Original Research Article

The potential distribution and disappearing of Yunnan snub-nosed monkey: Influences of habitat fragmentation

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A R T I C L E   I N F O

A B S T R A C T

Analysis: of environmental variables and organism occurrence records offers insight that can be used to predict potential distribution areas and habitat fragmentation. For large landscapes, modeling is the most convenient and effective way to conduct habitat research. Two species distribution models, BIOMOD2 and FRAGSTATS 4.2, were given data on environmental variables and organism occurrence records as input and used to predict the potential suitable habitat and habitat fragmentation for the Yunnan snub-nosed monkey (*Rhinopithecus bieti*). Our results estimated the total area of potentially suitable habitat for *R. bieti* as 7412.82 km^2, but only 4164.58 km^2 was found to be inhabited by *R. bieti*. We found that the main land cover type in the potential suitable distribution area of *R. bieti* was evergreen needle-leaf forest (6153.95 km^2, 83.02%). Comparison of inhabited and suitable but uninhabited habitats showed that areas actually inhabited by *R. bieti* had a lower patch density (PD) and higher largest patch index (LPI) than uninhabited habitats only in evergreen needle-leaf forest. The potential suitable habitats of *R. bieti* has increased significantly, but the actual distribution has shrunk from 1997 to 2017. Although the government has made great progress in protecting *R. bieti*, logging that took place before the regulations and the boundary effect of roads and rivers resulted in the local extinction of *R. bieti* in some potentially suitable areas. In view of this, we propose to establish a national park for Yunnan snub-nosed monkeys. We also suggest protecting the potentially suitable but currently empty habitats for later release of *R. bieti*. Successfully reintroducing *R. bieti* into areas where it formerly lived will require continual and careful habitat monitoring.

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1. Introduction

Species distribution models (SDMs), a tool used to predict the potential distribution of species (Guisan and Thuiller, 2005) has steadily become more popular over the last three decades (Antoine et al., 2013). The increase in sharing of organismal presence data by many organizations (Graham et al., 2004; Wisz et al., 2008; Syfert et al., 2013), wide availability of GIS data layers capturing environmental variables (Franklin, 2009), and a proliferation of user-friendly software packages (Syfert et al., 2013) have contributed to the broader adoption of SDMs (Guisan and Thuiller, 2005; Thuiller et al., 2008; Elith and Leathwick, 2009). SDMs can be used to identify potential distributions for species or communities (Andreas et al., 2010; Clements et al., 2012), estimate conservation performance (Carvalho et al., 2010; Doko et al., 2011; Tulloch et al., 2016; Bosso et al., 2017a; Kabir et al., 2017), forecast spatial patterns of biological invasions (Thuiller et al., 2010; Bosso et al., 2017a), inform species reintroduction (Olsson and Rogers, 2009; Ardestani et al., 2015), and simulate past, present, and future distributions under climate change and human disturbance (Rebelo et al., 2010; Beaumont et al., 2010; Hu and Jiang, 2011; Jian et al., 2016; Bosso et al., 2017b; Rubio-Salcedo et al., 2017; Zhao et al., 2018, 2019a, 2019b).

Habitat fragmentation is a serious threat to worldwide biodiversity (Delgado and Stefanic, 1998; Pereira et al., 2010; Rands et al., 2010), reducing biodiversity between 13% and 75% (Haddad et al., 2015). Habitats become fragmented as human-transformed areas divide suitable habitat into smaller, more isolated patches (Haddad et al., 2015). Fragmentation creates detrimental edge effects along the boundaries of habitat patches, leading to the reduction of indigenous species as ecological conditions change (Gibbs and Stanton, 2001), limited animal movement and gene flow, and diminished connectivity between patches (With and Crist, 1995; Pither and Taylor, 1998; Bélisle et al., 2001). More than half of the world's nonhuman primate species are currently threatened with extinction (Chapman and Peres, 2001), and habitat fragmentation caused by intensifying land use (Arroyo-Rodríguez and Fahrig, 2015). Nonhuman primate species depend on intact, biodiverse forests for a nutritionally balanced diet, making populations that inhabit small and isolated forest fragments especially vulnerable to extinction. (Benchimol and Peres, 2013; Arroyo-Rodríguez and Fahrig, 2015).

The endangered Yunnan snub-nosed monkey (*Rhinopithecus bieti*) lives at the highest altitudes of any non-human primate (Long et al., 1994). *R. bieti*'s geographic distribution is restricted to the Hengduan Mountains of Northwestern Yunnan and Southeastern Tibet, and the species is primarily associated with high-elevation evergreen needle-leaf forests (Kirkpatrick and Long, 1994; Xiao et al., 2003; Li et al., 2008; Grueter et al., 2008). For the past twenty years, the Chinese government has protected the snub-nosed monkeys occurring within China, promoting the recovery of their populations (Zhao et al., 2019b). Despite the increase in the total population of *R. bieti*, their distribution has shrunk (Zhao et al., 2019b). Habitat fragmentation has constrained the monkeys' to respond to seasonal climate and vegetation changes through altitudinal migration and long-distance migration (Long et al., 1994; Xiao et al., 2003; Li et al., 2008; Ren et al., 2010). Since field studies of the ecology of *R. bieti* have always been limited by difficult environmental conditions (Long et al., 1994), most of the population counts and distribution maps produced have been rough estimates.

SDMs rely on environmental variables and records of organism occurrence to predict potential distribution areas across large landscapes (Elith and Leathwick, 2009). In this study, we constructed SDMs based on previous survey data that defined inhabited and uninhabited areas for *R. bieti* to infer the potential distribution of suitable habitat. We further assessed how present habitat fragmentation influences this species' distribution by comparing the potentially suitable area to the currently inhabited area.

2. Materials and methods

2.1. Study area

In the southern part of the Hengduan Mountains, deep valleys between mountain ranges are formed by the upper Yangtze−Jinshajiang and upper Mekong−Lancangjiang river complexes (Wong et al., 2013). The study area encompassed all of *R. bieti's* range in the southwest of China, a total area of approximately 61,810 km^2 spanning northwestern Yunnan and Mangkang County in Tibet. This region occurs between latitudes 25°−30°N and longitudes 101°−98°E and ranges in elevation between 1108 and 5238 m. The 2013 survey of this area found 17 groups of monkeys and estimated a total population of about 3,000 individuals (Fig. 1). The known geographic distribution of *R. bieti* is smaller than our study area, stretching from Longmashan in the south to Xiaochangdu in the north over an area of less than 20,000 km^2 (Long et al., 1994).

2.2. Presence records

We collected location records of monkeys occurred from observations made during nature reserve patrols, infrared camera surveys, and some historical data from the literature (Ren et al., 2010). From 2005 to 2013, we surveyed the northern areas (Zhina, Xiaochangdu, Milaka, and Bamei) and the central areas (Wuyapuya) several times. Excluding areas outside the boundary and repeated points, we collected a total of 1712 occurrence points (see Table 1). Considering that *Macaca mulatta* and *R. bieti* have overlapping ecological niches in this area, we excluded 460 points defined from trace materials. According to Ren et al.'s (2009) GPS analysis of the Laojunshan *R. bieti* group, with a range of 45.66 ha (0.01−45.67 ha), the maximum active diameter was estimated to be 0.78 km, and we set the buffer distance to 1 km 137 records were screened in ArcGIS (version 10.2.2) for spatial autocorrelation using average nearest neighbor analyses to remove spatially correlated data points and
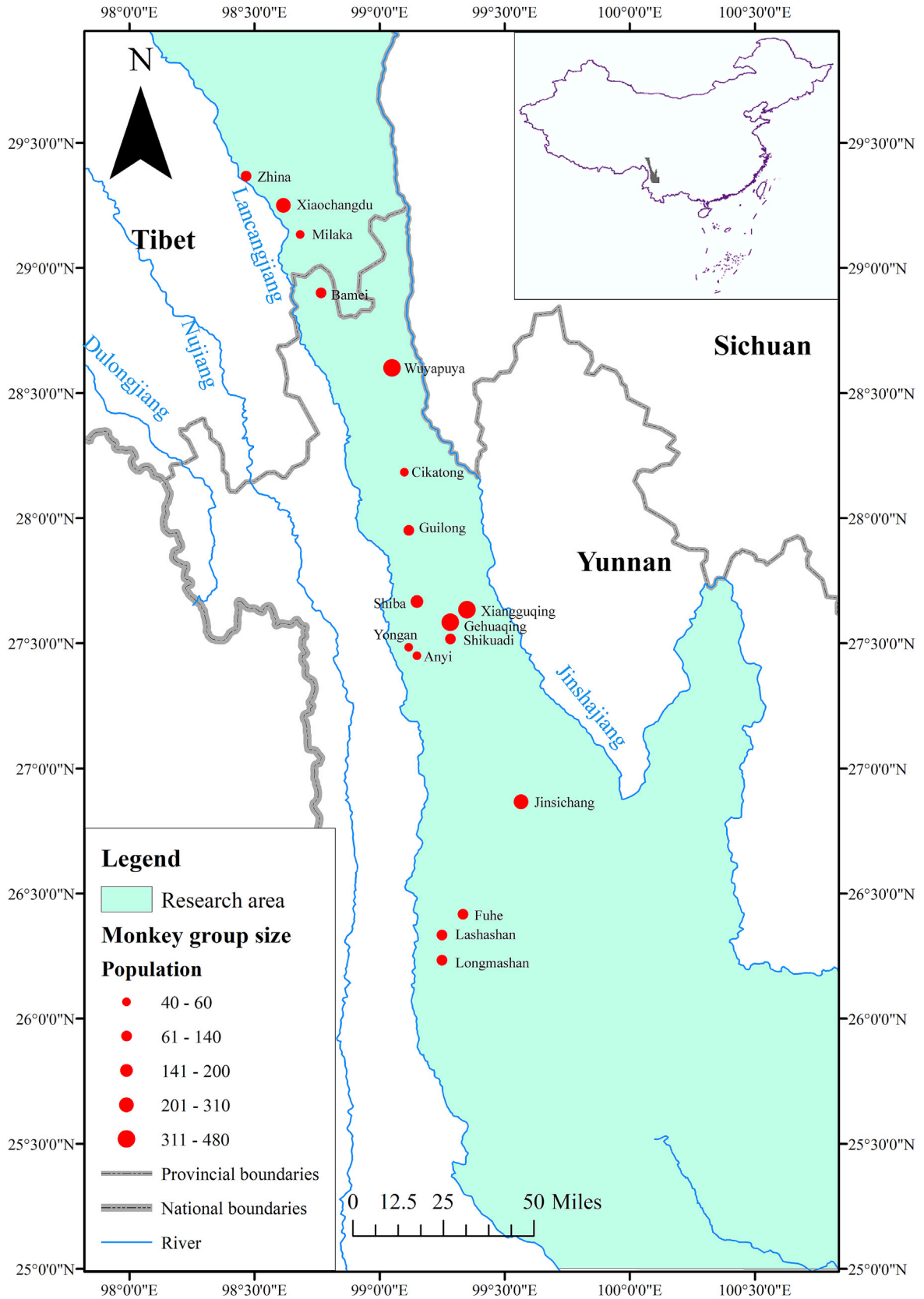
Fig. 1. The research area and locations of *R. bieti* groups.

Table 1
The number of occurrence points in each geographic area.

Geographical group	All records	Fecal and meeting monkey points	1 km buffer points
Northern section	147	38	18
Wuyapuya	41	35	14
Central part	1350	1043	69
Jinsichang	105	79	25
Southern part	69	55	11

Table 2
The true skill statistic (TSS) calculated for different models.

	RF	GBM	GLM	MAXENT. Phillips
TSS	0.76 ± 0.05	0.74 ± 0.04	0.75 ± 0.05	0.70 ± 0.06

guarantee independence (Dormann et al., 2013; Bosso et al., 2017a). The number of points for each geographical area is shown in Table 1.

2.3. Environmental variables and species distribution modelling

As an initial set of environmental predictors for training the SDM, we considered terrain variables (including elevation, slope, and aspect) derived from digital elevation model (DEM) data (resolution of 30 m for this study area http://www.gscloud.cn/), and an additional 19 bioclimatic variables derived from the WorldClim dataset (www.world clim.org/current).

We built SDMs using an ensemble forecasting approach, as implemented in the R package BIOMOD2 (https://cran.r-project.org/bin/windows/base/). These included ten modeling techniques: generalized linear modeling (GLM), generalized additive modeling (GAM), generalized boosting modeling (GBM), classification tree analysis (CTA), artificial neural network (ANN), flexible discriminant analysis (FDA), surface range envelop (equivalent to BIOCLIM), multivariate adaptive regression splines (MARS), random forest (RF), and maximum entropy (MAXENT.Phillips). The variance inflation factor (VIF) was utilized to reduce collinearities (Dormann et al., 2013), and 7 variables (Table 3) were selected for modeling. Before ensemble modeling, we tested all ten modeling techniques to determine the optimal technique according to the true skill statistic (TSS) and the area under the curve (AUC) (Allouche et al., 2006). The best four modeling techniques for the species distribution models (RF, GLM, GBM, MAXENT.Phillips) were applied to develop the models in BIOMOD2. We randomly selected 70% of the occurrence records as calibrations the models and used the other 30% to assess model performance. The TSS and AUC of the receiver operating characteristic (ROC) curve are ecological measures for assessing the accuracy of ordinal score models (Allouche et al., 2006; Thuiller, 2003).

2.4. Landscape data and spatial analysis

We derived figures in 2017 for 13 land cover variables from the Database of Global Change Parameters, Chinese Academy of Sciences (http://globalchange.nsdc.cn). Landscape metrics for *R. bieti* were calculated using FRAGSTATS 4.2 (McGarigal and Marks, 1994). Five indices were used to assess potential habitat quality and fragmentation: Patch density (PD), with low PD indicating a more connected landscape and high PD indicating a more fragmented landscape (Olsoy et al., 2016); Percentage of landscape (PLAND), a measure of landscape composition, where low PLAND indicates that the patch type is rare and a value equal to 100 indicating the whole landscape consists of only one type of patch; Aggregation index (AI), with high AI indicating a greater concentration of a certain type of patch and low AI indicating greater dispersal; Largest patch index (LPI), which approaches 0 when the largest patch of the corresponding patch type becomes extremely small and equals 100 when the largest patch comprises 100% of the landscape; and CONTAG, used to describe the degree of agglomeration or extension trend of patch types in the landscape, with larger CONTAG scores indicating that dominant patch types in the landscape are well-connected. The following formulas were used to calculate these indices:

$$PD = \frac{n_i}{A}(10,000)(100)$$

Table 3
Relative importance of variables for *R. bieti* distribution in research area.

Variable	ALT	B_3	B_{15}	B_{13}	B_2	SLO	ASP
Relative importance	0.4	0.159	0.144	0.106	0.037	0.005	0.003

ALT, Altitude; B_3, Isothermality; B_2 Mean Diurnal Range; B_{13} Precipitation of Wettest Month; B_{15} Precipitation Seasonality (Coefficient of Variation); SLO, Slope; ASP, Aspect.

$$PLAND = P_i = \frac{\sum_{j=1}^{n} a_{ij}}{A} (100)$$

$$LPI = \frac{\max_{j=1}^{n} (a_{ij})}{A} (100)$$

$$AI = \left[\frac{g_{ik}}{\max \rightarrow g_{ik}} \right] (100)$$

$$CONTAG = \left[1 + \frac{\sum_{i=1}^{m} \sum_{k=1}^{m} \left[(p_i) \left(\frac{g_{ik}}{\sum_{k=1}^{m} g_{ik}} \right) \right] \left[\ln(p_i) \left[\frac{g_{ik}}{\sum_{k=1}^{m} g_{ik}} \right] \right]}{2 \ln(m)} \right] (100)$$

A represents total landscape area, $n = n_i$ the number of patches in the landscape of patch type (class) i, a_{ij} Area (m^2) of patch ij, g_{ik} the number of adjacencies (joins) between pixels of patch types (classes), and m the number of patch types (classes) present in the landscape, excluding the landscape border if present.

3. Result

3.1. Species model performance

Based on the empirical data from field surveys, 137 occurrence records for *R. bieti* (Table 1), the models showed high levels of predictive performance (TSS = 0.874, AUC = 0.984). On average, the RF technique resulted in the highest TSS value (TSS = 0.76 ± 0.05), while the MAXENT.Phillips resulted in the lowest value (TSS = 0.70 ± 0.06) (Table 2). Altitude (ALT) was by far the most important variable (Table 3).

3.2. Potential suitable habitat and land cover types for R. bieti

The model predictions showed that *R. bieti* inhabits a narrow area (Predicted area = 8701.62 km^2) between the Mekong River and Yangtze River (Fig. 2). Excluding unsuitable habitats (cropland, grassland, shrubland, wetland, water, tundra, impervious surfaces and bare areas) in the predicted area, the potential suitable habitat of *R. bieti* was 7412.82 km^2 (Fig. 2). Forest fragmentation spatially impeded *R. bieti* population communication and exchange. According to the location of major roads, rivers and patches, we divided the potential suitable habitat of *R. bieti* into fifteen isolated patches. Survey results showed that only six of those fifteen patches were inhabited by *R. bieti*, and the actual habitat area was only 4164.58 km^2. Four (1282.3 km^2, 17.3%) of the remaining nine isolated patches had populations of *R. bieti* in 1994 (Long et al., 1994), but those groups may have disappeared (see Fig. 2).

We found that the main land cover types in the potential suitable habitat of *R. bieti* were evergreen needle-leaf forests (6153.95 km^2, 83.02%), followed by mixed needle-deciduous forest (1004.86 km^2, 13.56%), then evergreen broad-leaf forest (197.1 km^2, 2.66%). There was very little deciduous broad-leaf (56.79 km^2, 0.77%) or deciduous needle-leaf forest (0.12 km^2, 0.0016%) within the potential suitable habitats.

3.3. Comparison of inhabited and uninhabited habitats within the potentially suitable area

At a landscape level, patch connectivity and aggregation did not directly drive local population extinction for *R. bieti*. CONTAG was not significantly different between inhabited and uninhabited patches within the potentially suitable habitat (Mann-Whitney U, U = 23, P = 0.637). Likewise, the aggregation index (AI) was not significantly different (U = 14, P = 0.126). On the other hand, patch density (PD) of inhabited areas was significantly lower than uninhabited areas (U = 10, P = 0.045) while inhabited areas had a higher largest patch index (LPI) than uninhabited areas (U = 4, P = 0.007).

We broke the landscape indices down further by vegetation type to compare habitats with and without *R. bieti*. Only the LPI and PD in evergreen needle-leaf forest had significant differences. Habitats where *R. bieti* occurs had a lower PD (Mean ± SD = 0.17 ± 0.03, Independent sample T test, P < 0.01) and larger LPI (Mean ± SD = 77.2 ± 16.8, P < 0.01) than uninhabited areas (LPI, Mean ± SD = 36.49 ± 17.82; PD, Mean ± SD = 0.5 ± 0.34) in evergreen needle-leaf forest (Fig. 3).
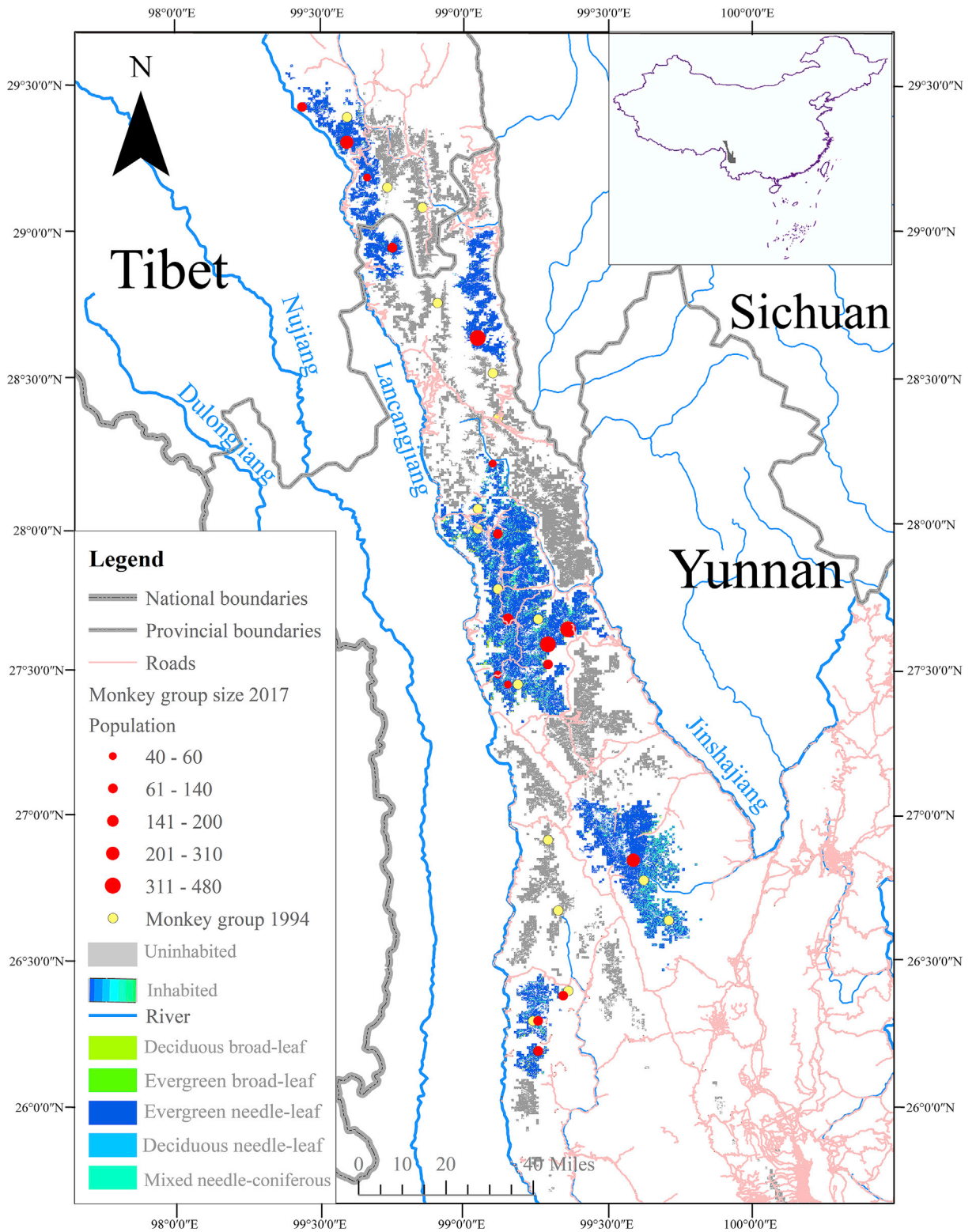
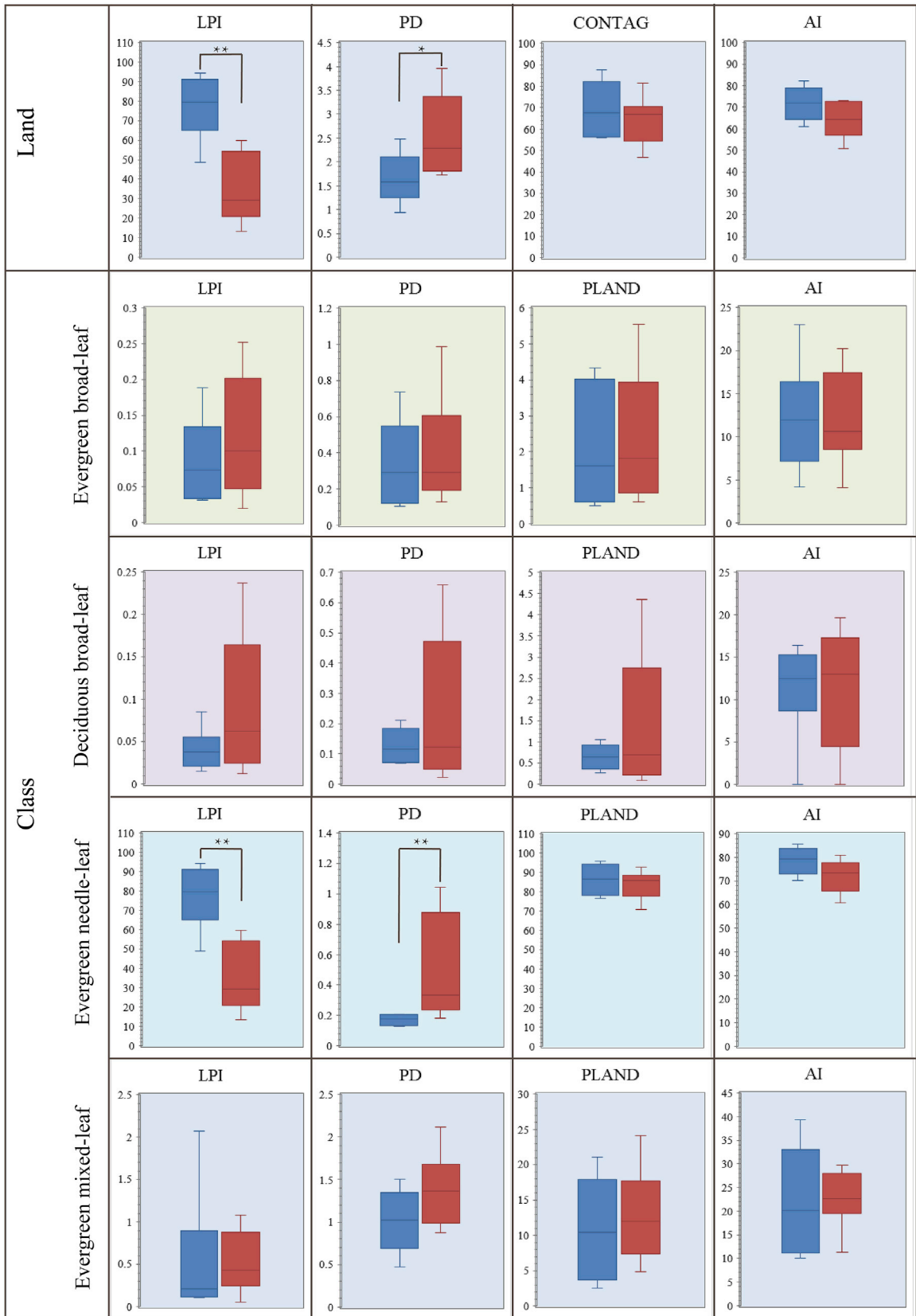**Fig. 2.** The potential suitable distribution area of *R. bieti*.

Fig. 3. Comparison of area between inhabited and suitable but uninhabited habitats.

4. Discussion

4.1. SDMs provide insight to predict R. bieti potential distribution

Tracking groups of snub-nosed monkeys on foot in the field has proven difficult (Jablonski, 1998), due to high altitudes, dense forests, complex terrain, deep gorges, sheer peaks and the monkey's avoidance of human observers. Therefore, most previous studies of *R. bieti* focused on the movements of a single group over a short period of time, obtaining rough home range sizes for that group only (Bai and Ding, 1988; Kirkpatrick, 1998; Grueter et al.,2008, 2009). Most estimates have been based on sightings by local residents over several years (Kirkpatrick, 1996). The negative effects of human disturbance, habitat fragmentation, climate change, and other factors have caused changes in the geographic distribution of the Yunnan snub-nosed monkey (Zhao et al., 2019b). *R. bieti* are rare, elusive, and difficult to observe, so adequate conservation planning may require identifying potential habitat rather than relying on observed occurrence alone (Amorim et al., 2014; Cooper-Bohannon et al., 2016; Herkt et al., 2017). SDMs rely on environmental variables and occurrence records to calculate the potential distribution of a species (Elith and Leathwick, 2009).

4.2. The number of R. bieti has increased, but the range has declined sharply in the past 20 years

Xiao et al. (2003) considered dark-coniferous forest, mixed coniferous and broadleaf forest, and oak forest patches as suitable habitats for *R. bieti* and estimated their total area in Yunnan as 4169 km^2 in 1997. The area of potential suitable habitat of *R. bieti* has expanded in the past twenty years, to nearly 7412.82 km^2 based on this study. 6442 km^2 of the potential suitable habitat is within Yunnan province, an increase of 2273 km^2 (54.5%) from the 1997 estimate. Yunnan stopped deforestation of natural forests in 1998, and the Natural Forest Protection Project implemented in 2002 allowed forests in the region to recover faster. As a consequence, the population of *R. bieti* has increased from more than 2,000 in 1994 to more than 3,000 in 2017 (Long et al., 1994; Zhao et al., 2019b). However, their actual range has declined sharply, with a reduction in the total distribution area of 1282.3 km^2. As the region's economy is relatively backward, with few local industries, local governments relied on the clear-cutting of timber to collect revenue before the prohibition of logging within the natural forest. Thus, within the study area, only parts of the virgin forests in Tibet's Mangkang County, Yunnan's Deqin County, and the central part of Yunnan's Weixi County have survived, forming the most important habitat for *R. bieti*. Though the monkeys were well-adapted to rarely-disturbance habitats in 1994(Long et al., 1994), when they are frequently disturbed by humans and their habitats are destroyed they will abandon their original habitats and migrate to new areas. However, roads and rivers prevented the migration of some of these disturbed groups, and groups of *R. bieti* living in other regions were similarly blocked from migrating back to recovered areas that had become suitable for habitation again, eventually leading to localized extinction.

Consistent with previous studies, we found that the main land cover types in the potential suitable distribution area of *R. bieti* were evergreen needle-leaf forest, followed by evergreen mixed leaf and evergreen broadleaf forest (Kirkpatrick and Long, 1994; Yang, 2003; Li et al., 2008, 2010). *R. bieti* feeds on young leaves, flowers, and fruits within the broadleaf and evergreen mixed forests during spring, summer and autumn (Grueter et al., 2008; Li et al., 2010) In the winter, the monkeys depend on the omnipresent beard lichens as a staple, which are more abundant in evergreen needle-leaf forests than in other vegetation types (Grueter et al., 2008).

4.3. Evergreen needle-leaf may be a vital habitat type for R. bieti

Habitat fragmentation and land-use intensification are major threats to more than half of the world's nonhuman primate species (Chapman and Peres, 2001; Arroyo-Rodríguez and Fahrig, 2015). Fortunately in the case of *R. bieti*, the government's prohibition on cutting down natural forests in 1998, implementation of the Natural Forest Protection Project in 2002, and establishment of three nature reserves (Baimaxueshan, Yunling, and Tianchi) between 1983 and 2003 allowed regional forests to recover, inhibiting the continued fragmentation of monkey habitats (Zhao et al., 2019b). Even after twenty years of vegetation restoration, there is no significant difference in habitat connectivity (CONTAG) and patch aggregation (AI) in suitable areas with and without monkeys. However, habitats where monkeys presently occur have lower patch density, and the largest patches have a bigger absolute area. The main reason may be that in most places where monkeys still live, the virgin forests were never cut, especially in the central and northern parts of their range. The relative inaccessibility of these areas protected the habitat on which *R. bieti* depends.

We analyzed each vegetation type in habitats with and without *R. bieti* and found that only the LPI and PD in evergreen needle-leaf forest had significant differences. Evergreen needle-leaf forests may be vital habitat for *R. bieti*. Previous studies have shown that Yunnan snub-nosed monkeys preferred evergreen needle-leaf forest for their sleep sites (Li et al., 2010). Moreover, the lichens that serve as dietary staples for *R. bieti* are abundant in evergreen needle-leaf forest. *R. bieti* routinely travels within its home range and does not use any given sleep site continuously (Xiang et al., 2013). This makes evergreen needle-leaf habitat particularly important, especially in the winter when food is scarce. Evergreen needle-leaf forest is the most suitable habitat type for *R. bieti*, but as PD becomes larger and LPI becomes smaller, monkeys must increase their energy expenditure to travel between the smaller patches. *R. bieti* was found to spend 38.8% of their time feeding, 27.4% moving, 20.9% resting, and 12.9% engaged in other activities. Larger PD and smaller LPI means that *R. bieti* has to move between

suitable patches in order to find enough food and quality sleeping sites, which will increase their activity cost and may also increase the cost of feeding. This is a problem, as *R. bieti* subsists on lichens that provide little energy for much of the year. Primate populations inhabiting small and isolated forest fragments are especially vulnerable to extinction (Arroyo-Rodríguez and Fahrig, 2015; Benchimol and Peres, 2013). Although the results of this landscape analysis indicate that the PD and LPI of the evergreen needle-leaf forest may affect habitat quality for the Yunnan snub-nosed monkeys, the effects of conditions in the understory such as the slope aspect, tree height and trunk diameter, food richness, and human interference are unknown. These factors also affect habitat quality for *R. bieti*.

5. Conclusion

The potentially suitable distribution area for *R. bieti* has increased significantly, along with the total population, but their actual geographic range is shrinking. In our landscape analysis, we found that the PD and LPI of patches of evergreen needle-leaf forest were the main factors restricting the habitat use of *R. bieti*. Although the government has made great strides in the protection of *R. bieti*, pre-regulation destructive logging and the barrier effect of roads and rivers have resulted in the local extinction of *R. bieti* in some potential suitable habitats. In view of this, we propose the following solutions to help restore *R. bieti* to its historical range: (1) Establish a Yunnan snub-nosed monkey national park encompassing all parts of their range to enforce protection and management; (2) Include the areas where *R. bieti* has gone locally extinct within its historical range in the zone of protection; (3) Reintroduce *R. bieti* into the areas of potentially suitable habitat and monitor the recovery of those populations.

Authors' contributions

Wancai Xia, Chao Zhang, Hongfei Zhuang, Xiaofeng Luan conceived the ideas and methodology. Dayong Li, Wancai Xia, Baoping Ren, Xinming He, Jiang Zhou collected the data. Wancai Xia, Chao Zhang, and Hongfei Zhuang analyzed the data. Ali Krzton, Dayong Li and Xiaofeng Luan and Jiang Zhou revised the manuscript and contributed to the development of ideas.

Declaration of competing interest

The authors have no conflicts of interest to declare.

Acknowledgments

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.gecco.2019.e00835.

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