristics (ROC). The area under the ROC curve is the probability that a randomly chosen truly deforested parcel is correctly ranked relative to a randomly chosen truly non-deforested site (Pearce 2002). Based on the ROC curve, we created a map with two classes: deforested and non-deforested for both road projects, from which we calculated the number of hectares deforested for each of the two roads (Chetumal-Flores and the Belize route).

Forecast the habitat loss for female and male jaguar's in the Mayan Forest from the alternative road investments

To forecast the impact of the alternative road projects on jaguar habitat we created a probabilistic habitat landscape by applying the coefficients obtained from the jaguar habitat model developed in Chapter IV to the data layers of road distance that included both investments. We updated the vegetation layer by classifying as disturbed those pixels that were predicted as deforested from the alternative road investments. To create a habitat-no habitat map from the probabilistic landscape we used the area under the ROC curve that maximized the number of true positive and true negative values. Then we estimated the number of hectares of habitat lost for both alternative road investments.

Results

Deforestation model

The most parsimonious model included the following variables: 1) country (Mexico, Belize and Guatemala); 2) only one soil type C; 3) distance to previous deforestation; 4) elevation; 5) the interaction of distance to old roads and country; 6) interaction of protected areas and country; and 7) the interaction of new roads and the level of previous development (high/low, Table 13).

Table 13 Deforestation model for the year 2000

Covariates	Coefficients	SE	Sign
(Intercept)	-2.544628	0.271127	***
Guatemala	2.233936	0.293291	***
Mexico	0.862473	0.277943	**
Elevation (m)	-0.003272	0.000583	***
Soil C dummy	1.210964	0.162644	***
Distance to def. 80-90 (Km)	-0.087084	0.024518	***
Belize * Road distance	-1.320477	0.382363	***
Guatemala * Roads distance	-0.178306	0.028471	***
Mexico * Roads distance	-0.261643	0.029808	***
Roads 2000 *High Development (close to old roads)	0.166677	0.444085	
Roads 2000 * Low development (Far to old roads)	-0.311736	0.096844	**
Belize * Protected Areas	0.010902	0.551756	
Guatemala * Protected Areas	-1.541769	0.168637	***
Mexico * Protected Areas	-0.557514	0.32497	.
Null deviance	6105.9		
Residual deviance	4899.6		
Number of y= 1 (deforested parcels)			
Total number of observations	14753		
AIC	4930		

Sign codes for pvalue: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

We plotted the probability of deforestation by road distance for high development and low development areas (Figure 9). It is clear that, for both high and low development areas, the probability of deforestation are higher the closer the parcel is to a road. However, the probability of deforestation is higher when a new road is placed in a highly developed area (close to other roads) than when it is placed in a relatively undeveloped area (far from other roads). This is consistent for the three countries as well as for in or outside protected areas. If we compare between countries we can see that the probability of deforestation is always higher for Guatemala than for Mexico and Belize. Moreover, the probability of deforestation inside a protected area in Guatemala is close to the probability of deforestation outside one in Mexico. In the case of Belize, there is no clear difference in the probability of deforestation outside or inside of a protected area.

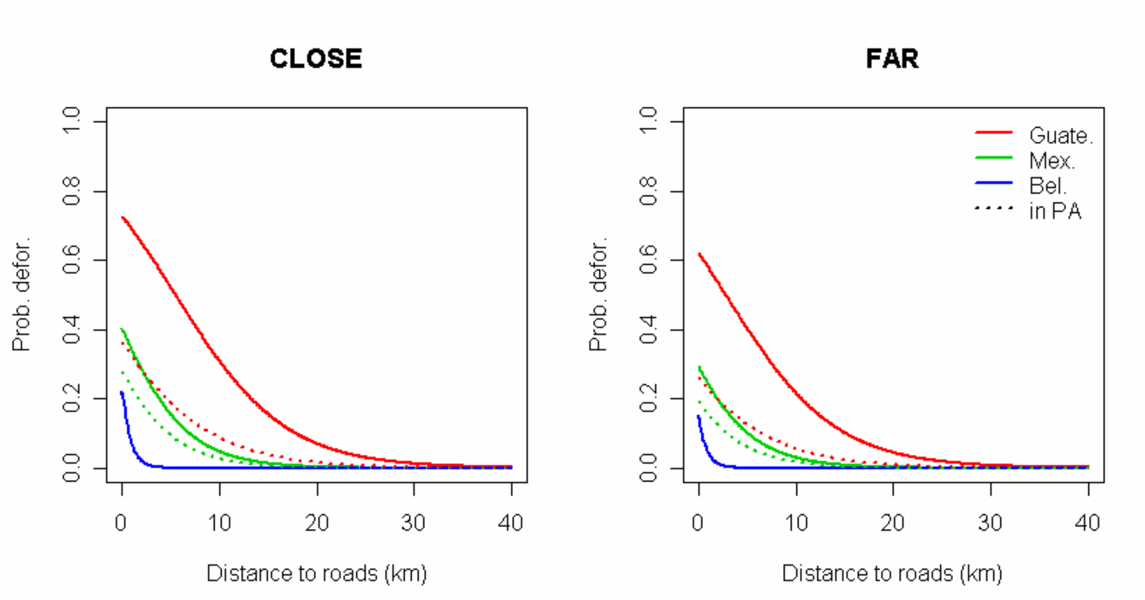

Figure 10. Probability of deforestation in: A) areas close to previous development and B) areas far from previous development.

The dotted lines show the probability of deforestation inside of a protected area (PA). We ran the model (table 13) controlling for an elevation of 50 m and a soil type C.

Forecasting the probability of deforestation

In the first scenario (A) we only considered the effect of roads on deforestation given the context of development (table 14). Under this scenario the Chetumal-Flores road has the lowest effect on deforestation when compared to the route through Belize. This is due to the fact that it will be built in a low developed area (figure 9). In the second scenario (Figure 10 B) we included the protected area variable; in this case we forecast an even lower impact from the Chetumal-Flores road than the Belizean route (Figure 10 B). However, in the third scenario we forecast a higher deforestation rate from the Chetumal-Flores route than the alternative route through Belize. This forecasting is based on the final deforestation model (Figure 10 C), and highlights the importance of considering all this components when modeling deforestation. We can observe that even if the road is placed in a highly developed area and outside a protected area, the country effect totally changes the prediction of deforestation (Figure 10 C). We can observe that while we would expect that deforestation will be lower inside of a protected area, when the road is placed in Guatemala, it does not matter that is inside of a protected area, the rate of deforestation is higher than placing a road outside of a protected area in Belize. As can be seen in figure 2, the probability of deforestation from 1990-2000 is higher in Guatemala, whether is inside or outside of a protected area.

Table 14. Forecasting scenarios

A dot shows the variables included in the model.

Variables	**Scenario A**	**Scenario B**	**Scenario C (coefficients in Table 2)**
Country (dummy for Mexico, Belize, Guatemala)			º
Elevation (m)	º	º	º
Soil	º	º	º
Distance to deforestation in 1980 -1990 (Km)	º	º	º
Distance to roads old = roads present until 2000 (Km)	º	º	º
Roads new (Km) = proposed roads (1. Belize route and 2. Chetumal- Flores route)	º	º	º
Level of development = close (≤ 25 km) or far (> 25km) from old roads	º	º	º
Protected Areas in 2000		º	
Protected Areas * Country			º
Country * Road distance			º
Roads new * level of development	º	º	º
AIC	**5323**	**5268**	**4930**

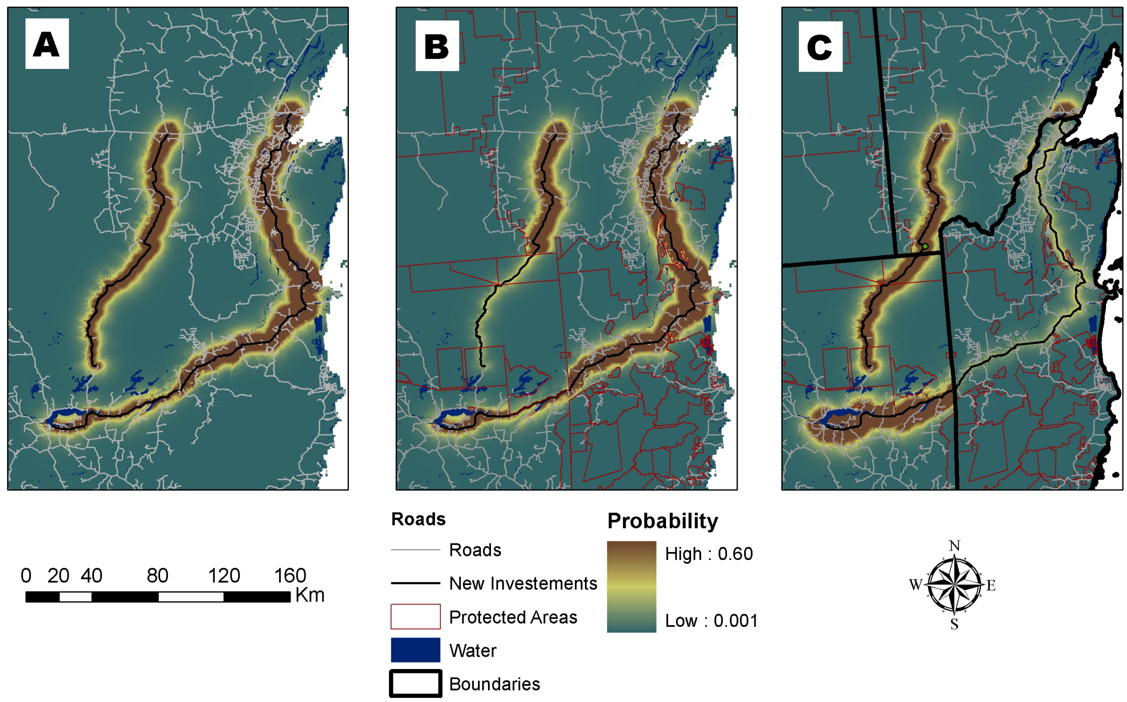

Figure 11. Scenarios of probability of deforestation from both road investments.

A) Probability of deforestation if we only consider that the new investments will be placed in areas with high or low development. B) Probability of deforestation if we consider as well the protected areas. C) Probability of deforestation if we consider the Protected Areas and the country variables.

Estimating deforestation

The cross-validation of the model (scenario C) with the area under the ROC curve had a maximum true positive of 0.8 rate and a false positive rate of 0.28 (figure 10A) for a probability of 0.057, indicating reasonable discrimination ability (Pearce and Ferrier 2000, Figure 4). Therefore, we used this threshold (Figure 3B) to generate a deforestation map and estimate the hectares that would be deforested from the road projects.

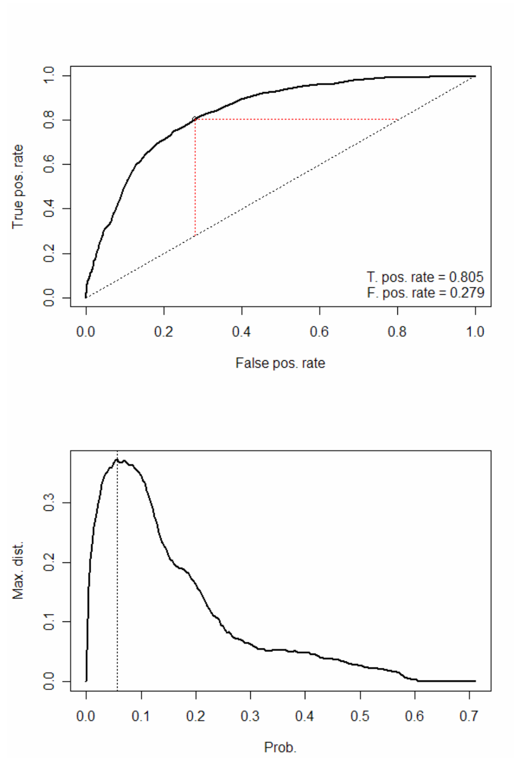

Figure 12. A) Area under the receiver operating characteristics (ROC) curve; B) Probability at which the ratio between true positive and false positive rates is maximized.

Approximately 22,964 hectares of forest will be lost from the Chetumal- Flores roads if we include both segments: Caobas- Arrollo Nergro (AN) and the AN-Tikal segments. However, the first segment of this project is almost finished, which means that the deforestation that policy makers can avoid by not developing the second segment will be of around 16,851 hectares for the first decade. This is more than six times the amount of deforestation that we can expect from the alternative route through Belize, which will generate only approximately 2,419 hectares. It is important to note that the Central American Development Bank has already invested in the upgrade of road that goes from El Remate to the Belize frontier, which will promote a significant amount of deforestation (Table 15). Therefore, the construction of the Chetumal-Flores road will generate a high impact in the region.

Estimating jaguar habitat loss

The construction of the road segment Arrollo Negro –Tikal will severely fragment the biggest continuous patch of the Mayan Forest, from two million hectares into two patches of approximately 414,800 and 1,562,500 ha, whether the Belize route will not have any effects on habitat fragmentation. For females, the Chetumal–Guatemala International (CGI) road will promote the 177,444 hectares in the first decade, compared with the 899 hectares that will be lost from the alternative route through Belize. The implementation of the Melchor de Mencos and Flores roads, which are necessary for both the CGI and the Belize route, will have a low impact on jaguar habitat loss (Table 15).

Table 15. Estimation of deforestation and jaguar habitat loss from road projects

Project	Segment	Type of investment	Project Status	Deforestation (has)	Jaguar Habitat Loss
Belize route	Belize route	Upgrading	Alternative	2519	899
Chetumal Guatemala (CGI)	Caobas-Arrollo Negro	Upgrading	In construction	6113	30,515
Chetumal-Guatemala (CGI)	Arrollo-Nergo Tikal	construction	Planned	16851	146,929
Roads necessary for both projects					
El Remate – Flores	Flores	Upgrading	Planned	33710	1442
Guatemala frontier Belize	Melchor Mencos - El Remate	Upgrading	Finished	23189	7789

Discussion

This is a study case of the temporal and spatial tradeoffs facing policy planners given two alternative road investments in the Mayan Forest. If the policy goal is to promote the transport sector while minimizing its impact on the tropical forest, policy makers have two clear choices. The first one is to invest in the CGI road that crosses the Biosphere Mayan Reserve which will be constructed in a previously undeveloped area. The second choice is to invest in the Belize route, which is an already developed area. Based on the labor and capital necessary to settle in the tropical forest we expected that in the short term the CGI will have a lower impact than the Belizean route. However, if we include the possible effect of protected areas, the impact may be even lower, this could be a statistical artifact since really few roads are built in protected areas, but with some degree of protection we can expect lower deforestation probability. However, if we consider the impact on species habitat and the fragmentation of the largest patch of tropical forest, it is clear that the CGI road is a bad option.

The impact of the CGI versus the route through Belize, dramatically changes when we consider the country effect. Although in the short term roads have a higher impact when they are placed in an already-developed area, the magnitude of this impact drastically changes from one country to another. Our results show that the country is a significant determinant on the impact of a road investment. The country and state covariates for 1990 to 2000 deforestation as well reflect the effects of the country and state policies during that decade. As we discussed in chapter II, the change in policies in Belize towards an ecotourism industry have slowed down deforestation, meanwhile Guatemala has shown a constant increase in the deforestation rates. This is mainly because the Petén has been the receptor of landless people looking for subsistence agriculture, since the bust in cotton prices and other products in the South of Guatemala left a high percentage of the population without jobs (Grandia 1992). The migration to the Petén was as well enhanced by the discovery of oil and by the repatriation process of political refugees from the Guatemalan civil war (sees Chapter II).

Our findings show the importance of the impact of roads on deforestation given the context of previous development and the country. When including the country variable it is clear that the CGI road will have a higher impact on deforestation than the route through Belize. This is not surprising since Guatemala has shown the highest deforestation rates in the last decades due to its increasing migration towards the forest (Chapter I, FAO 2005) and the illegal settlements inside the MBR (Parks Watch 2006). Moreover, the CGI road will be determinant of the long term future path of development and deforestation in the Mayan Biosphere Reserve. In the long term, we can expect that the impact of this road will increase by promoting development in the region, which as a result will promote new roads by providing political and economic incentives for further investments (Chomitz and Gray 1996; Croopper, et al. 2001; Geist and Lambin 2002).

However, the impact of a road on deforestation is only one of the effects, and the easiest to measure, on tropical forest. It is clear that the CGI road that crosses the Mayan Biosphere reserve will have the highest impact on the species habitat since it will fragment the biggest patch of the Mayan Forest. Nevertheless, it is essential to estimate the extent of its effect on the species habitat. We found that for female jaguars the CGI road will promote the loss of approximately 177,444 ha of habitat versus the 899 ha from the Belize route. However, the first segment of the CGI is almost finished. Therefore, policy planers will only be able to mitigate the deforestation and habitat loss from the second segment. However, in order to decide how to mitigate the impact of road investments our results show that considering only the impact on deforestation is somewhat misleading. For example by investing in the route through Belize instead in the CGI road it will avoided the deforestation of 16,851 has in 10 years. On top of that, if we take into account the loss of habitat for species such as jaguars, the number of hectares that will be avoided from degradation increases to 146,929. This clearly shows the importance of measurements other than deforestation to assess the impact of infrastructure projects in the tropical forests.

Our model is conservative since we used the deforestation dynamics in the 2000 to project the probability of deforestation from the alternative road investments. However, its impact will vary since due to changes in National policies and Macro-level dynamics that will affect the migration patters, subsidies among other issues that will have an effect on deforestation or due to changes in agricultural and timber prices among others. However, this model is a good approach to estimate the probability of deforestation, which controls for the country variable and the development context. Moreover, we estimated as well road impact on a species habitat loss and fragmentation.

Road impacts on habitat quality and fragmentation should play a key role in decisions on road locations. Consideration of this factor will reduce future conflicts between wildlife and people. The access that roads provide to the forest inevitably increases the hunting of large carnivore prey by people (Wilkie 2000). Habitat loss and fragmentation also reduce jaguar prey abundance and diversity, which are factors that has shown to promote highest cattle depredation rates by jaguars (Polisar, et al. 2003). Therefore, the alternative route through Belize will represent a better investment, which will avoid further deforestation, and the loss of further jaguar and other species habitat.

What other issues do policy planners face? The route by Belize, will only be 2% longer than the one proposed by the CGI and the costs will be certainly lower since it will only be improvement of the existing road. The section that connects the Guatemalan side with this road has been already improved with a loan from the Centro American Development Bank (BCIE 2005). Therefore, from the costs and environmental perspective it seems logical to invest in this road rather than continue with the development of the CGI road. However, the CGI road avoids the transit through Belize allowing a direct connection between Quintana Roo, Mexico to Guatemala, this is considerably important since both countries have signed a free trade agreement with the USA, and Belize is not yet part of CAFTA. The route through Quintana Roo toward Central America is important, since will place a direct route toward Florida. Florida has been named the main investment gateway to the CAFTA countries since about 300 multinational firms have their Latin American & Caribbean regional Head Quarters in Florida. In all, some 2,000 companies based outside the U.S. operate in Florida (CIC 2008). However, the investment of the CGI road will have a high cost globally and locally due to deforestation and species habitat loss and fragmentation. The alternative route through Belize will result in lower deforestation and habitat loss, but will necessarily involve a third party in trading negotiation. However, it is essential that the policy planners consider that Mexico and Central America have the highest deforestation rate in the Western Hemisphere (FAO 2005) and that the Mayan Forest is the largest remnant tropical forest in the region. The costs of including Belize in negotiations appear small relative to the important environmental services that the Mayan Forest provides.

5. General Conclusion

Our results show road investments in the Mayan Forest have an impact on deforestation that last for at least two decades, independently of the country. However, the magnitude of the impact will be different given its context such as the country and the level of development in the area. We found that a road placed into a previously undeveloped area will be *the* only determinant of the long term future path of development and deforestation of the Mayan Forest. This road will shape the new forest frontier, even if, in the short term, its impact it is expected to be lower than a road placed in an already developed area. This could be related with the high cost of clearing primary forest, and the lag effect of road construction on subsequent colonization. When a road enters a less developed or pristine area, the labor and capital required to carry out all of the land-cover change that may suddenly be economically worthwhile are not present. In contrast, in locations where some economic activity and forest clearing have already occurred, in the short term, a new road will promote higher deforestation since the rise in profitability due to a change in transport cost may more quickly be responded to and thereby may generate more additional deforestation in the first decade after the new road investment. Nevertheless, this does not implies that roads placed in low development areas have a lower impact on deforestation, in the long term we can expect that the impact of roads in these areas will increase by promoting development in the region. This will promote new roads by providing political and economic incentives for further investments.

On the other hand, we found that even if the magnitude of road investments in the forest frontiers is higher when a road is placed in an empty area, the magnitude of its impact drastically changes given the country of investment. Our results show that a road placed in Belize will have a lower impact on deforestation than a road placed in Guatemala or Mexico, even if these differences of the development are true for the three countries. These results are consistent with what we would expect to be the indirect and direct effects of national and state policies, and local institutions on deforestation. Therefore, our results show that a regional analysis should incorporate in their models the interaction of road investments and the country effects. However, to assess only the impact of road on deforestation it is insufficient to understand the extent of its impact on the tropical ecosystems.

We analyzed the impact of road investments on the jaguar habitat as a proxi of road impact on the degradation of tropical forest. Our results show that jaguars are sensitive to the presence of roads and the infrastructure that they convey. Moreover, we found that the effects of roads differ by gender, being females considerably more vulnerable to the proximity of roads. We also found the largest home range for jaguars reported in the literature, ranging from 122 to 970 Km^2. This implies the need of larger habitat patches for the species than the ones previously reported. Our reconstruction of the species habitat in 1980's, 1990's and early 2000's shows the dramatic impact of roads and subsequent deforestation on jaguar habitat. In only two decades, females lost 34% of their habitat and males 22%, while fragmentation doubled. This results show the high impact of road investments from 1980-2000 in the Mayan forest.

We found that if further road investments are developed to connect Quintana Roo and Guatemala, the best alternative will be to improve the exiting route through Belize. This could be counterintuitive since the route through Belize will be placed in an area that has been already developed against the CGI road that will be in the forest frontier. However, even if we previously found that a road placed in an already developed area has higher impact on deforestation, its magnitude could drastically change depending on the country. Therefore, a road placed in the forest frontiers in Guatemala will have a higher impact than a road placed in an already developed area in Belize. Moreover, the impact of the CGI is even higher when we analyze its effect on jaguar habitat. From both analyses, namely deforestation and habitat loss of an umbrella species, we can conclude that to invest on the route trough Belize will have the lower impact on the Mayan Forest ecosystem.

Our study is the first to analyze the impact of road investments on both deforestation and habitat loss for an important endangered species. Further, it forecasts both impacts from alternative planned projects. However, our models can be further improved in several ways: a) by controlling for roads endogeneity, with other methods than the timing of road placement and deforestation; b) by particularly assessing the effectiveness of protected areas in the region, on both reducing deforestation and protecting species habitat; c) by incorporating other variables that clearly explain the effects of each country on deforestation: such as demographics, income, education, etc; and c) by incorporating in the model other species' habitat, in this particularly case other than jaguars. Further development these models will provide a better understanding of the extent and magnitude of the impact of conservation and development policies on tropical forest. However, this study is the first step to have a broader understanding of the implications of two alternative policies, such as deciding between alternative roads.

To incorporate both perspectives will be especially important for the formulation of programs such as REDD, whose goal is to develop policy approaches and incentives to reduce emissions from deforestation and degradation in developing countries (Gullison *et al.* 2007). Therefore, by assessing the impact of alternative infrastructure projects on deforestation we can estimate which type of policies will reduce emissions from deforestation. Likewise, understanding its effects on a species habitat could be a valuable proxy to determine the impact of either policy on the ecosystem degradation. Because they are the target of so many infrastructure projects, conservation of remaining tropical forest will depend on additional studies of this kind.

References

Achard, F., H. D. Eva, et al. (2002). "Determination of Deforestation Rates of the World's Humid Tropical Forests." Science 297(5583): 999-1002.

Akaike, H. (1974). "A new look at the statistical model identification. IEEE." Transactions on Automatic Control (AC 19): 716-723.

Amor, D., I. Burgues, et al., Eds. (2007). Analisis Ambiental y Economico de Proyectos Carreteros en la Selva Maya. Serie Tecnica,. San Jose Costa Rica, Conservation Strategy Found.

Andersen, L. E. et al. 2002. The Dynamics of Deforestation and Economic Growth in the Brazilian Amazon, Cambridge Univ. Press.

Aranda J.M. 1990. [The jaguar (Panthera onca) in the Calakmul Reserve: morphometrics, food habits, and population density.] M.S. thesis, Univ. Nacional, Herdia (in Spanish).

Augustin, N. H., M. A. Mugglestone, et al. (1996). "An autologistic model for the spatial distribution of wildlife." The Journal of applied Ecology 33: 339-347.

Bawa, Kress, et al. (2004). "Tropical Ecosystems into the 21th Century." Science 306: 227-230.

Bartra, A., (2002), Mesoamerica, Los Rios Profundos, 2ª edition, Instituto Maya, , El Atajo, Casa Juan Pablos and UNORCA editors, Mexico City, pp. 396.

BCIE 2005, Banco Centroamericano de Integración Económica, Guatemala, Inversiones, [http://www.bcie.org/spanish/publicaciones/memorias/2005/guatemala.pdf, consultado Noviembre 2006]

Beyer, H. L. (2004). "Hawth's Analysis Tools for ArcGIS." Available at http://www.spatialecology.com/htools.

BID, B. I. d. D. (2005). Red Internacional de Carreteras Mesoamericanas (RICAM), Presentación: Plan Puebla. Ciudad de Panamá, Panamá.

BID. (2003). "(Project GU-0171) Plan Puebla Panama Guatemala-Mexico Electricity Interconnection Project, Loan Proposal."

Bolland, N. O. (1986). Belize, a New Nation in Central America, Westview Press, Boulder Co. USA. pp. 157

Boyce, M. S., P. R. Vernier, et al. (2002). "Evaluating resource selection functions." Ecological Modelling 157(2-3): 281-300.

Burgués, I. (2005). Inventario de Proyectos de Infraestructura en Mesoamerica. San José, Costa Rica, CFS: 41.

Ceballos, G., C. Chávez, et al. (1999). Tamaño poblacional y conservacion del jaguar en la reserva de la Biosfera de Calakmul, Campeche, México. In Jaguars in the new millennium. A status assessment, priority detection and recommendation for the conservation of jaguars in the Americas. R. A. Medellin, C. Chetkiewicz, A. Rabinowitzet al. Mexico, Universidad Nacional Autonoma de Mexico/Wildlife Conservation Society.

CFE (2004). Gerencia de Programación de Sistemas Eléctricos. Programa de Obras e Inversiones del Sector Eléctrico 2004 – 20013., Comisión Federal de Electricidad, Subdirección de Programación.

Chomitz, M. K. and D. A. Gray (1996). "Roads, Lands, Markets and Deforestation, A spatial Model of Land Use in Belize." World Bank Economic Review 10(3): 487.

CIC, 2008, The CAFTA Intelligence Service, http://www.caftaintelligencecenter.com/. consulted May 2008

Coppolillo, P., H. Gomez, et al. (2004). "Selection criteria for suites of landscape species as a basis for site-based conservation." Biological Conservation 115(3): 419-430.

Corzo, M. (2001). Estado Socioeconómico del Parque Nacional Laguna del Tigre Hasta el Año 2000. Guatemala PROPETEN.

Crawshaw, P. G. J. and H. B. Quigley (1991). "Jaguar spacing, activity and habitat use in a seasonally flooded environment in Brazil." Journal of Zoology 223(3): 357-370.

Croopper, M., J. Puri, et al. (2001). "Predicting the Location of Deforestation: The Role of Roads and Protected Areas." Land Economics 77(2): 172-186.

Cropper, M., C. Griffiths, et al. (1999). "Roads, Population Pressures and Deforestation in Thailand, 1976-1989." Land Economics 75(1): 58-73.

de Azevedo, F. C. C. and D. L. Murray (2007). "Spatial organization and food habits of jaguars (Panthera onca) in a floodplain forest." Biological Conservation 137(3): 391-402.

Economist (2008). A tale of two Mexicos, North and Shouth: Why can't the stagnant souther statest catch up with the rest of Mexico? The Economist. London: 53-54.

Emmons, L. H. (1986). "Comparative feeding ecology of felids in a neotropical rainforest." Behavioral Ecology and Sociobiology 20(4): 271-283.

Fa, J. E., G. Yuste, et al. (2000). "Bushmeat Markets on Bioko Island as a Measure of Hunting Pressure." Conservation Biology 14(6): 1602-1613.

FAO (2005), Global Forest Resources Assessment 2005, United Nations, Rome.

Forman, R. T. T. and L. E. Alexander (1998). "Roads and their Major Ecological Effects " Annual Review of Ecology and Systematics 29(1): 207-231.

Fuller, T. K. and P. R. Sievert (2001). Carnivore demography and the consequences of changes in prey availability. Carnivore conservation. J. L. Gittleman, S. M. Funk, D. Macdonald and R. K. Wayne. Cambridge, Cambridge University Press: 163–178.

Galletti, H. (1998). The Mayan Forest of Quintana Roo: Thirteen Years of Conservation and Community Development, In Primark, R. B., Bray, D., Galletti, H. and Ponciano, I. (Eds): *Timber Tourist and*

García, G. and F. Secaira (2006). Una visión para el futuro: Cartografía de las Selvas Maya Zoque y Olmeca: Plan Ecorregional de las selvas Maya, Zoque y Olmeca. San José, CR PPY- TNC. TNC Infoterra Editores.

Geist, H. J. and E. F. Lambin (2002). "Proximate Causes and Underlying Driving Forces of Tropical Deforestation." BioScience 52(2): 143-150.

Grandia L. (2000). *Cuantas personas cree usted que vivan en el Peten?*, In FLACSO (Ed), Nuevas Perspectivas de desarrollo sustentable en el Peten. FLACSO, Guatemala, pp. 137-156

Gullison, R. E. and J. J. Hardner (1993). "The effects of road design and harvest intensity on forest damage caused by selective logging: empirical results and a simulation model from the Bosque Chimanes, Bolivia." Forest Ecology and Management 59(1-2): 1-14.

Hespenheide, H. A. (1975). Prey characteristics and predator niche width. Pages 158-180 in M. L. Cody and J. M. Dia- mond, editors. *Ecology and evolution of communities*. Harvard University Press, Cambridge, Massachusetts, USA.

HEED (2000). No Gold Mining in the Rainforest! The International Law Project; A project of the , . Human, Economic & Environmental Defense. Los Angeles., Human Economic and Environmental Defense, National Lawyers Guild: Online at: http://www.heed.net/bill/cr/gold4.html.

Holdridge, L. R., W.C. Genke, W.H. Hatheway, T. Liang, and J.A. Tosi Jr. (1971). Forest environments in tropical life zones: a pilot study. P. Press. Oxford, England.

Jensen, J. R. (2000). Remote Sensing of the Environment: An Earth Resource Perspective. Saddle River, N.J.

IPCC, 2001, Climate change (2001). Synthesis report. Contribution of working groups I, II, III, to the 3rd assessment report of the Intergovernmental Panel on Climate Change, editors R. T. Watson and Core writing team. Cambridge University Press, Cambridge.

Kaimowitz, D. and A. Angelsen (1998). Economic Models of Tropical Deforestation: A Review. Bogor, Indonesia, CIFOR.

Kaiser, J. (2002). "FOREST ECOLOGY: Satellites Spy More Forest than Expected." Science 297(5583): 919-.

Kinnaird, M. F., E. W. Sanderson, et al. (2003). "Deforestation Trends in a Tropical Landscape and Implications for Endangered Large Mammals." Conservation Biology 17(1): 245-257.

Klar, N., N. Fernandez, et al. (2008). "Habitat selection models for European wildcat conservation." Biological Conservation 141(1): 308-319.

Kerley L., J. M. Goodrich, et al. (2002). "Effects of Roads and Human Disturbance on Amur Tigers." Conservation Biology 16(1): 97.

Maffei, L., E. Cuellar, et al. (2004). "One thousand jaguars (Panthera onca) in Bolivia's Chaco? Camera trapping in the Kaa-Iya National Park." Journal of Zoology 262(3): 295-304.

Malcolm, J. R. and J. C. Ray (2000). "Influence of Timber Extraction Routes on Central African Small-Mammal Communities, Forest Structure, and Tree Diversity." Conservation Biology 14(6): 1623-1638.

McGarigal, K., S. A. Cushman, et al. (2002). "FRAGSTATS: Spatial Pattern Analysis Program for Categorical Maps. Computer software program produced by the authors at the University of Massachusetts." Amherst. Available at the following web site: www.umass.edu/landeco/research/fragstats/fragstats.html.

McLellan, B. N. and D. M. Shackleton (1988). "Grizzly bears and resource extraction industries: effects of roads on behavior, habitat use, and demography." Journal of Applied Ecology 16 451– 460.

Medellin, R. A., C. Chetkiewicz, et al. (2000). Jaguars in the new millennium. A status assessment, priority detection and recommendation for the conservation of jaguars in the Americas. Mexico, Universidad Nacional Autonoma de Mexico/Wildlife Conservation Society.

Mena-Rivero, R., E. O. Ruiz-Martinez, et al. (2004). Evaluación Social del Proyecto "Modernización del Tramo Carretero Caobas-Arrollo Negro" en el Municipio de Othón P. Blanco, Quintana Roo. Obra que forma parte del proyecto carretero Caobas, México-Flores, Guatemala. Chetumal, Quintana Roo, Secretaria de Comunicaciones y Transportes, Gobierno del Estado de Quintana Roo: 76.

Miller, K., E. Chang, et al. (2001). Defining Common Ground for the Mesoamerican Biological Corridor. Washington DC, USA, , World Resources Institute,: 45.

Mills, S. L., M. E. Soulé, et al. (1993). "The Keystone-Species Concept in Ecology and Conservation." BioScience 43(4): 219-224.

Myers, N. (1992). The Primary Source. NY, Norton.

Myers, N., R. A. Mittermeier, et al. (2000). "Biodiversity hotspots for conservation priorities." Nature 403: 853-854.

Nepstad, D., G. Carvallo, et al. (2001). "Road paving, fire regime feedbacks, and the future of Amazon forests." Forest and Ecology Managment 154: 395-407.

NFN (1997). Chainsaw Concessions Threaten Primary Rainforest in Nicaragua. Native Forest Network Action Alerts and Info, . Native Forest Network., Naitive Forest Network: No. 7, October, Online at:http://www.nativeforest.org/alerts/alert11.html.

Noss, R. F., H. B. Quigley, et al. (1996). "Conservation Biology and Carnivore Conservation in the Rocky Mountains." Conservation Biology 10(4): 949-963.

Novack, A. J., M. B. Main, et al. (2005). "Foraging ecology of jaguar (Panthera onca) and puma (Puma concolor) in hunted and non-hunted sites within the Maya Biosphere Reserve, Guatemala." Journal of Zoology 267(2): 167-178.

Nunez, R., B. Miller, et al. (2000). "Food habits of jaguars and pumas in Jalisco, Mexico." Journal of Zoology 252(3): 373-379.

Parks Watch. 2004. Parks Watch. Guatemala Park Profiles. http://www.parkswatch.org/parkprofile.php?l=eng&country=gua. Accessed on January 15, 2008.

Pearce, D. (2002). "ENVIRONMENT: Gold from Green Paths." Science 297(5583): 941-.

Pennington, T. D., and J. Sarukhan (1968). Arboles tropicales de Mexico. Mexico, Distrito Federal, Mexico, Instituto Nacional de Investigaciones Forestales.

Pfaff, A. et al. (2008). The Impacts of Roads in the Process of Deforestation. Amazonia and Global Change. M. Keller, J. Gash and P. Silva Dias. in prep.

Pfaff, A. (1999). "What Drives Deforestation in the Brazilian Amazon?" Journal of Environmental Economics and Management 37: 26-43.

Pfaff, A. et al (2006). "Econometric Estimation of Deforestation Impacts from Roads and Other Drivers". Conference talk, NASA-MCT LBA Project, Brasilia, October.

Pfaff, A., A. Barbieri, et al. (2008). The Impacts of Roads in the Process of Deforestation. Amazonia and Global Change. M. Keller, J. Gash and P. Silva Dias. in prep.

Pfaff, A., J. A. Robalino, et al. (2007). "Road Investments, Spatial Intensification and Deforestation in the Brazilian Amazon." Journal of Regional Science 47: 109-123

Pfaff, A., R. Walker, et al. (2007). "Roads and Deforestation in the Brazilian Amazon." submitted.

Polisar, J., I. Maxit, et al. (2003). "Jaguars, pumas, their prey base, and cattle ranching: ecological interpretations of a management problem." Biological Conservation 109(2): 297-310.

Primarck, R. B., D. Bray, et al. (1998). Timber Tourists and Temples. Washington D. C, Island Press.

Rabinowitz, A. R. (1986). "Jaguar Predation on Domestic Livestock in Belize." Wildlife Society Bulletin 14(2): 170-174.

Rabus, B., M. l. Eineder, et al. (2002). "The shuttle radar topography mission—a new class of digital elevation models acquired by spaceborne radar." Journal of Photogrammetry and Remote Sensing 57(4): 241-262.

Rhodes, J. R., T. Wiegand, et al. (2006). "Modeling Species' Distributions to Improve Conservation in Semiurban Landscapes: Koala Case Study." Conservation Biology 20(2): 449-459.

Riverson, J., J. Gaviria, et al. (1991). Rural roads in sub-Saharan Africa: lessons from World Bank experience. . Washington, D.C, The World Bank.

RUCIA (2002). Parque Industrial y Pesquero de Puerto Morelos S.A. de C.V. Estudio de Factibilidad de Mercado, Mexico D.F., Ruiz, Urquiza y Compañía, S.C., http://www.economia.gob.mx/pics/p/p2757/potencial_pymes_QROO.pdf: 11-12.

Schaller, G. B. and P. G. Crawshaw, Jr. (1980). "Movement Patterns of Jaguar." Biotropica 12(3): 161-168.

Silver, W. L., R. Ostertag, et al. (2000). "The Potential for Carbon Sequestration Through Reforestation of Abandoned Tropical Agricultural and Pasture Lands." Restoration Ecology 8(4): 394-407.

Soisalo, M. K. and S. M. C. Cavalcanti, (2006). "Estimating the density of a jaguar population in the Brazilian Pantanal using camera-traps and capture-recapture sampling in combination with GPS radio-telemetry." Biological Conservation 129(4): 487-496.

Sutherland, A., (1998). The Making of Belize: Globalization in the Margins, Westport, Conn.: Bergin & Garvey, pp. 202

Spellerberg I. F., (2002). *Ecological Effects of Roads*. Science Publisher, Inc. New Hampshire, pp. 248.

Taber, A. B., A. J. Novaro, et al. (1997). "The Food Habits of Sympatric Jaguar and Puma in the Paraguayan Chaco." Biotropica 29(2): 204-213.

Thiel, R. P. (1985). "Relationship between road densities and wolf habitat suitability in Wisconsin." American Midland Naturalist 16: 404– 407.

Verburg, P. H., K. P. Overmars, et al. (2004). "Accessibility and land-use patterns at the forest fringe in the northwestern part of the Philippines." The Geographical Journal 170(3): 238-255.

Weckel, M., W. Giuliano, et al. (2006). "Jaguar (Panthera onca) feeding ecology: distribution of predator and prey through time and space." Journal of Zoology 270(1): 25-30.

Westphal, M. I. and H. P. Possingham (2003). "Applying a Decision-Theory Framework to Landscape Planning for Biodiversity: Follow-Up to Watson et al." Conservation Biology 17(1): 327-329.

Wikramanayake, E. D., E. Dinerstein, et al. (1998). "An Ecology-Based Method for Defining Priorities for Large Mammal Conservation: The Tiger as Case Study." Conservation Biology 12(4): 865–878.

Wilkie, D., E. Shaw, et al. (2000). "Roads, Development, and Conservation in the Congo Basin." Conservation Biology 14(6): 1614-1622.

Yuan, D., Elavidge, C. D., and Lunetta, R. S., (1998), Survey of multispectral methods for land cover change analysis. *In Remote Sensing Change Detection: Environmental Monitoring Methods and Applications*, edited by R. S. Lunetta and C. D. Elvidge, Chelsea, MI: Ann Arbor Press, pp. 21–39.

Biography

Dalia Amor Conde, native from Mexico obtained a bachelor degree in Biology from the National Autonomous University of Mexico (UNAM) in 2000. From 1996-2000 she worked as director of environmental conservation projects in a Mexican NGO, Unidos para la Conservacion. Among the projects she worked are: the pronghorn reintroduction program, the bighorn sheep conservation project in Tiburon Island and the community development project in the Caobas ejido of the Mayan forest. Before starting her PhD she worked in the Ministry of the Environment of Mexico City. Dalia as well worked with the indigenous community of Mazatlan Villa de Flores in Oaxaca, supporting the forest conservation and shade tree coffee project, and the development of the first high school in the town. Dalia has as well conducted research across Southern Africa and has had field research training in India. In 2003 she obtained the American Association of University of Women International Award and in 2005 she received the WINGS WORDL QUEST field research award. Dalia as well was invited to join the Satyagraha initiative from the Garrison Institute. Dalia has obtained financial support for her research and applied projects from CONACYT, The Mesoamerican Biological Corridor Initiative of the World Bank, the National Fish and Wildlife Foundation, CONABIO, Conservation Strategy Found, Safari Club International, World Wild Life Found. Dalia is leaving to Rostock Germany to pursue post doctoral research at the Max Plank institute of demographic research.