Title

Early successional habitats created through plantation harvesting benefit the Gray Nightjar (*Caprimulgus jotaka*): An 8-year survey in central Hokkaido, northern Japan

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Abstract (204/250 wards)

Early successional habitats and their associated species have been decreasing globally. In contrast, plantations have been expanding and their young stages (stand age ≤ 10 years) can serve as early successional habitats. The Gray Nightjar (*Caprimulgus jotaka*), a nocturnal bird species, breeds and forages in early successional habitats surrounded by forests; its populations have declined since the 1970s in Japan. Because nightjars are more abundant in warmer areas across Hokkaido, northern Japan, habitat creation through plantation harvesting was expected to promote nightjar abundance or occupancy in this region. To explore the effects of plantation harvesting on nightjar occupancy, we conducted an 8-year playback survey in a plantation landscape in central Hokkaido. We considered the effects of elevation as a surrogate for temperature. The results indicated that increasing young forest cover within 500 m of the centroid of each site enhanced nightjar occupancy, whereas elevation negatively affected occupancy. Therefore, at lower elevations, we predict a larger increase in occupancy probability with increasing young forest cover following plantation harvesting. Our results suggest that young forest creation in landscapes can contribute to Gray Nightjar conservation. To effectively create early successional species habitats through plantation harvesting, it is important to consider climate and elevation in the target area.

Keywords (5/5 wards)**:** *Abies sachalinensis*, clear-cut, disturbance-dependent species, forest management, retention forestry

Main text

Introduction

Plantations have been expanding worldwide (FAO 2020), many replacing natural forests (Puyravaud et al. 2010; Hua et al. 2018). Compared with natural forests, plantations have simpler composition and structure, leading to lower species richness and abundance of organisms (Brockerhoff et al. 2008; Chaudhary et al. 2016). Plantations are sometimes converted from open lands (Lindenmayer et al. 2019; Iezzi et al. 2020; Ueno et al. 2022), and their young stages (stand age ≤ 10 years) can serve as alternative or superior habitats for species that prefer early successional habitats created by disturbances (Schlossberg and King 2009; Yamaura et al. 2012a; Ohwaki et al. 2018; Ram et al. 2020). As early successional species have decreased globally (Swanson et al. 2011; King and Schlossberg 2014), plantation harvesting has been expected to contribute to their conservation (Yamaura et al. 2012a, 2012b; Iezzi et al. 2020).

However, the effects of harvesting on early successional species vary spatially. For example, biological responses to habitat management can be contingent on climate (Amano et al. 2011; Betts et al. 2019; Spake et al. 2020), which affects organisms directly via physiological processes (such as maintaining body temperature for animals: Cooper et al. 2006; Camacho 2013), and indirectly via prey resources, vegetation, or land uses (Huston and Wolverton 2009; Yamaura et al. 2011). Temperature can be critical for determining species distributions in mountainous regions, because it changes greatly with elevation (Sergio and Pedrini 2007; Jarzyna et al. 2021). Large-scale, rapid changes in climate due to global warming, coupled with land use change, have threatened many species (Oliver and Morecroft 2014; Díaz et al. 2019). Therefore, it is important to consider the effects of climate or elevation on the distribution or habitat selection of target species for effective conservation by managing habitats.

In Japan, conifer plantations largely replaced natural forests and grasslands from the 1950s to the 1980s (Yamaura et al. 2012b; Ueno et al. 2022), and now account for 41% of the total forest area (Forestry Agency 2017). In the decades following plantation expansion, this forestry decline prevented the creation of young forests (Yamaura et al. 2012b). Semi-natural grasslands and farmlands have been degraded or lost as a result of agricultural intensification and forest transition

from abandoned farmland (Koshida and Katayama 2018; Ushimaru et al. 2018). Furthermore, natural disturbances such as wildfires, landslides, floods, and avalanches are suppressed by humans (Ushimaru et al. 2018; Ohwaki 2018). These land-use changes cause population declines in early successional species (Yamaura et al. 2009; Nakamura 2011; Ohwaki 2018). Recently, large-scale clear-cutting of mature plantations has been conducted to increase the domestic wood supply in Japan (Forestry Agency 2021), potentially enhancing the recovery of early successional species (Yamaura et al. 2012a, 2012b).

The Gray Nightjar (*Caprimulgus jotaka*) is a nocturnal bird species that breeds in open lands surrounded by forests, including young plantations after harvesting and planting (Fujimaki 1973; Tada 2022). The breeding range of this species across Japan contracted by 57% between 1978 and 2002 (Amano and Yamaura 2007). In a previous study, we reported that nightjars were more abundant in warm areas across Hokkaido, northern Japan (Kawamura et al. 2016), where plantation harvesting was anticipated to promote nightjar abundance or occupancy. In this study, we conducted an 8-year playback survey for the Gray Nightjar in 18 plantations, 15 of which were harvested before or during the study period, and three natural forests, for a total of 21 stands in central Hokkaido. We analyzed the effects of stand harvesting and surrounding young forest cover on nightjar occupancy, as well as the effects of elevation as a surrogate of temperature.

Materials and Methods

Study area and sites

This study was conducted from 2015 to 2022 at sites established in the Retention Experiment for Plantation Forestry in Sorachi, Hokkaido (REFRESH project: 43°34′37″–43°39′26″N, 142°05′27″–142°09′33″E). In this experiment, we have been investigating whether retaining some canopy trees in harvested Todo fir (*Abies sachalinensis*) plantations might enhance the multifunctionality of the plantations, including the habitat functions of plants, insects, and birds (Yamaura et al. 2018). The study sites were located at the foot to mid-slope of Mt. Irumukeppu (864 m a.s.l.) in landscapes dominated by Todo fir plantations (Fig. 1). Sites were selected in five harvesting types: clear-cutting (CC); small (SS), medium (SM), and large (SL) amounts of single-tree retention; and group retention

(GR). We also had two types of control site: unharvested mature plantation (PC) and natural forest (NC). Each treatment had three replicates, for a total of 21 stands.

PC sites were dominated by planted Todo firs with stand ages of 48–51 years in 2015. NC sites consisted mainly of linden (*Tilia japonica*), Mongolian oak (*Quercus crispula*), and painted maple (*Acer pictum*). In the SS, SM, and SL sites, 10, 50, and 100 sparsely distributed broadleaved trees/ha were retained, respectively; these mainly consisted of birch (*Betula platyphylla*, *Betula ermanii*, and *Betula maximowicziana*), linden, and Mongolian oak. In each GR site, one unharvested patch of Todo firs (60 m × 60 m: 0.36 ha) was retained at the center of the harvested area. From 2014 to 2016, one site per harvesting type was harvested each year (Appendix S1). Todo fir seedlings had been planted the next spring after harvesting, and weeding had been conducted once or twice annually (Akashi et al. 2017; Yamaura et al. 2018).

We did not survey sites undergoing harvesting operations in 2015 and 2016 (Appendix S1). Thereafter, we surveyed all sites in each year except 2021 (and except NC3 in 2022: Appendix S1). For five plantation sites, we surveyed before and after harvesting. As unharvested sites, eight plantations (CC3, GR3, SS3, SM3, SL3, PC1, PC2, and PC3) and three natural forests (NC1, NC2, and NC3) were surveyed and only PC and NC sites were surveyed on multiple years (Appendix S1). The area of each site was 5.0–8.2 ha (Appendix S1), and the distance between sites was at least 200 m, except for one pair (SM2 to SS3, Fig. 1; Akashi et al. 2017; Yamaura et al. 2018). The elevation range for all sites was calculated by averaging the values in a 10 m digital elevation model, as 236–506 m (Fig. 1).

Bird survey

To examine Gray Nightjar occupancy, we conducted a playback survey (Bibby et al. 2000). In each survey year, we visited each site three times during the Gray Nightjar breeding season, from late May to early August (Saiki 2016). Surveys were conducted between 19:30 and 03:30, when Gray Nightjars are active, avoiding rain and strong winds (> 5 m/s). The male territorial call of the Gray Nightjar was broadcast for 5 min at the edge of each site and any nightjar callbacks within the next 5 min were recorded (i.e., 10 min/survey). In 2022, the survey consisted of three repetitions of 2 min of playback and 1 min of recording, plus an additional 1 min of recording at the end, for a total of 6 min

of playback and 4 min of recording, corresponding to the sampling method used in our ongoing study. We recorded the number and location of any individual calling in each site and a 50 m buffer zone around each site, and considered these individuals as users of the site.

Kawamura et al. (2016) used a similar survey method and found that the detection probability of each territorial individual per survey was ~0.65 and remained constant regardless of the survey season and time of day. We surveyed most sites within one night for each visit. Therefore, we judged that differences among survey seasons or survey conditions were relatively small and that nearly all individuals could be detected within three visits to each site. For playback, we used speakers [(2015–2020) PDX-B11: Yamaha, Hamamatsu, Japan; (2022) SoundLink Revolve+ II: Bose Corporation, Framingham, MA, USA] connected to players (iPhone 5s or iPhone SE: Apple, Cupertino, CA, USA). Although speaker and player models varied, the nightjars responded similarly to playback through different devices, and the amplitudes were nearly identical (66.8 dB for PDX-B11 and 66.7 dB for SoundLink Revolve+ II at 5 m from the speaker) and audible throughout each site (Kawamura and Yamaura, personal observation).

Environmental variables

Harvesting treatment and forest type

As a categorical variable at the stand scale, we considered the harvesting treatment (i.e., harvested vs. not harvested). Although retained trees may provide perches for nightjars (Winiger et al. 2018; Camacho 2014) and different harvesting types (i.e., CC, SS, SM, SL, and GR) may have different conservation roles, we treated all harvesting types as a single category. This was because the Gray Nightjar prefers both sparse forests and clear-cut areas (Yamaguchi and Mitarai 2018; Tada 2022) and nightjar occupancy or abundance was too low to examine the effects of harvesting types or the proportion of retained trees. The other categorical variable at the stand scale was forest type (plantation vs. natural forest).

Young forest cover in landscapes

We considered that the young forest cover (stand age ≤ 10) surrounding each site could affect nightjar occupancy. This was because the Gray Nightjars observed in this study were expected to use

the areas surrounding the study sites, considering that the territories (i.e., defended areas for breeding) of the closely related European Nightjar (*Caprimulgus europaeus*) have a radius of approximately 200 m and their home range has a radius of several kilometers (Alexander and Cresswell 1990; Sharps et al. 2015). Furthermore, windthrow occurred in forests adjacent to some sites, immediately followed by harvesting and planting in these sites (Fig. 1, Appendix S1). Kawamura et al. (2016) detected the effects of forest cover within 4 km (home range scale) of each sampling site on nightjar abundance. However, since the large (e.g., 3.0 km radius) buffers of YFC overlapped well (Holland et al. 2004), we used YFC within 500 m (500 m YFC; i.e., territory scale) from the centroid of each site (Fig. 1). This was performed using the ArcGIS Desktop v10.6 software (ESRI, Redlands, CA, USA). Forest data were obtained from Hokkaido Government Opendata CC-BY4.0 (https://www.pref.hokkaido.lg.jp/sr/dyr/DOP.html [date of data acquisition: March 31, 2016, in Japanese] and https://www.pref.hokkaido.lg.jp/sr/srk/98818.html [date: December 31, 2019, in Japanese]). The 500 m YFC was calculated for each survey year, but the young forests created by harvesting or windthrow after the date of the original data acquisition were not reflected. Therefore, we revised the data when we found young forests in the field.

Elevation

We used elevation as a surrogate for temperature at the local scale (Fig. 1, Appendix S2). Elevation was calculated by averaging the 10 m digital elevation model provided by the Geospatial Information Authority of Japan (https://fgd.gsi.go.jp/download/ref_dem.html (in Japanese]). The 500 m YFC and elevation were standardized prior to model runs. The 500 m YFC, elevation, and the area of sites were not strongly correlated ($|r| < 0.5$).

Statistical analyses

We analyzed using the R v4.1.0 software (R Core Team 2021). To examine the effects of young forest creation through plantation harvesting on nightjar occupancy, we used generalized linear mixed models, using the 'glmer' function in the *lme4* package (Bates et al. 2015). Our analyses involved two steps: the harvest/non-harvest model, examining the effects of harvesting treatment (plantation harvesting at the stand scale); and the young forest cover model, analyzing the effects of

500 m YFC (young forests mainly created by plantation harvesting). Because the harvesting treatment and 500 m YFC were not independent (i.e., harvested survey stands had more young forest cover, they were not incorporated into the model simultaneously.

Harvest/non-harvest model

We constructed models, where the response variable was nightjar occurrence/non-occurrence in each site in each year and the explanatory variables were the harvesting treatment and elevation. We assumed that the response variable followed a binomial distribution, and used a complementary log-log link function (cloglog). Thus, we modeled the probability that at least one individual occurs at each site assuming that nightjar abundance followed a Poisson distribution (Kajihara et al. 2016). This method allowed us to correct nightjar occupancy for site area differences by considering the proportional increase in abundance with the area of site *i* [i.e., setting log($Area_i$) as an offset term]. To account for pseudoreplication associated with repeatedly surveying the same sites, we used site ID as a random intercept (i.e., random site effect). We constructed models for all possible combinations of the explanatory variables and selected the model with the lowest AIC as the best model (Burnham and Anderson 2002), using the 'dredge' function in the *MuMIn* package (Bartoń 2022). We calculated the 95% confidence intervals of the coefficients of each explanatory variable in the best models using the 'confint.merMod' function (method = "profile"). We interpreted the effect of an explanatory variable as significant when the interval did not overlap with zero. The effect of forest type was confounded with that of harvesting treatment because most plantation sites were harvested and all surveyed natural forests were unharvested controls. Forest type was therefore excluded from the explanatory variables.

Young forest cover model

Using the same method, we also conducted the analysis with 500 m YFC, forest type, and elevation as explanatory variables. In contrast to the harvest/non-harvest model, forest type was included as an explanatory variable in this model because the VIFs of 500 m YFC and forest type were below 10 (5.10 for 500 m YFC and 6.22 for forest type: Dormann et al. 2013). We interpreted the effect of young forest creation through harvesting on nightjar occupancy considering the results of both the harvest/non-harvest model and the young forest cover model.

Consideration of the effect of site area and the annual variation in nightjar occupancy

Although we assumed that the nightjar abundance was directly proportional to the area of site *i* ($Area_i$) and set the offset term to $\log(Area_i)$ [i.e., the coefficient of $\log(Area_i)$ was fixed at 1], the coefficient of $\log(Area_i)$ could exceed 1 if Gray Nightjars preferred larger harvested areas. In our preliminary analysis, we extracted data only from harvested sites and constructed a model using $\log(Area_i)$ and elevation as explanatory variables. The results showed a coefficient of $\log(Area_i)$ of −0.40, with a wide confidence interval (Appendix S3). Therefore, we used $\log(Area_i)$ only as an offset term in our analyses, assuming a small area effect of harvesting on nightjar occupancy.

In our preliminary analysis, we considered annual variation in nightjar occupancy in addition to the random site effect, by including the survey year as a random intercept (random year effect) in the models. However, the random year effect did not explain the variation in our data, and intercept and coefficient estimates were nearly identical to those for the model that did not include a random year effect (Appendix S4). Therefore, only the random site effect was considered in our analyses.

Results

During the 8-year study period, we detected Gray Nightjars in seven of the 21 sites (Fig. 1). Five of these sites were harvested, including one clear-cutting site (CC2), two single-tree retention sites (SS2 and SL1), and two group retention sites (GR1 and GR3). The earliest nightjar occurrence after harvesting was the following year (GR3), and the latest was 6 years later (CC2) (Appendix S1). Nightjars also occurred in two unharvested natural forest sites, although they only began to occupy NC2 after windthrows occurred in adjacent forest areas (the 500 m YFC was relatively low at both sites: Appendix S1, Fig. 2). Nightjars were detected in multiple years at four sites (GR3, SS2, SL1, and NC2) and in GR3 during four consecutive years after harvesting (Appendix S1). The only site where multiple individuals occurred during a single visit was SS2, where two individuals were detected in 2017; one of these individuals may have also called back in the neighboring site, GR3. Other occurrences in the same year were not considered to be in the territory of the same individual because the calling locations were >700 m apart.

Harvest/non-harvest model

In the best model, harvesting treatment and elevation were included as the explanatory variables (Appendix S5). There was a single competing model (i.e., the second-ranked model with $\Delta AIC < 2$) that had the elevation as an explanatory variable (Appendix S5). The coefficient of harvesting treatment in the best model was estimated to be positive, but the difference in nightjar occupancy between harvested and unharvested sites was not significant at the 5% level (Table 1a). Elevation negatively affected occupancy (Table 1a, Fig. 1).

Young forest cover model

The young forest cover model showed a positive effect of YFC and a negative effect of elevation. The best model contained 500 m YFC, elevation, and forest type as explanatory variables (Table 1b). Nightjar occupancy was higher in areas with higher 500 m YFC and lower elevation (Table 1b, Fig. 2). Occupancy tended to be low in plantation sites compared to natural forest sites (Table 1b). Although we did not use the interaction term between 500 m YFC and elevation, the absolute increment in occupancy probability at plantation sites with increasing 500 m YFC was predicted to differ greatly according to elevation (Fig. 2). In 5 ha plantation sites where the 500 m YFC was 0%, occupancy probability was almost zero across the elevation range. As the 500 m YFC increased to 25% (~20 ha increase in young forest), the occupancy probability increased to 0.94 and 0.25 in areas with low (236 m) and medium (383 m) elevation, respectively. In contrast, in high-elevation sites (506 m), occupancy probability increased only slightly (0.04).

Discussion

Positive effects of young forest cover within 500 m

Nightjar occupancy was enhanced by increasing young forest cover (mainly young plantations, including the harvested survey sites) within 500 m. These results indicate that young forest creation through plantation harvesting can contribute to Gray Nightjar conservation. The Gray Nightjar tends to nest in young plantations (Fujimaki 1973; Tada 2022), and assuming a similar territory range to the European Nightjar, its territory is likely contained within an area of 500 m (Bowden and Green

1994; Sharps et al. 2015). Open environments near forests are also suitable for the visual detection of insects flying from mature forests (e.g., large moths) and for catching these prey insects in the air (Sierro et al. 2001; Evens et al. 2018, 2020). Studies have shown that European Nightjars frequently forage near their nests during the incubation period (Alexander and Cresswell 1990; Cross et al. 2005) and that foraging sites near the nests are superior in terms of low commuting costs (Sharps et al. 2015; Evens et al. 2018). Therefore, forested areas with high young forest cover provide suitable nesting and foraging environments.

In western Tokyo, central Japan, Yamaguchi and Mitarai (2018) reported that nightjar occupancy within 250 m increased with increased clear-cutting area, exceeding 0.5 in sites with a clear-cut area of > 4.3 ha (0.5–27.9 ha). Young forest creation through clear-cutting also positively affects European Nightjar occupancy, although the minimum area for occupation differs among studies (0.7 ha: Wichmann 2004; 1 ha: Scott et al. 1998; 10 ha: Ravenscroft 1989). In this study, the maximum area of harvested survey stands was 8.2 ha and the effects of the stand harvesting treatment were not significant. However, nightjar occupancy increased more clearly in sites where young forest cover within 500 m exceeded 10% (7.9 ha). These results suggest that harvesting stands (5.8−8.2 ha in our case) only would not provide sufficient habitat and surrounding young forests also contribute to nightjar occupancy. Thus, in landscapes where forestry is active, multiple harvested stands would support nightjar occupancy.

Broadleaved natural forests and retention cutting stands as habitats

Natural forests tended to be preferred by nightjars over plantations, although only three natural forest sites were surveyed. Given that young forest cover, including harvested plantations, positively affected nightjar occupancy, our results indicate a poor habitat function of unharvested plantations. This finding is supported by those of a previous study conducted in eastern Hokkaido (Toyoshima et al. 2013), which occasionally detected early successional birds in stands older than 60 years in natural forests, but not in mature plantations. Nightjars did not prefer mature plantations with a high tree density. Nightjars were also found in harvested sites with retention rates of 10–100 broadleaved trees/ha and in a site with a retained Todo fir patch (0.36 ha). Retention cutting areas generally harbor fewer early successional species than clear-cutting areas because of their smaller open spaces

297 (Fedrowitz et al. 2014). However, both the Gray Nightjar and European Nightjar prefer sparse forests
298 as habitats (Sierro et al. 2001; Winiger et al. 2018; Tada 2022), and the Red-necked Nightjar
299 (*Caprimulgus ruficollis*) selects areas near tall vegetation as refuges from predators when descending
300 to the ground (Camacho 2014). Future studies should investigate the habitat function of natural and
301 sparse managed forests for early successional species. For nightjars with a large home range and
302 wide range of habitat uses, the interactive effects of forest type with landscape variables are also
303 needed for evaluation (Knight et al. 2021b).
304
305 **Negative effects of elevation**
306 Nightjars were more abundant in areas with lower elevation. By contrast, no elevation effect on
307 nightjar occupancy was detected in a previous study conducted in warmer regions of Japan
308 (Yamaguchi and Mitarai 2018); thus, our results were likely a reflection of temperature effects. The
309 negative impacts of cool conditions have been reported for nocturnal insectivores (European
310 Nightjar: Morris et al. 1994, bats: Rydell et al. 1996; Davy et al. 2022). There are two possible
311 explanations for this finding. First, the availability of flying insects (e.g., moths) as prey resources
312 tends to be lower under cooler conditions (Turner et al. 1987; McGeachie 1989; Roy et al. 2001). In
313 this area, elevation was the most important determinant of understory plant communities, with a
314 negative impact on their species richness (Akashi et al. 2022). Second, nocturnal flying vertebrates
315 may be physiologically sensitive to temperature; for example, the Red-necked Nightjar prefers warm
316 roads on cold nights (Camacho 2013; De Felipe et al. 2019). Although nightjar occupancy may also
317 be influenced by other environmental factors correlated to elevation, such as young forest cover at
318 larger scales (Kawamura et al. 2016), pastures, or farmlands (Evens et al. 2021), cool areas tend to
319 restrict the distributions of Caprimulgidae species, which evolved in the tropics and share similar
320 morphology and foraging ecology (Cleere and Nurney 1998).
321
322 **Implications for forest management and conservation**
323 Despite the moderate spatial extent of our study region (6 × 12 km), there were spatial differences in
324 the nightjar occupancy increment with increasing young forest cover. The absolute increment in
325 nightjar occupancy with young forest creation was predicted to be larger in areas with lower

elevation, according to nightjar distribution model results for Hokkaido. The determining factors of species distribution and the degree of these effects can vary regionally (Crosby et al. 2019; Knight et al. 2021a). For example, Knight et al. (2021b) demonstrated that the disturbance type and time since disturbance were important factors for determining the nesting habitat use of the Common Nighthawk (*Chordeiles minor*) and their habitat usage duration in disturbed areas was longer in landscapes with a higher proportion of pine forests. The area effects of harvesting itself and landscape/climate-dependent area effects on nightjar occupancy may be detected using sites with wider ranges of each environment. Nevertheless, our results provide valuable insights for designing conservation strategies for early successional species. Considering the effects of climate, elevation, and landscape on conservation target species is essential to implement plantation harvesting for effective habitat creation and promoting biodiversity conservation efforts in balance with timber production.

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Declaration of interest statement

None declared.

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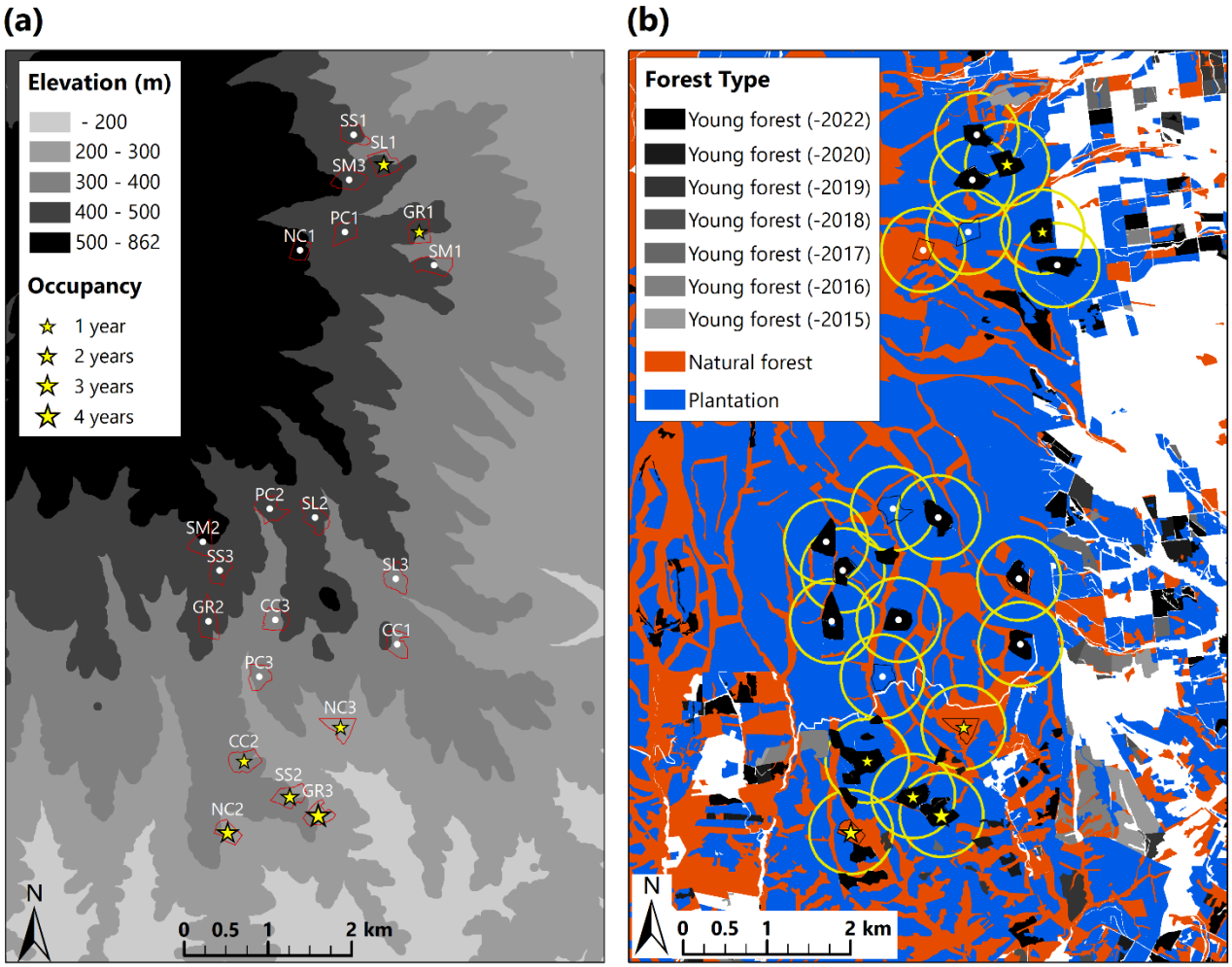
540

Table and Figures

Table 1. Parameter estimates of the best (a) harvest/non-harvest and (b) young forest cover models. For each model, R^2 was calculated using the 'r2beta' function in the package *r2glmm* (Jeager 2017). Abbreviations: Harvest, the categorical explanatory variable of harvested sites compared to non-harvested sites; 500-m YFC, young forest cover within 500 m; Plantation, the categorical explanatory variable of plantations compared to natural forests; 95%CI.l and 95%CI.u, lower and upper 95% confidence interval limits, respectively. The intercept represents (a) non-harvested forests and (b) natural forests. Bold indicates the significance of the explanatory variable at the 5% level. The ΔAIC (i.e., AIC difference from the best model) of the second-rank model was (a) 1.44 for the harvest/non-harvest model and (b) 2.02 for the young forest cover model. These second-rank models contained fewer explanatory variables (see details in Appendix S5).

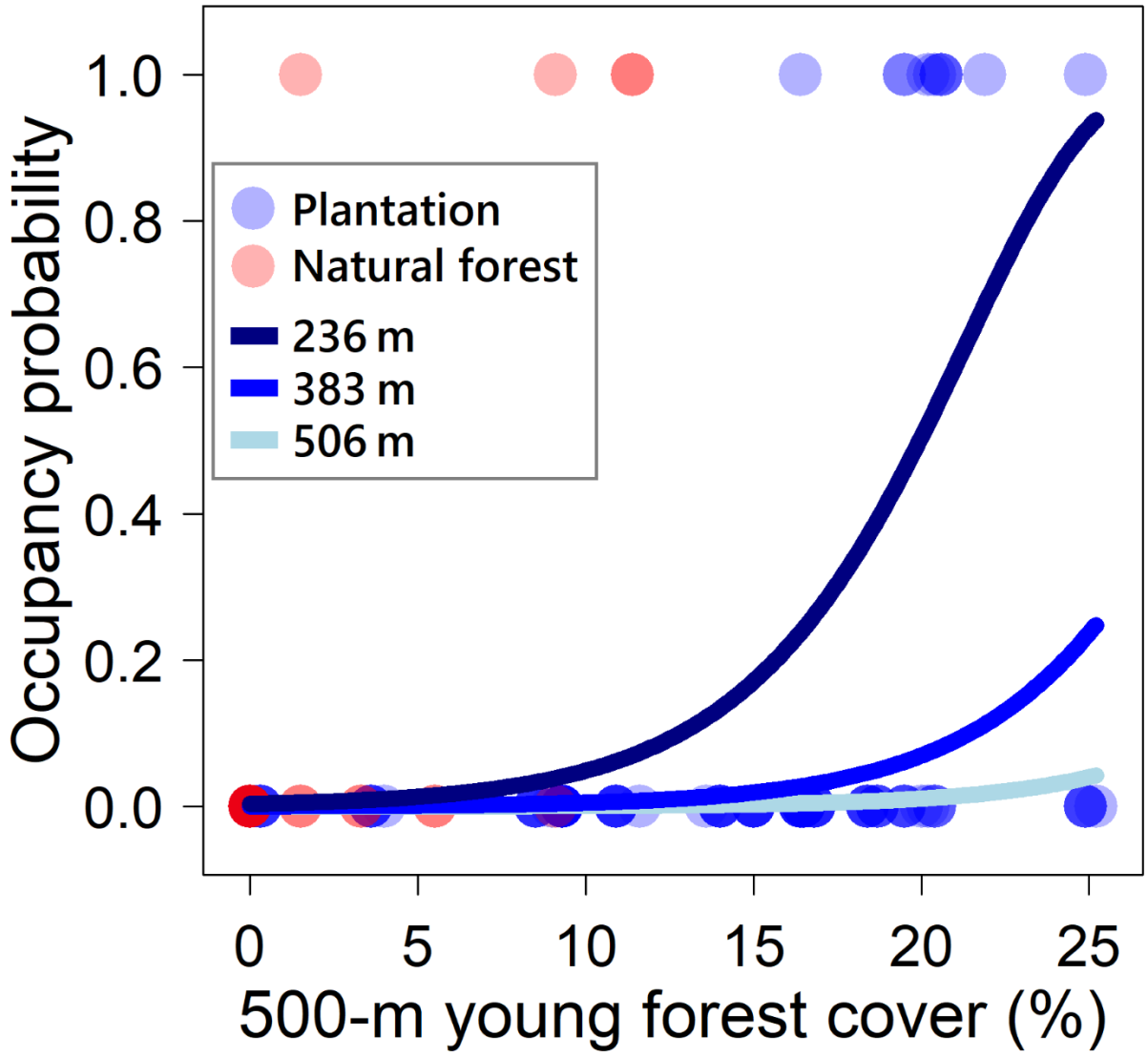
	Variables	Estimates	SE	z value	Wald *p*	95%CI.l	95%CI.u
(a)	R^2	0.21				0.10	0.36
	Intercept	–6.37	1.12	–5.68	< 0.001		
	Harvest	1.47	0.86	1.70	0.089	–0.08	4.06
	Elevation	**–1.83**	0.51	–3.57	< 0.001	**–3.38**	**–0.95**
(b)	R^2	0.16				0.07	0.31
	Intercept	–3.07	1.45	–2.11	0.035		
	500-m YFC	**1.96**	0.92	2.13	0.033	**0.66**	**4.65**
	Elevation	**–1.09**	0.54	–2.02	0.044	**–2.40**	**–0.04**
	Plantation	**–3.33**	2.08	–1.60	0.110	**–9.36**	**–0.07**



554

555 **Fig. 1.** Map of the study area. Symbols at the center of each site indicate nightjar occupancy status;
556 the size of the star indicates the number of years in which nightjars occurred, and white circles
557 indicate no occurrence. Shading represents differences in (a) elevation and (b) forest type. In young
558 forests, lighter colors indicate higher stand age. For example, "Young forest (–2016)" indicates that
559 these stands were treated as young forest stands until 2016 and thereafter as > 10-year-old stands.
560 The yellow circles around each site represent 500-m radius zones.

561



562

563 **Fig. 2.** Estimated relationship between nightjar occupancy probability and explanatory variables in
564 the young forest cover models. Solid lines are predictions by the best performing models; colors
565 indicate different elevations. Blue and red translucent circles represent the observed values in
566 plantation and natural forest sites, respectively. Site area differed among these observed values, and
567 was fixed to 5 ha for predictions.

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