The Impact of Deforestation on Nature-Based Recreation: Evidence from Citizen Science Data in Mexico

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Abstract

We use citizen science eBird data to investigate the impact of deforestation on birdwatching across Mexico. Utilizing detailed data on individual trips reported to eBird, we construct annual count data on visits to 1,810 Mexican municipalities from 2008 to 2016 to examine how deforestation affects visitation patterns. We find that a municipality, experiencing an increase in deforestation equivalent to the average deforestation over our study period, would see birdwatching visits decrease by 4.4% per year. The results also provide an application of citizen science data using an underutilized source of information on economic behavior in an otherwise limited data environment.

Appendix materials can be accessed online at:
https://uwpress.wisc.edu/journals/pdfs/LE-98-1-02-Chen-app.pdf.

1. Introduction

The presence of forests offering aesthetic and recreational opportunities attracts tourists and generates revenue, yielding considerable benefits to local governments, firms, and communities (Edwards et al. 2012; Zhang 2016.). Because tourism in many developing contexts depends on forests, nature-based recreation is often hailed as a way to promote sustainable development (Gurung and Seeland 2011). Given the increasingly important role of nature-based recreation and associated tourism in many economies (e.g., Naidoo and Adamowicz 2005a, 2005b; Hutchins 2007; Bhuiyan et al. 2011; Ramos and Prideaux 2014; Hunt et al. 2015; Eshun and Tagoe-Darko 2015; Liang 2016; Snyman 2017), research is needed to better understand how the characteristics of forest ecosystems affect individuals' decisions on where and how much to recreate. Understanding the drivers of recreation and tourism choices provides important insights into the role of sustainable development in seeking to balance economic growth and the preservation of key ecological features making locations attractive for tourism.

A large number of studies in the natural sciences establish that deforestation and forest degradation result in decreases in traditional forest-dwelling species richness and abundance and in increases in invasive and introduced species (e.g., Fahrig 2003, 2017; Laurance et al. 2014; Ochoa-Quintero et al. 2015; Giam 2017).[1] A handful of studies focusing on the economic impact of biodiversity (broadly defined) on nature-based recreation suggest tourists highly value biodiversity (Naidoo and Adamowicz 2005b; Naidoo et al. 2011; Booth et al. 2011; Kolstoe and Cameron 2017). In the specific context of birdwatching, studies show that nature-based recreators value the presence of a greater number of species (e.g., Kolstoe and Cameron 2017) as well as rare species (e.g., Booth et al. 2011). While there is a substantial body of theoretical and empirical literature on the impact of deforestation on species richness and abundance, very little is known about how large-scale deforestation and forest degradation affect nature-based recreation behavior, a topic particularly relevant in developing country settings.

We focus on the impact of deforestation on birdwatching in Mexico using an underutilized source of citizen science data to model birder behavior. Birdwatching has long been a pastime in developed countries where observing wild birds became popular in the late 19th century (Weidensaul 2008). Birdwatching is the fastest growing segment in the tourism market and generates significant revenue as birders are generally well educated with above-average incomes (Cordell and Herbert 2002; Şekercioğlu 2002; Lee et al. 2010; Steven et al. 2015; Kronenberg 2016; Ocampo-Penuela and Winton 2017). For example, the United States Fish and Wildlife Service reported that in 2011, American birders' total expenditure for birding was $40.94 billion, with $14.87 billion spent on trips (Carver 2013). According to the 2016 National Survey of Fishing, Hunting and Wildlife Associated Recreation (NSFHWAR), 45.1 million Americans aged over 16 reported themselves as wild bird observers in 2016, while 36% of them (16.3 million) took birding trips away from home for an average of 16 days a year.[2]

Many birders, known as "listers", keep track of their trips and the species they observe using online tools such as eBird, a checklist program launched by a large citizen-science project at the Cornell University Laboratory of Ornithology (Fink et al. 2017). This project is similar to many other volunteer environmental monitoring programs where citizens contribute to scientific research (Evans et al. 2005; Ricker et al. 2013; Luz et al. 2014; Stepenuck and Green 2015; Venturelli et al. 2017). With the growing popularity of birding, an increasing number of bird watchers are willing to travel long distances for new species that cannot be observed elsewhere. Estimates suggest the annual number of international trips worldwide with a main purpose of birdwatching is 3 million.[3] According to American Birding Association's survey, 49% of its members would like to travel outside the US to observe birds.[4]

In developing countries tourism is often viewed as a tool to simultaneously achieve economic development and conservation goals. Birdwatching in particular has become popular as it is considered a sustainable form of tourism (Biggs et al. 2011; Li et al. 2013), i.e. birders are generally interested in conservation (Hvenegaard and Dearden 1998; Connell 2009).[5] Mexico is the country most visited by U.S. nature-based tourists.[6] Largely influenced by the development of birdwatching activities in the U.S., observing birds as a pastime appeared in Mexico from at least the beginning of the 20th century (Gómez de Silva and Alvarado Reyes 2010). Considered one of the mega-diverse countries in the world (Llorente-Bousquets and Ocegueda 2008), Mexico is endowed with large areas of forests (Hendee et al. 2012). According to BirdLife International (del Hoyo and Collar 2014, 2016), Mexico is home to 10% of all bird species on the planet. Among these species, 10% are only observed in Mexico, making the country fifth in number of endemic bird species in the world.

As a result of extensive biodiversity and relative proximity to U.S. birders, Mexico is rapidly becoming one of the most popular destinations for birdwatching in the world (Revollo-Fernández 2015; Peterson and Navarro-Sigüenza 2016). In the eBird database, checklists that reported a birdwatching trip in Mexico increased from 8,366 entries to 33,708 between 2012 and 2016. At the same time, Mexico is also experiencing widespread deforestation: the country lost 5.8% of its forest coverage between 2000 and 2016 with a total loss area over 30,000 km^2, which is equivalent to 1.52% of the total land area of the country (Hansen et al 2013; calculations by authors). The combination of increasing nature-based tourism and at-risk natural resources makes Mexico an ideal context to investigate the impact of deforestation on birdwatching.

To estimate the effects of deforestation on birdwatching visits in Mexico, we combine citizen science data (the eBird database) with high-resolution satellite data of forest coverage (Hansen et al. 2013). Using the eBird data, we construct annual visitation counts for 1,810 Mexican municipalities[7] from 2008 to 2016 to examine how changes in forest conditions affect birdwatching visitation patterns. To empirically estimate this impact, we first consider binary probability models to examine the role of deforestation in altering visitation patterns along the extensive margin by modeling the probability that a municipality receives a non-zero number of birdwatching visits. We then use count data models to investigate the effect of deforestation on the intensive margin by modeling the number of birding visits a municipality receives. Our results indicate that deforestation not only decreases the probability of a municipality being visited by birders along the extensive margin, but also reduces the number of visits on the intensive margin for municipalities with birdwatching visits. Specifically, our results indicate that during the period 2008-2016, if a municipality loses one percent of its forest area, the number of birdwatching trips to that municipality would decrease by 29.4%, *ceteris paribus*.

This paper makes three primary contributions to the literature. First, we provide new insights into forest conservation and tourism in a developing country context. Although there have been many studies that evaluate the effects of tourism on forests, few focus on the quality of the forest as a driver of nature-based recreation. Several related papers focus on a single birdwatching hotspot (e.g., Naidoo and Adamowicz 2005a) or sites in a small region (e.g., Kolstoe and Cameron 2017). In this paper, we conduct a comprehensive empirical investigation of deforestation and birdwatching at the municipality level across Mexico over a 9-year period.

Second, we demonstrate the potential role of citizen science data to conduct empirical research in settings where data availability from traditional sources is often lacking or prohibitively expensive to obtain across large areas. Recent research has suggested that social media data can be used as an alternative to traditional surveys to better understand individual preferences for recreation, as there may be no significant differences between tourists' stated preferences in surveys and revealed preference information gleaned from social media content (Tenkanen et al. 2017; Hausmann et al. 2018). Several studies have compared the use of crowd-sourced data (i.e. data from cell phones, online forums, citizen science applications, etc.) with conventional data gathering approaches and found that crowd-sourced data performs similarly when modeling visitation behavior (Merrill et al. 2020; Martin et al. 2014; Papenfuss et al. 2015). In a developing country context, there is a significant potential for citizen science data to play an important role in empirical research as these data are readily available and a relatively inexpensive substitute for traditional on-the-ground data collection (e.g., Kolstoe & Cameron 2017; Roberts et al. 2017; Arcanjo et al. 2016; Keeler et al. 2015; Hale 2018; Spalding et al. 2017; Donahue et al. 2018). Our work provides an application of the use of citizen-science eBird data in Mexico to study nature-based recreation.

Lastly, we provide new information on the potential economic impacts of deforestation on nature-based recreation and tourism across Mexico by adding new information on the potential scale of tourism revenue loss. Our empirical results suggest that deforestation discourages birders from visiting a municipality in Mexico, reducing both the probability of visitation and the number of visits. The loss in tourism revenue associated with decreased visitation is likely to affect not only local economies but also government tax revenue, further reducing the budget for conservation and leading to potential negative feedback effects. Thus, our results have important implications for the design of development and forest conservation polices in tourism-dependent communities seeking to enhance sustainable development.

2. Data

The data we use to examine nature-based recreation behavior consists of citizen science eBird visitation reports, remote sensing data on forest coverage and deforestation, as well as a number of covariates that are hypothesized to impact visitation patterns. In what follows, we discuss each of these data sources and the representativeness of the citizen-science eBird data.

eBird Dataset

Curated by Cornell University, the eBird dataset (Fink et al. 2017) aggregates bird species reports from global birdwatching volunteers who use an online checklist protocol to report their species sightings. Although the dataset was launched in 2002, the early eBird data was limited until 2008 when the number of participants and observations increased rapidly, coinciding with the mass market release of smart phones, including the iPhone in 2007, that enabled birders to more easily report their sightings while in the field (Sullivan et al. 2009; Kolstoe and Cameron 2017). While eBird is a real-time online dataset, it also allows input of historic sightings that occurred before a participant began using the reporting tool. To aid in identification of trip locations, birders are able to pick from a number of "hotspots" that are public birding locations nominated by other eBird users. In addition, birders are also able to input their own location data and create new hotspots. To minimize inaccurate recall, we focus our use of the eBird records for visits occurring after 2008 as those are more likely to represent contemporaneous reports associated with visitation.[8]

Each checklist submitted to eBird includes detailed information for a birding trip, such as location, time, duration of observation period, distance traveled, as well as individual and species bird counts. Birders are asked to report the latitude and longitude of the observation location (starting position if travelling), which we use to link individual observations to specific municipalities in Mexico. We only consider coordinates on land and exclude sea tours, because we cannot unambiguously link these observations to municipalities.

Because the dates of the birding trips are reported in these checklists, the eBird database allows us to separate observations occurring on different days. The duration of the actual birdwatching period is recorded in the checklists and is usually several hours but never exceeds 24 hours. In other words, eBird users are required to submit a separate checklist each day with a corresponding location if they spent more than a single day partaking in birdwatching activity. As a result, a multiple-day trip would result in multiple checklists, enabling us to count each checklist as a day trip. If a birder reported several trips in different locations within a single day, we only consider the trip with the longest duration, the longest travel distance, or the largest number of birds observed, in that order[9]. During our study period, the eBird dataset contains information on 64,572 day trips occurring in Mexico (Figure 1). Most trips occurred in the southern part of the country, as much of the north is arid and supports fewer bird species.

We aggregate individual trips into an annual count of all day trips in each Mexican municipality and use this count as our primary measure of birdwatching in the econometric analysis that follows. The reasons we aggregate the data to a municipal level is twofold. First, a municipality is the smallest administrative unit in Mexico for which many important economic control variables are reported. Second, the eBird data is sparse across much of the country. At the municipality level, 77.59% of our municipality-by-year observations experienced zero visits. Smaller spatial units than municipalities will lead to a significant increase in increase in units with zero visits. If aggregated larger than municipality units, we increasingly smooth out variation in forest coverage needed for identification.

eBird Sample Representativeness

As we only observe a day trip if a birder reported it to eBird, the magnitude of recreation visits by eBird participants does not capture the total birding activity by all tourists. Examining this issue in more detail, several studies have assessed the spatial representativeness of the eBird data. Calculating birding days in a similar way to our study, Warnell (2019) compares the eBird and NSFHWAR (See Section I) data focusing on the southeastern US in 2011, where the latter survey provides the best available estimates of the total amount of recreational birding by state. Results show that birding days reported by eBird participants consistently account for 0.1%-0.5% of the overall NSFHWAR estimates across states, suggesting that birding activity captured by eBird data is roughly proportional to total visitation across different locations[10]. Providing a similar comparison for our Mexican study area, Cantú and Sánchez (2011) estimate that the country received 78,820 bird watchers in 2006. Meanwhile, the eBird data contains information from 215 unique observers visiting Mexico in 2006, accounting for 0.27% of Cantú and Sánchez's (2011) estimate of the total number of birders during that year. As the number of birders participating in eBird expanded rapidly beginning in 2008, we expect that the share of overall birders captured by eBird during our study period is likely significantly higher than in 2006.

One of the eBird project's goals is to maximize public participation, and the reporting platform is designed to appeal to the broader birding community rather than being restricted to professional experts

(Sullivan et al. 2014). This inclusive strategy may have a negative impact on the data quality in terms of the accuracy of observed bird species; however, our study focuses on visitation days and not specific recalls of species observed. According to the 2016 NSFHWAR, among 23.7 million away-from-home wildlife watchers in the US (aged over 16; 72% are bird watchers), individuals who are male, white, aged over 45, or have received a college education account for 67%, 95%, 53% or 72%, respectively. Moreover, the median income of these recreators is $79,000, which is higher than the contemporaneous national median of $71,000 in the US. Given the accessibility of eBird to diverse users, it is likely that our sample has similar demographic properties to those reported in NSFHWAR.

In addition to using the eBird data to examine visitation, we use the panel nature of the data to infer the population composition of eBird participants in our sample. The eBird checklists identified as day trips in Mexico during our study period contain 4,432 unique birders reporting 16,290 day trips in Mexico. Each of these birders has been to Mexico at least once but may also report birding trips to other locations. For robustness we also examine sensitivity to excluding resident eBird users. To remove resident birders, we identify all global day trips reported by each birder in our sample period and calculate the median longitude and latitude for each birder (Appendix A Figure A1). The resulting map illustrates that 61.2% of the birders' median coordinates are located outside of Mexico and that most of these are concentrated in the U.S. To identify resident eBird participants we examine the 38.8% of residents with median coordinates inside Mexico using two additional criteria: a) birders whose number of day trips located in Mexico exceeds those in other countries and b) birders who report trips in Mexico across multiple years. Using these criteria, we ultimately identify 12.9% of the eBird users in the original sample as residents, leaving a final sample of 3,860 international birders who report 19,893 day visits in Mexico.

Forest Covariates

We use the Hansen datasets on percent tree cover and annual deforestation (Hansen et al. 2013) to recover information on forests.[11] To convert the original percent tree cover in 2000 to percent forest cover in 2000, we use a cutoff of 25% to define forests (Sexton et al. 2015) and retain only deforestation events that occurred on forested cells in the base year of 2000. Using this data, we calculate annual forest loss area in each Mexican municipality from 2008 to 2016 (Figure 2).[12] As our primary variable of interest, we define deforestation as the ratio of deforested area in a year relative to municipality size. Municipality areas are calculated based on the GADM database of global administrative areas (GADM 2009).

Because the forest configuration is likely to affect biodiversity, we include covariates related to core and edge forests (Fahrig 2003; Fahrig et al 2017). Specifically, we construct a ratio of forest edge to core forest using a reclassified version of the Hansen data with 90 meters to define an edge (Davies et al. 2000); we calculate the forest edge and core areas using the Landscape Fragmentation Tool (LFT) (Vogt et al. 2007). The edge to core ratio is a measure of patch-level forest fragmentation and disturbance, which is known to affect patch-level biodiversity (Graham and Blake, 2001), with more fragmented forests having higher ratios. For our application, we expect that more fragmented forests will have fewer core and endemic species and may have more introduced or invasive species that may not be of value to birders (Krebs 1972; Fahrig 2003). However, forest edges also often provide easier viewing of species located in canopies that are difficult to observe when in the interior sections of forest tracts, making the overall effect an empirical question.

In addition to the edge to core ratio, we also calculate above-ground woody biomass (average density per municipality) based on Baccini et al. (2012). This variable indicates the presence of old growth (i.e. high carbon value) forests vs. younger less mature forests. We expect that older forests are likely to contain more species, especially rare species (Jenkins et al. 2013), making them more likely to attract birders.

Additional Covariates

In addition to forest covariates, we construct a series of additional covariates that are likely to affect birdwatching behavior. We divide these into groups including climate and geography covariates and economic and infrastructure covariates. To account for the impact of climate, we include measures of both precipitation and temperature. These are likely to play an important role in tourism behavior by affecting the demand for outdoor recreation as well as the presence and likelihood of observing certain bird species. The temperature data are sourced from the Climate Prediction Center and from National Oceanic and Atmospheric Administration (Fan and Van den Dool 2008). We focus on the temperature in January and July to distinguish seasonal effects. Precipitation data are obtained from Climate Hazards Group InfraRed Precipitation with Station data (CHIRPS, Funk et al. 2015). In addition, we construct an indicator variable to identify municipalities located along the coasts. These coastal regions contain beaches that likely attract a broad set of recreators, including birders.

We construct controls for economic and infrastructure factors from a variety of sources. At the municipality level, we control for GDP (in 2005, measured in US dollars) (INAFED 2005), and population (in 2005) (INEGI 2005). We further consider the length of railway (in 2000) (INEGI 2000), which mainly provides freight service, as a control for local economic activity.[13] We also include the proximity to large urban areas as those are more likely to be influenced by urbanization, potentially discouraging birders from visiting. Specifically, we control for the Euclidean distance to the nearest major urban center, which is defined as the geometric center of an urban area with population greater than 500,000 in 2000. Using this population threshold,[14] we calculate distance to the nearest of 17 urban centers based on a map of urban areas from the National Institute of Statistics and Geography of Mexico (INEGI by its name in Spanish: Instituto Nacional de Estadística y Geografía) (INEGI 2000). For transportation infrastructure, we measure distance to the nearest airport using the Mexican airport map in 1998 from the North American Transportation Atlas (Bureau of Transportation Statistics 1998). We also measure the length of highways (in 2014) (INEGI 2014).

To further control for tourist infrastructure, we consider the number of global chain-branded hotels in each municipality. The data is gathered from hotel chain websites accounting for 8 mainstream hotel chains including Best Western, Choice, Club Carlson, Hilton, Hyatt, Intercontinental, Marriott, and Wyndham. We identify whether there is an archaeological site or national protected area within a municipality based on the official maps from INEGI (INEGI 2000) and Mexico's National Commission of Natural Protected Areas (CONANP 2018). We only consider national protected areas that were created before 2000 to exclude any endogenous factors caused by later deforestation and tourism. Finally, we control for the ratio of forest ejido area to municipality size. An ejido is an area of communally owned land for agriculture and forestry. According to Bray et al. (2010), ejidos cover at least 69% of the forests in Mexico, and studies show that they correlate with a generally a positive impact on forest conservation while providing economic opportunities for local residents (Ellis and Porter-Bolland 2008; Cooper and Huff 2017). We utilize maps of common use lands published by Mexico's National Agrarian Registry to calculate the area of forest ejidos in each municipality (RAN 2019).[15]

Table 1 provides a list of all the variables and their descriptive statistics. On average, each of the 1,810 Mexican municipalities in our sample received 3.84 day trips per year. However, a significant number of municipalities, 77.59% of our municipality-by-year observations, experienced zero visits. Among the municipalities that received birdwatching visits, there is significant heterogeneity, with some having as many as 848 visits in a single year. Deforestation is also unevenly distributed across our sample. The highest annual deforestation in a municipality was 6.52%, while the average was 0.13% of land area. Moreover, 21.63% of the municipality-by-year observations did not experience any deforestation. These statistics are suggestive of the significant heterogeneity in our deforestation measure across municipalities and years which form the basis for our econometric analysis.

3. Empirical Models

To empirically model the impact of deforestation on birdwatching, we first estimate a binary probability model of eBird visitation. Using a logistic distribution, we calculate the extensive margin probability of a municipality receiving non-zero visits as:

$$P(y_{it} = 1|\mathbf{x}_{it}) = \frac{1}{1 + e^{-\mathbf{x}'_{it}\boldsymbol{\beta}}}$$

where y_{it} is a binary variable equal to 1 if a municipality i received at least 1 visit in a year t, $\mathbf{x}_{it}$ is a vector of the municipality characteristics that may affect the likelihood of receiving birding visits, and $\boldsymbol{\beta}$ is a vector of coefficients.[16] To aid in interpretation, we report the average marginal effects of the estimated coefficients. We calculate the marginal effect at each observation and use the mean of the estimated marginal effects for each municipality characteristic to reveal how the characteristic affects the probability of the municipality being visited.

In addition to modeling the probability of receiving visits, we use count data models to measure the intensive margin impact of deforestation on the number of visits occurring in a municipality. We consider several count data models to address both overdispersion in the data and the large number of zero visits observed across municipalities. Standard Poisson regression models may lead to inefficient estimates if the distribution of the count data is overdispersed (Gardner et al. 1995). To address the potential for overdispersion, negative binomial (NB) regression models introduce an idiosyncratic error term to capture unobserved characteristics (shown in the Appendix A).

Given the large number of municipalities with no day trip visits, we also estimate zero-inflated negative binomial (ZINB) models (Greene 1994), where visitation is modeled as a two-stage process by expressing the count variable y_{it} as a product of two latent variables

$$y_{it} = h_{it} y_{it}^{*}$$

where h_{it} is a binary variable, with 0 indicating y_{it} is a constant zero and 1 indicating y_{it} follows a discrete distribution where y_{it}^{*} follows a NB distribution. In the first stage, we use a logit model to capture the probability of y_{it} being a constant zero count, i.e. the municipality is not in birders' choice set. In the first stage logit model, a positive coefficient indicates that the corresponding characteristic increases the probability of receiving zero visits, denoted as q_{it}.

For second stage estimation, we assume y_{it} follows a NB distribution, as this distribution allows y_{it} to be zero. This is important in our application as some destinations are likely to receive zero visits despite being in birders' choice sets. To estimate the model, the distribution of y_{it} is given as follows:

$$P(y_{it} = 0|\mathbf{x}_{it}) = P(h_{it} = 0|\mathbf{x}_{it}) + P(h_{it} = 1, y_{it}^{*} = 0|\mathbf{x}_{it}) = q_{it} + (1 - q_{it})F(0) \quad [3]$$

$$P(y_{it} = k|\mathbf{x}_{it}) = (1 - q_{it})F(k), k \geq 1$$

where $F(\cdot)$ is the probability distribution of y_{it}^* and $\mathbf{x}_{it}$ is a vector of municipality characteristics. Embedding the features of the NB model that capture omitted unobservable characteristics, the ZINB model places additional weight on the probability of observing zero counts through the added binary process to address excess zeros. Consequently, the ZINB model is less likely to suffer from overdispersion, resulting in more efficient estimates.

The magnitudes of the coefficients in count data models cannot be as directly interpreted as those in the binary models. To interpret these coefficients, we calculate the incidence rate ratio (IRR) of the estimates to illustrate the effects of each independent variable on the visitation counts. When the kth municipality characteristic increases by a unit, the expectation of the change in visit count relative to baseline is defined as the IRR for that characteristic. A value greater than 1 indicates an increase in visitation and a value less than 1 indicates a reduction in visitation.

When estimating both binary models of visitation and count data models, $\mathbf{x}_{it}'\boldsymbol{\beta}$ is specified as:

$$\mathbf{x}_{it}'\boldsymbol{\beta} = \beta_0 + \beta_1 x_{it}^{(1)} + \beta_2 x_{it}^{(2)} + \cdots + \beta_m x_{it}^{(m)} + \beta_{m+1} x_i^{(m+1)} + \cdots + \beta_K x_i^{(K)} + \boldsymbol{\mu}_s + \boldsymbol{\tau}_t \quad [5]$$

where $x_{it}^{(1)}$ denotes deforestation in municipality i in year t, $x_{it}^{(2)}$, ..., $x_{it}^{(m)}$ denote other time-varying covariates; $x_i^{(m+1)}$, ..., $x_i^{(K)}$ are time-invariant. β_0 is a constant, while β_1, ..., β_K are coefficients on the independent variables. We also include state fixed effects, $\boldsymbol{\mu}_s$, and year fixed effects[17], $\boldsymbol{\tau}_t$ (s = 1, …, 32 and t = 2008, …, 2016). $\boldsymbol{\beta} = \{\beta_0, \beta_1, \dots, \beta_K, \boldsymbol{\mu}_s, \boldsymbol{\tau}_t\}$ are the estimated parameters of interest.[18]

4. Results

We divide our results discussion into several sections. We report results from binary visitation probability models followed by count data models. We conclude by reporting estimates of deforestation impacts under several hypothetical scenarios.

Probability of Visitation

Column (1) in Table 2 reports the estimated coefficients from the logit visitation model. The negative coefficient on deforestation indicates that a municipality with increasing forest loss is significantly less likely to be visited by birders. In terms of the magnitude, the marginal effects in Column (2) suggest that, on average, birdwatching visitors are 10.6% less likely to visit a municipality with an additional percentage point of deforestation. The covariates on municipality size, forest area, biomass density, the number of bird species as well as threatened species and coastal area all increase the probability of a municipality being visited. The results also indicate that birders value the level of economic development, transportation infrastructure and presence of other tourist attractions when making location choices. We see this by observing that birders are more likely to visit municipalities that have higher GDP, are closer to airports, and contain more railways and highways, archaeological sites, and national protected areas.

We also estimate a binary probit model and report results in Appendix A Table A1. We adopt the Akaike information criterion (AIC) to compare the quality of different models, where the model with a smaller AIC value better fits the data. The AIC value of the logit model, 12,804.44 (in Table 2), is smaller than that of the probit model, 12,847.67 (in Appendix A Table A2), indicating that the logit model is preferred. Comparing the two models, the marginal effects from the probit model are qualitatively similar to those from the logit model.

Count Models of Visits

We report estimates for the first and second stages of the ZINB model in Table 3, Columns (1) and (2). In comparison to the binary logit model discussed previously (Table 2), the estimated signs in the first stage of the ZINB model are expected to be the opposite because the dependent variables (probabilities) are defined differently. Focusing on the deforestation variable, we find results that are consistent with our prior findings. The positive coefficient on deforestation in Column (1) indicates that deforestation leads to a higher probability of a municipality experiencing zero visits. The impacts of other municipality characteristics on birding tourism are also consistent with expectations.

Turning to the intensive margin count of visits, the estimated coefficient on deforestation in the second stage of the ZINB model is significant and negative, which suggests that visit counts in a municipality decrease when deforestation increases (Table 3, Column 2). In addition, covariates for larger municipalities, biomass density, and number of threatened bird species are found to lead to increasing numbers of visit days. Municipalities with higher July temperatures receive fewer visits.

As for additional control variables, we see that higher GDP and the presence of freight railways are associated with an increasing number of visits. In terms of location, we find that birdwatching tourists prefer to visit locations farther from large urban centers and closer to airports, which confirms the hypothesis that birdwatching activity is likely higher in more rural locations. The lack of significance in 2^{nd} stage estimates for other transportation covariates is intuitive if access primarily influences the extensive margin participation decision rather than the intensive margin of visit days.[19]

To aid in interpretation of the two-stage ZINB model, we report the IRR[20] in Columns (3) and (4) in Table 3. Examining the IRR estimates, we see that the number of visits in a municipality decreases by 30.3% given an additional unit of deforestation, where the unit is 1% in our study. The interpretation for other coefficients is similar, where an IRR smaller than 1 indicates a negative percentage impact on visitation days.

In Table 4, we conduct robustness checks by excluding all birding activities reported by Mexican residents as described in Section II and recalculating the count of day trips in each municipality using the remaining sample identified as international tourists. The resulting estimate of deforestation is of similar magnitude and significance to that in our primary specification in Table 3. An additional unit of deforestation is associated with a decrease in the number of birding visits by 33.3%. Other results are also qualitatively similar to Table 3, though notably, international tourists are more likely to be attracted to areas with chain-branded hotels and protected areas than the larger sample including local residents.[21]

As certain areas of Mexico are less likely to be visited by tourists due to security concerns, we estimate a robustness model removing these destinations using our original sample of eBird participants. The U.S. State Department issues warnings against travel in Mexico. The advisories are ranked on a scale of 1 to 4, where Level 4 advises “Do not travel”. There are 5 Mexican states classified as Level 4 due to widespread violent crime. Travel for U.S. government employees is limited in these areas and the warnings are publicly announced. To examine sensitivity to excluding these areas, we exclude states with travel warnings from our data.[22] The results are presented in Appendix A Table A5, where the negative impacts of deforestation on birdwatching remain significant and the magnitudes are similar to the previous results using the nationwide data in Table 3.

Deforestation Predictions

To place our results in context with recent deforestation in Mexico, we examine a number of scenarios for deforestation occurring in a typical Mexican municipality and predict the consequent changes in the number of visits based on our nonlinear ZINB model. Table 5 presents the results for these simulations. When a municipality experiences deforestation of 0.13%, which is the mean deforestation during our sample period in Mexico, the number of total visits is expected to decrease by 4.6% per year in comparison with a baseline scenario of no deforestation. For deforestation of 0.36%, which represents the 90th percentile of deforestation in our sample, the number of visits is expected to decrease by 12.2%. In an extreme scenario where deforestation reaches the maximum in our sample of 6.52%, the number of visits would decrease by 90.5%. For the non-resident tourist robustness model, the predicted percentage changes in visit numbers are similar.

5. Discussion

Using publicly available citizen science data, we examine the impact of deforestation on birdwatching across Mexico. We show that deforestation has a negative impact on the probability that birders choose a municipality to visit, as well as the total number of days spent by birders in a location. Our results indicate that when deforestation increases by an additional percentage point, the probability of choosing a municipality as a destination for birding and the number of visits decrease by 10.6% and 30.3%, respectively. Considering that the average size of a Mexican municipality is 1,290 km^2, a loss of 1 km^2 of forests implies deforestation of 0.075% in an average municipality and would lead to a decrease in visit counts by 2.8%. Given potential reductions in visit counts, the impacts of deforestation on local tourism and sustainable development are far reaching.

The decrease in visitation days due to deforestation is likely to result in significant revenue losses to local communities, firms, and governments. To illustrate the potential economic impact for a municipality, we take Xochimilco, a municipality in the Federal District of Mexico City as an example. Xochimilco contains 26.57 km^2 of natural protected area, which provides an ideal habitat for migratory birds. Revollo-Fernández (2015) provides an estimate of the annual economic value of birdwatching in this location at 8.75 million US dollars. Since the size of Xochimilco is 122 km^2, a loss of 1 km^2 of forests in Xochimilco implies deforestation of 0.82%. Using our estimates, this forest loss would lead to 25.6% fewer birdwatching visits. If we further assume that each birder's daily expenditure is the same and that the economic impact is proportional to visit counts, this will result in an annual loss of 2.24 million US dollars for this municipality.

In addition to direct losses due to reductions in tourism revenue, in the current information age, social media is widely used as an advertising platform. As fewer tourists are present to share their birding experiences on platforms such as Instagram and Flickr (Hausmann et al. 2018), this form of word-of-mouth type advertising is likely to further imperil tourism in these areas over time. The reduction in tourism income affects not only the local economy and residents' well-being, but also the fiscal budget for forest conservation (Sukserm et al. 2012; Satyanarayana et al. 2012). With reduced funding to prevent deforestation, the loss of forests is likely to continue.

Our analysis also demonstrates the potential to use citizen science data in data-limited settings common in many developing countries. Given the large costs of acquiring data in many developing countries, this is a promising new venue to address issues related to conservation and tourism and develop more rigorous analysis to aid policymakers seeking to balance sustainable development and tourism. Using nationwide data for citizen science data on birdwatching data in Mexico, we find clear linkages between patterns of deforestation and changes in tourism behavior.

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Table 1. Descriptive Statistics

Variables (N = 16,290)	*Mean*	*SD*	*Min*	*Max*
Annual count of day trips (#)	3.84	25.43	0	848
Deforestation (%)	0.13	0.32	0	6.52
Municipality size (1,000 km^2)	1.29	3.16	0.011	69.71
Forest area in 2000 (1,000 km^2)	0.38	1.17	0	24.43
Edge to core ratio	3.64	9.24	0	167.5
Biomass density (Mg/ha)	46.62	41.71	0.039	185.49
Bird species (#)	260.76	55.9	156	408
Threatened bird species (#)	2.48	1.47	0	7.31
Temperature in January (ºF)	61.55	9.56	37	91
Temperature in July (ºF)	72.81	10.33	53	109
Precipitation (1,000 mm)	1.06	0.62	0.054	5.27
Coast (0/1)	0.09	0.28	0	1
GDP (billion US dollars)	0.55	1.87	0.0013	22.9
Population (100 thousand)	0.55	1.39	0.0024	18.21
Length of railway (100 km)	0.10	0.24	0	2.98
Distance to the nearest urban center (100 km)	1.56	1.39	0.015	7.52
Distance to the nearest airport (100 km)	0.47	0.27	0.0087	1.99
Length of highway (100 km)	0.95	1.28	0	14.45
Number of chain-branded hotels (#)	0.19	1.16	0	17
Archaeological site (0/1)	0.05	0.22	0	1
Protected area (0/1)	0.14	0.34	0	1
Forest ejido area ratio (%)	4.58	9.79	0	83.76

Table 2. Results from Logit Model

	Estimate	*Mean Marginal Effect*
	(1)	*(2)*
Variables	*Probability of Visitation*	*Probability of Visitation*
Deforestation	-0.867***	-0.106***
	(0.161)	(0.0202)
Municipality size	0.0783***	0.00959***
	(0.0256)	(0.00313)
Forest area in 2000	0.206*	0.0252*
	(0.110)	(0.0133)
Edge to core ratio	0.00176	0.000216
	(0.00420)	(0.000515)
Biomass density	0.00435**	0.000533**
	(0.00216)	(0.000263)
Bird species	0.00300***	0.000368***
	(0.00116)	(0.000141)
Threatened bird species	0.163***	0.0199***
	(0.0614)	(0.00752)
Temperature in January	0.00305	0.000374
	(0.0194)	(0.00237)
Temperature in July	-0.0267	-0.00327
	(0.0196)	(0.00236)
Precipitation	-0.0487	-0.00596
	(0.172)	(0.0211)
Coast	1.312***	0.161***
	(0.236)	(0.0274)
GDP	0.339**	0.0416**
	(0.155)	(0.0189)
Population	-0.186	-0.0228
	(0.201)	(0.0246)
Length of railway	0.608***	0.0745***
	(0.210)	(0.0266)
Distance to the nearest urban center	-0.0271	-0.00332
	(0.125)	(0.0153)
Distance to the nearest airport	-1.201***	-0.147***
	(0.296)	(0.0349)
Length of highway	0.383***	0.0469***
	(0.101)	(0.0116)
Number of chain-branded hotels	0.0256	0.00314
	(0.0792)	(0.00972)
Archaeological site	0.569**	0.0697**
	(0.271)	(0.0326)
Protected area	0.684***	0.0837***
	(0.163)	(0.0191)
Forest ejido area ratio	0.00649	0.000795
	(0.00438)	(0.000537)
Observations	16,254	16,254
Method	Logit	Logit
Year FE (9)	Yes	Yes
State FE (32)	Yes	Yes

All regressions include a constant. Robust standard errors in parentheses (clustered by states). *** p<0.01, ** p<0.05, * p<0.1.

All the municipalities in the states of Baja California and Baja California Sur received non-zero visits every year during the sample period. 36 observations are dropped given no within-state variation of the dependent variable.

Table 3. Results from ZINB Model

	Estimate		*Incidence Rate Ratio*	
	(1)	*(2)*	*(3)*	*(4)*
Variables	*Constant zero*	*Visit count*	*Constant zero*	*Visit count*
Deforestation	0.878**	-0.361*	2.406**	0.697*
	(0.364)	(0.204)	(0.875)	(0.142)
Municipality size	-0.199***	0.0337	0.819***	1.034
	(0.0721)	(0.0241)	(0.0591)	(0.0249)
Forest area in 2000	-0.692**	0.120**	0.501**	1.128**
	(0.321)	(0.0478)	(0.161)	(0.0539)
Edge to core ratio	0.00735	0.0249	1.007	1.025
	(0.0113)	(0.0282)	(0.0113)	(0.0289)
Biomass density	-0.00262	0.00556*	0.997	1.006*
	(0.00293)	(0.00310)	(0.00293)	(0.00312)
Bird species	-0.00706**	-0.00601**	0.993**	0.994**
	(0.00334)	(0.00275)	(0.00332)	(0.00274)
Threatened bird species	-0.109	0.275***	0.897	1.316***
	(0.150)	(0.0794)	(0.134)	(0.105)
Temperature in January	-0.0180	0.0128	0.982	1.013
	(0.0359)	(0.0220)	(0.0352)	(0.0223)
Temperature in July	0.0253	-0.0493**	1.026	0.952**
	(0.0351)	(0.0240)	(0.0360)	(0.0229)
Precipitation	-0.130	-0.0937	0.878	0.911
	(0.213)	(0.179)	(0.187)	(0.163)
Coast	-2.046***	0.396	0.129***	1.486
	(0.351)	(0.254)	(0.0453)	(0.378)
GDP	-1.382	0.185**	0.251	1.203**
	(4.242)	(0.0922)	(1.065)	(0.111)
Population	0.481	-0.116	1.618	0.890
	(3.834)	(0.127)	(6.205)	(0.113)
Length of railway	-0.353	0.375*	0.702	1.455*
	(0.458)	(0.214)	(0.321)	(0.312)
Distance to the nearest urban center	0.695***	0.489***	2.004***	1.631***
	(0.191)	(0.142)	(0.383)	(0.231)
Distance to the nearest airport	1.119***	-1.100***	3.063***	0.333***
	(0.430)	(0.293)	(1.316)	(0.0976)
Length of highway	-1.331***	0.0707	0.264***	1.073
	(0.302)	(0.0520)	(0.0798)	(0.0558)
Number of chain-branded hotels	0.0554	0.0660	1.057	1.068
	(0.611)	(0.0520)	(0.646)	(0.0556)
Archaeological site	-0.970	0.179	0.379	1.196
	(0.690)	(0.310)	(0.262)	(0.371)
Protected area	-0.948***	0.245	0.388***	1.277
	(0.286)	(0.157)	(0.111)	(0.200)
Forest ejido area ratio	-0.0247**	-0.00926	0.976**	0.991
	(0.0111)	(0.00905)	(0.0108)	(0.00897)
ln(α)		1.130***		3.095***
		(0.0987)		(0.305)
AIC		34355.33		
Observations	16,290	16,290	16,290	16,290
Method	ZINB	ZINB	ZINB	ZINB
Year FE (9)	Yes	Yes	Yes	Yes
State FE (32)	Yes	Yes	Yes	Yes

All regressions include a constant. Robust standard errors in parentheses (clustered by states). *** p<0.01, ** p<0.05, * p<0.1.

Table 4. Results from ZINB Model, Excluding Resident Records for Visits

	Estimate		*Incidence Rate Ratio*	
	(1)	*(2)*	*(3)*	*(4)*
Variables	*Constant zero*	*Visit count*	*Constant zero*	*Visit count*
Deforestation	1.086***	-0.405**	2.961***	0.667**
	(0.333)	(0.199)	(0.986)	(0.133)
Municipality size	-0.127	0.0509	0.881	1.052
	(0.0795)	(0.0376)	(0.0700)	(0.0395)
Forest area in 2000	-0.334	0.0905*	0.716	1.095*
	(0.341)	(0.0477)	(0.244)	(0.0522)
Edge to core ratio	-0.0222	-0.0109	0.978	0.989
	(0.0247)	(0.0115)	(0.0241)	(0.0114)
Biomass density	-0.00533	0.00916**	0.995	1.009**
	(0.00384)	(0.00363)	(0.00382)	(0.00366)
Bird species	-0.00266	-0.00282	0.997	0.997
	(0.00372)	(0.00382)	(0.00371)	(0.00381)
Threatened bird species	-0.162	0.273***	0.850	1.314***
	(0.119)	(0.0882)	(0.101)	(0.116)
Temperature in January	-0.0610	-0.0343	0.941	0.966
	(0.0422)	(0.0358)	(0.0397)	(0.0346)
Temperature in July	0.0616	0.0105	1.064	1.011
	(0.0398)	(0.0373)	(0.0423)	(0.0377)
Precipitation	-0.0847	-0.359	0.919	0.699
	(0.217)	(0.260)	(0.200)	(0.182)
Coast	-1.978***	0.478	0.138***	1.613
	(0.441)	(0.337)	(0.0610)	(0.543)
GDP	-2.507	0.108	0.0815	1.115
	(1.707)	(0.103)	(0.139)	(0.115)
Population	0.992	-0.118	2.695	0.888
	(1.381)	(0.111)	(3.721)	(0.0987)
Length of railway	-0.761	0.280	0.467	1.323
	(0.786)	(0.273)	(0.367)	(0.361)
Distance to the nearest urban center	0.592**	0.241	1.807**	1.273
	(0.234)	(0.150)	(0.423)	(0.191)
Distance to the nearest airport	1.205**	-0.917***	3.336**	0.400***
	(0.561)	(0.340)	(1.870)	(0.136)
Length of highway	-1.320***	-0.0880*	0.267***	0.916*
	(0.371)	(0.0529)	(0.0990)	(0.0485)
Number of chain-branded hotels	0.688*	0.164***	1.990*	1.178***
	(0.400)	(0.0536)	(0.795)	(0.0631)
Archaeological site	-0.956	0.354	0.384	1.425
	(0.848)	(0.325)	(0.326)	(0.464)
Protected area	-0.514*	0.454***	0.598*	1.574***
	(0.293)	(0.167)	(0.175)	(0.263)
Forest ejido area ratio	-0.0184**	0.00281	0.982**	1.003
	(0.00797)	(0.0144)	(0.00782)	(0.0144)
ln(α)		1.054***		2.868***
		(0.170)		(0.487)
Observations	16,290	16,290	16,290	16,290
Method	ZINB	ZINB	ZINB	ZINB
Year FE (9)	Yes	Yes	Yes	Yes
State FE (32)	Yes	Yes	Yes	Yes

All regressions include a constant. Robust standard errors in parentheses (clustered by states).*** p<0.01, ** p<0.05, * p<0.1.

Table 5. Implications of Deforestation

Scenarios	*Deforestation*	*Predicted number of visits decreases by*	
		All visitors	*Non-residents*
Mean deforestation in Mexico	0.13%	4.6%	5.1%
Percentiles of deforestation in Mexico			
50th (Median)	0.013%	0.5%	0.5%
75th	0.11%	3.9%	4.4%
90th	0.36%	12.2%	13.6%
95th	0.62%	20.1%	22.2%
99th	1.38%	39.2%	42.8%
Max deforestation in Mexico	6.52%	90.5%	92.9%

All scenarios assume deforestation occurs in a typical Mexican municipality and predict the changes in the number of visits in the jurisdiction based our ZINB models. Statistics of deforestation percentages in Mexico are calculated from our data sample.

Figure 1. Aggregate eBird Reports of Day Trips to Mexico, 2008-2016

Figure 2. Forest cover (2000) and deforestation (2008-2016) in Mexico based on Hansen et al. (2013)

[1] We refer to "biodiversity" as the number of species and abundance as the population of a given species. With low levels of forest disturbance, it is possible that species richness will increase due to additional habitats created; however, the new species are less likely to be of lesser conservation value (Krebs 2001). At high levels of forest disturbance, biodiversity is expected to decrease.

[2] U.S. Department of the Interior, U.S. Fish and Wildlife Service, and U.S. Department of Commerce, U.S. Census Bureau. 2016 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation.

[3] Caribbean Tourism Organizations: Acorn Consulting Partnership, 2008. Bird Watching. Developing a Niche Tourism Market Database for the Caribbean: 20 Niche Market Profiles.

[4] American Birding Association, 1994. ABA Membership Survey

[5] There is an extensive literature focusing on birdwatching tourism in developing countries including South Africa (Conradie and van Zyl 2013), Peru (Puhakka et al. 2011), Colombia (Ocampo-Penuela and Winton 2017), Mexico (Cantú et al. 2011; Revollo-Fernández 2015), China (Li et al. 2013; Ma et al. 2013) and Thailand (Hvenegaard and Dearden 1998).

[6] World Tourism Organization, 2002. The U.S. Ecotourism Market

[7] Municipalities are the second-level administrative divisions below states.

[8] In 2012, eBird launched an app called "Bird Log" that allowed users to directly report birding trips, which was likely to further attract new users. In the following empirical analysis, we adopt dummy variables to control for year fixed effects.

[9] 28.8% of the day trips in our sample are associated with multiple checklists submitted by a birder within a day.

[10] The deviation of each ratio by state is consistently smaller than twice of the standard deviation of the sample.

[11] We use version 1.5 updated after the original journal article was published. The version includes maps of forest loss from 2000 through 2017 and is available as an online repository.

[12] We do not consider reforestation due to the relatively low amount occurring during our study period and challenges in measuring reforestation from the Hansen data.

[13] The Mexican railway system has suspended passenger service since 1997 except for a limited number of tourist trains such as Chihuahua-Pacific Railway (El Chepe) traversing the Copper Canyon, and thus is not a prevalent mode of public transport.

[14] Alternatively, we also consider other population thresholds including 200,000 (41 centers) and 100,000 (67 centers) to calculate distance yielding qualitatively similar results.

[15] Appendix A Table A9 shows the correlation matrix of all our explanatory variables. None of the pairwise correlation coefficients exceed 0.8 except for one between GDP and population.

[16] As a robustness check, we also estimate a probit model and report those results in the Appendix A.

[17] To account for the baseline level of deforestation, we consider a robustness model in Appendix A Table A7. We calculate the average deforestation over the years 2001-2007 before our sample period for each municipality and subtract that from the observed deforestation. Based on this, we re-estimate the ZINB model and the results are nearly identical to our main results in Table 3. Alternatively, we also consider a first-difference model to eliminate the time trend effect in Appendix A Table A8. The results are consistent with our preferred model using year fixed effects.

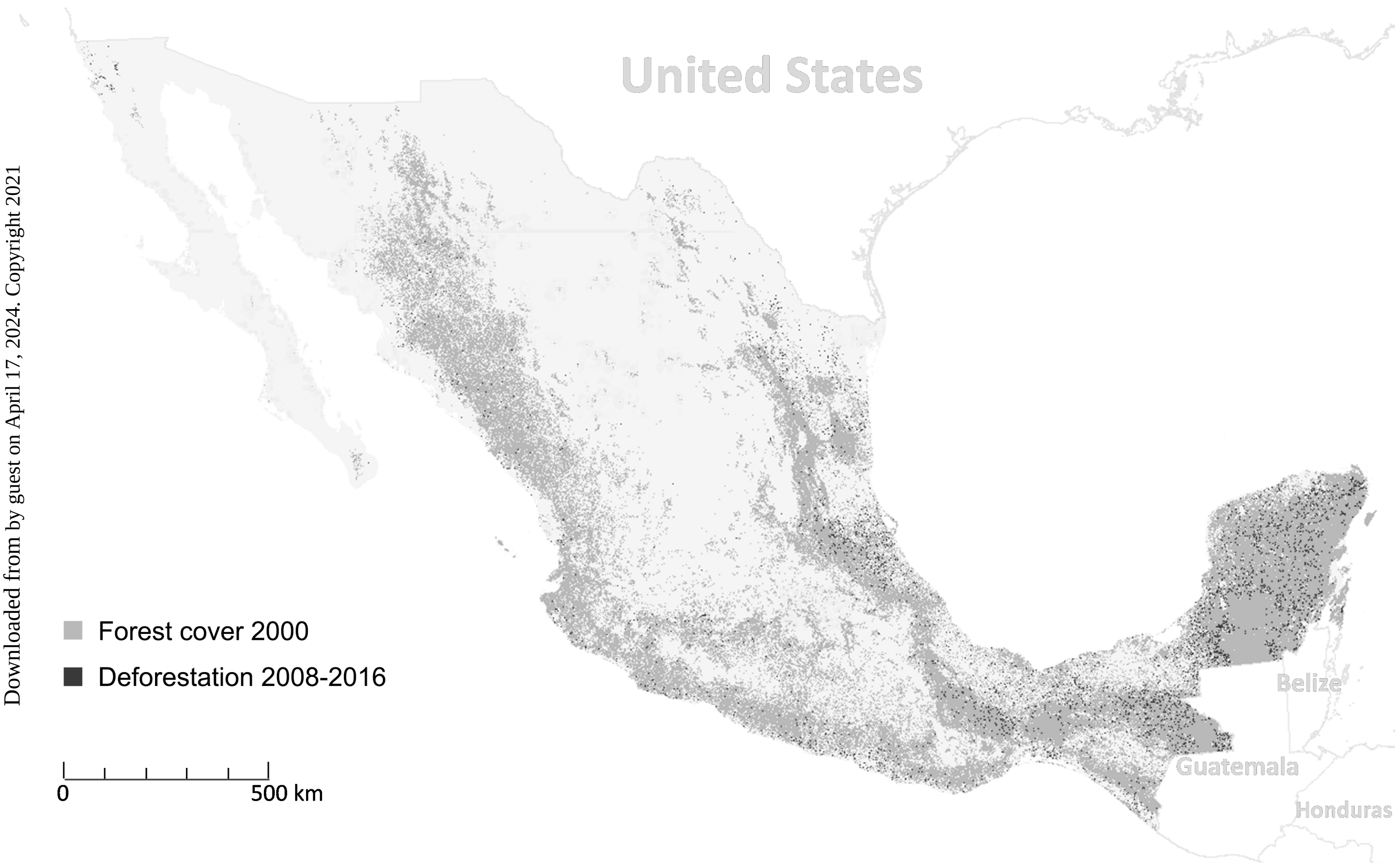
[18] Note that this linear combination of coefficients and characteristics is an exponent of e in the model. When interpreting coefficients using IRRs, the magnitudes of impacts are non-linear with respect to characteristics.

[19] We report results from Poisson and NB models in Appendix A Table A2. The AIC value of the ZINB model (in Table 3) is smaller than that of the NB model (in Appendix A Table A2), suggesting there are excess zeros in our visit counts and the ZINB model is preferred. Moreover, the significant $\ln(\alpha)$ in the ZINB model indicates overdispersion after the excess zero issue is addressed, showing that a NB process in the second stage fits our data better than a Poisson process. Even though these findings favor the ZINB model, the results in terms of deforestation across all the three count data specifications are qualitatively similar.

[20] For example, an estimated coefficient of deforestation of -0.361 is equivalent to an IRR of 0.697 ($= e^{-0.361}$).

[21] While this paper is focused on the impacts of deforestation on birdwatching, there is a potential for birdwatching affecting deforestation simultaneously, and the resulting reverse causality would bias our estimates. If some forests do not attract sufficient visitors to generate revenue, they might have a higher risk of being deforested. On the other hand, prosperous tourism accompanied with infrastructure construction is also likely to accelerate deforestation. To address these endogeneity issues, we also consider specifications using lagged deforestation by one year, which is unlikely to be affected by birders' visitations in the current year. The results are shown in Appendix A Tables A3 and A4 and are highly consistent with our primary specifications.

[22] We also estimate a model that includes indicator variables to indicate states with travel warnings and the results are shown in Appendix A Table A6.


United States
Forest cover 2000
Deforestation 2008-2016
0
500 km
Belize
Guatemala
Honduras