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Current nature reserve management in China and effective conservation of threatened pheasant species

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Protected areas, such as nature reserves, are essential for effective conservation of threatened species through protection and management of populations and habitats. Habitat evaluation is a key method that has been frequently used to assess the effectiveness of protected areas. Previous research has mainly focused on species conservation related to changes in land cover and habitat fragmentation while few studies have examined changes in microhabitat structure. Using a multi-scale habitat change analysis (i.e. regional, macro- and microhabitat) in a temporal framework, we assessed the effectiveness of current nature reserve management for the habitat protection of the Reeves's pheasant *Syrmaticus reevesii*, a vulnerable, forest-dwelling species. We measured land use/land cover changes inside and outside the Dongzhai National Nature Reserve (DNNR), in 2002 and 2013 corresponding to the times at and after the establishment of the DNNR. We also compared differences in habitat fragmentation patterns and microhabitat structure and composition between the two periods. Results show that the forest coverage has slightly increased both inside and outside DNNR, and habitat fragmentation metrics have not changed substantially since the establishment of DNNR. Significant differences were detected in microhabitat structure and composition between 2002 and 2013. After more than 10 years of no disturbance, canopy cover and density of the shrub layer increased, while herbaceous plant height declined. The observed changes reduced resource availability resulting in increased foraging time for pheasants and increased predation rates. This suggests that current nature reserve management systems may have negative impacts on the conservation of the Reeves's pheasant. We propose that the Regulations of Nature Reserves in China should be revised to account for the habitat requirements of different threatened species with varied life history traits.

Protected areas are an effective strategy to mitigate present biodiversity and habitat loss (CBD 2010), and now they cover over 15% of the world's land surface (Geldmann et al. 2015) and are cornerstones for biodiversity conservation (Chape et al. 2005, Rodrigues et al. 2004, Geldmann et al. 2013). Success of protected areas relies on effective management (Watson et al. 2014). However, protected areas can reduce rates of habitat loss and species population declines only if they are appropriately managed for target species (Geldmann et al. 2013). Without effective management for targeted species, protected areas exist only as 'paper parks' and are unlikely to deliver benefits for conservation (Joppa et al. 2008). Effective management of protected areas requires evaluation and analysis of species and habitats of interest.

Assessment of the success of protected areas in conserving rare species has been steadily increasing over the past decades (Leverington et al. 2008). In particular, assessing how well protected areas maintain or improve native habitat and threatened species populations has become a research priority (Craigie et al. 2010, Joppa and Pfaff 2011, Laurance et al. 2012). Previous studies have used the response of threatened species population size or occupancy as an index to assess protected areas effectiveness (Zafir et al. 2011, Maisels et al. 2013, Riggio et al. 2013). A number of other studies have used the trajectories of habitat change, loss, and degradation as a metric for measuring the performance of protected areas (Liu et al. 2001, Clark et al. 2013). Few studies have examined the importance of vegetation composition and structure changes at the microhabitat scale in assessing effectiveness of protected areas. This approach should be examined as animals often select different habitat components in a spatially hierarchical process (Johnson 1980, Mayor et al. 2009). At the microhabitat scale, the physical structure of vegetation provides important functions such as resting, roosting and perching sites as well as shelter against abiotic conditions and predators (Hildén 1965, Pounds 1988, Boonman et al. 1998). Plant composition also influences resource availability (Rotenberry 1985, Fleming et al. 1993), and vegetation structure and composition are significant components of wildlife habitat (DeGraaf et al. 1998, Linderman et al. 2005, Deppe and Rotenberry 2008). Therefore, a complete habitat assessment of protected area outcomes should include microhabitat structure and composition monitoring.

As one of the signatory countries to the Convention on Biological Diversity, China launched the Wildlife Conservation and Nature Reserve Construction Project in 2001 to reverse the trends of habitat degradation and biodiversity loss (State Forestry Administration 2001). Since then, nature reserves, which were the main part of protected areas, have doubled in number from 1227 in 2001 to 2729 in 2014, increasing the total of land coverage from 9.95% to 14.80% of China (State Council 2015). In China, nature reserves are divided into three zones, the core, buffer and experimental zones. The core zone is designed to protect threatened species or natural ecosystems, the experimental zone allows for sustainable development, and the buffer zone is placed in-between the two zones to reduce the impacts of human activities on the core zone (McNeely 1994, Xie et al. 2004). No human activity or disturbance is allowed in the core or buffer zones without permission (State Council 1994). This management system is targeted towards habitat maintenance and coverage (Regan et al. 2008), and as a result deforestation rates have significantly declined in many nature reserves (Ren et al. 2015). With this management strategy, forest habitat loss within nature reserves will likely be reduced or stopped completely. However, research has suggested that some forest-dwelling threatened species populations are still declining even in nature reserves, and habitat degradation is the major driver (He et al. 2011, Li et al. 2014, Zhou et al. 2015). This suggests that habitat coverage change analysis is not sufficient for assessing nature reserve management effectiveness, and information on the microhabitat structure and composition changes should be incorporated into these studies.

Galliformes are one of the most threatened groups of birds due to their high site fidelity and low dispersal ability (WPA and IUCN/SSC Re-introduction Specialist Group 2009), making them sensitive to habitat degradation (Wang et al. 2008). In China, most galliform species are endangered, vulnerable, or rare, and habitat degradation or habitat loss have been considered the main reasons for such a population decline (Zheng and Wang 1998). They are considered an important target species-group in the National Wildlife Rescue Program of China (State Forestry Administration 2001), and 38 of the 63 species are listed as nationally protected (Zheng and Wang 1998, Zhang et al. 2003). Galliformes are also the most studied order of birds in the country and research on them has contributed directly to conservation management and policy at all administrative levels (McGowan et al. 2009). Hence, many nature reserves were established to conserve endangered pheasant species and their habitats (State Forestry Administration 2001).

The Reeves's pheasant *Syrmaticus reevesii* is an endemic and threatened forest-dwelling galliform species in China (Cheng et al. 1978). This species naturally inhabits forests with sparse undergrowth (Wu et al. 1991, 1994). Because of illegal hunting, habitat loss and fragmentation, the distribution of this pheasant has become divided into eastern and western regions (Zheng and Wang 1998, Collar et al. 2001). Some genetic isolation might exist between these two regions, but no significant genetic isolation were discovered among the different parts of the eastern region (Wang et al. 2009). It is listed on the IUCN Red List as vulnerable (IUCN 2015) and is a national second-grade protected wildlife species in China (State Council 1988). More than 40 national nature reserves were established for targeted conservation of this pheasant species. However, its wild populations are still declining even within the nature reserves (Zhou et al. 2015). An urgent assessment of whether this species benefits from the current management system of nature reserves is needed.

We use the Reeves's pheasant as a model to evaluate the effectiveness of the current nature reserve management system in protecting threatened species habitat in China. We measured habitat characteristics of the Reeves's pheasant at regional, macro- and microhabitat scales in a nature reserve between early 2002 and 2013. Our objectives were to: 1) identify the dimensions of habitat variables that have changed significantly; 2) identify the primary drivers of habitat degradation under the current nature reserve management system; and 3) provide recommendations to improve the effectiveness of current nature reserves management for protecting threatened species and their habitats.

Material and methods

Study site

The Dongzhai National Nature Reserve (DNNR) (31°40′N, 114°24′E) is located on the northern slopes of the Dabie Mountains, Henan Province in central China (Fig. 1). It was a former forest farm and became a national nature reserve in 2001, located in the eastern region of Reeves's pheasant range. The climate is northern subtropical, warm and humid with a mean annual temperature of 15.1°C (range: 13.2°C–40.1°C) and a mean annual precipitation of 1209 mm. The native vegetation is characterized as mature forests dominated by various oak species *Quercus* spp., masson pine *Pinus massoniana*, dyetrees *Platycarya strobilacea*, beautiful sweetgum *Liquidambar formosana* and Hupeh rosewood *Dalbergia hupeana* (Xu et al. 2007, 2009).

The establishment of this nature reserve was primarily for habitat and population protection of the Reeves's pheasant (Xu et al. 2007). This pheasant is largely concentrated in the core areas of the DNNR (Song and Qu 1996, Xu et al. 2006), and no logging or human interventions have disturbed this area since it became a national nature reserve in 2001.

Image data acquisition and classification

Georeferenced Landsat 5 TM and Landsat 8 OLI images (path 123, row 38) were obtained from the USGS EarthExplorer for 3 September 2002 and 17 September 2013. These images were selected because they were cloud-free and correspond closely with the start of the dry season, when cultivated fields can be easily distinguished from forests under supervised classification.

Image classification was carried out using ENVI ver. 4.8 (ITTVIS, <www.ittvis.com/>), prior to the analysis. Sample training data were obtained from stable land cover

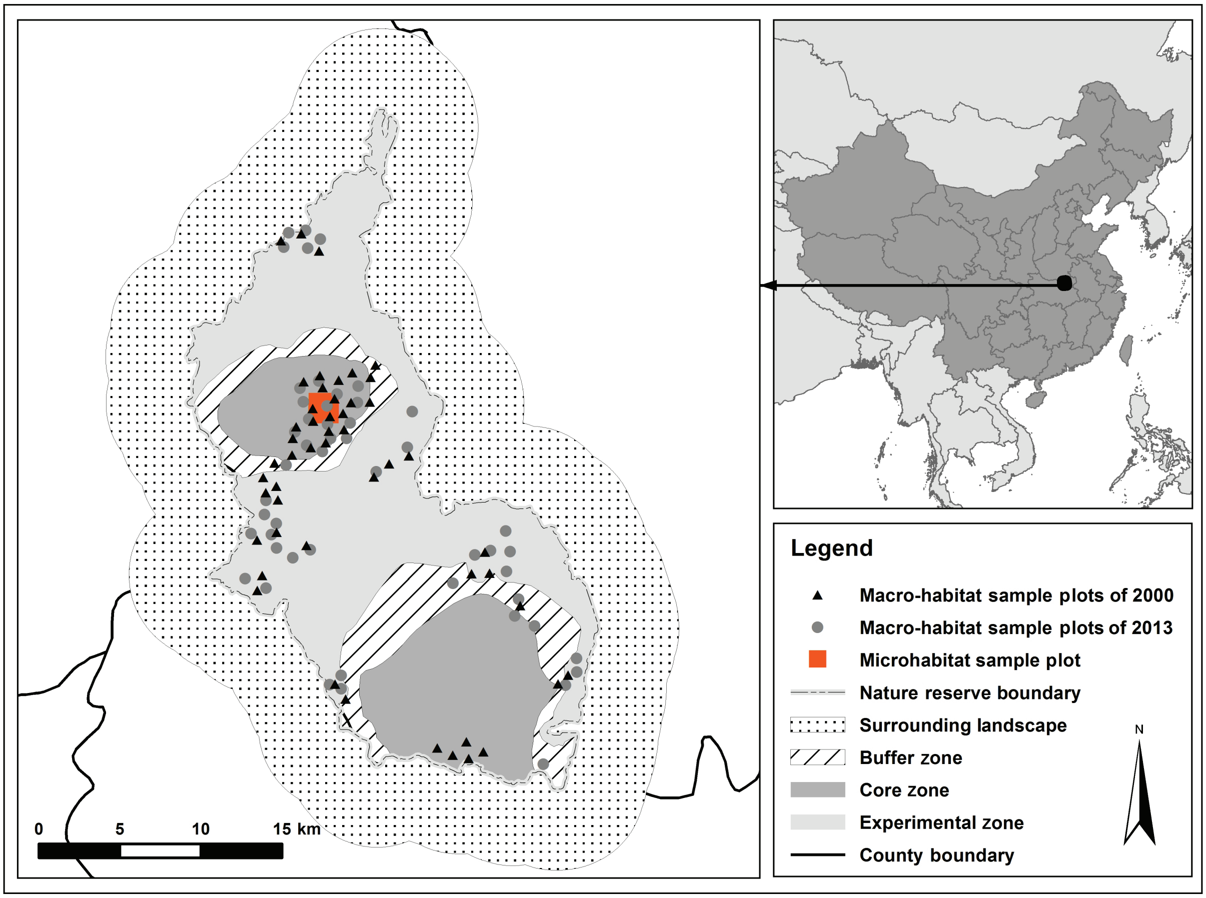

Figure 1. The location of core zone, buffer zone, experimental zone and surrounding landscape area, with the signs of habitat fragmentation sample plots and forest structure sample plot in the Dongzhai Nature Reserve. The upper right corner indicates the location of Dongzhai Nature Reserve in China.

areas identified from fieldwork and vegetation coverage data from the Management Bureau of DNNR. The 2013 Landsat OLI images were classified into coniferous forest, broadleaf forest, cultivated fields, wetland, water bodies, artificial surfaces and bare land by support vector machine classification. We assessed the accuracy of the land cover classification map using contemporary high spatial resolution images from Google Earth (Biradar et al. 2009). We processed the 2002 Landsat 5 TM images using the same procedures and the same sample training data set. The accuracy of the land-cover map obtained for 2002 could not be evaluated due to the unavailability of reference data for that time period. Nevertheless, the accuracy of this map is expected to be similar to that of the 2013 land-cover map, since map production followed the same procedures and used the same sample training data (Carter et al. 2013).

To compare land cover changes inside and outside DNNR, three subsets of the classified images were generated representing the core zones, the non-core zone (this includes the nature reserve's buffer and experimental zone), and surrounding landscape using ArcGIS ver. 10.2 (Esri Inc. <www.esri.com/>). The surrounding landscape was a 5 km buffer area around the boundary of the nature reserve. This particular buffer outside the DNNR provided similar area and environmental conditions to the reserve, while avoiding heterogeneity in spatial variables that could otherwise bias the assessment (Mas 2005).

Multi-scale habitat comparison

We chose regional (inside and surrounding the nature reserve), macrohabitat (species home range size) and microhabitat scales (10 × 10 m quadrats) to detect large to fine scale habitat changes between 2002 and 2013. At the regional scale, we employed an analysis of land use/land cover change to evaluate the impact of nature reserve establishment. We calculated the change rate and total extent between 2002 and 2013 in the core zones, non-core zone, and surrounding landscape. To measure the change of forest habitat cover, we merged the land cover type of cultivated fields, wetland, water bodies, artificial surfaces, and bare land as non-forest, and coniferous and broadleaf forest as forest. The forest was defined based on the plant species present (dominant tree species such as oak, fir or pine), their composition, and the proportions of the area that they occupied (Xu et al. 2007). Then we calculated reforestation and deforestation rate in the core zones, non-core zone and surrounding landscape.

At the macrohabitat scale, we compared landscape fragmentation patterns between 2002 and 2013 to quantify changes in habitat structure. In order to ensure the independence of the data, we selected 50 points (Fig. 1) with at least

1 km spacing from the surveys presence records in the two periods to represent suitable habitat for further analysis. This distance was close to the diameter of the largest home range size (i.e. ca 300 ha) in the study area, and the home range size remained stable during the past decade (Xu et al. 2007, Wang et al. 2009, 2012). We then used FRAGSTATS 4.2 (McGarigal et al. 2012) to quantify class- and landscape-level metrics separately in the two periods with a radius of 1 km from the 2002 and 2013 land cover map. Class-level metrics were calculated based on conifer forest type and landscape-level metrics were calculated based on the land cover within each circle (Table 1). Finally, we tested the data normality by Shapiro–Wilk's *W*-tests, and applied independent samples *t*-tests or Mann–Whitney *U*-tests to compare the differences between 2002 and 2013, depending on whether data were normally distributed or not.

The pheasants were mainly concentrated in the core areas of the DNNR (Song and Qu 1996, Xu et al. 2006). We used 10×10 m quadrats to detect changes in the structure and composition of the microhabitat in the core areas (Fig. 1). We randomly selected 25 quadrats in the core area from our systematically sampled (at 200 m intervals) habitat that was sampled in July 2002. The microhabitat structure and composition included 13 variables: 1) tree diversity; 2) tree density, counting the number of trees by species in each plot; 3) tree diameter (DBH in cm) (Bertin 1977); 4) tree height (m) estimated by a clinometer, using 10 trigonometric readings (Bibby 2000); 5) canopy cover (%), assessed as a percentage through a sighting tube (James and Shugart Jr 1970) with 10 readings in each plot; 6) shrub richness, counting the number of shrub species in five diagonal 1×1 m quadrats; 7) shrub height (m); 8) grass height (cm); 9) depth of leaf-litter (cm), measured by a ruler in representative quadrats; 10) shrub cover; 11) grass cover; 12) leaf-litter cover, estimated by visual estimation as a percentage (%) in five diagonal 1×1 m quadrats; 13) shrub density, calculated by means of a standard 30×50 cm checkered board in the shrub layer (Fuller et al. 1989). To evaluate the microhabitat changes of our study area, we resampled and selected an additional 25 plots in July 2013.

We applied Mann–Whitney *U*-tests to compare differences in microhabitat structure and composition between 2002 and 2013 because all variables were not normally distributed. Then we generated a correlation matrix between the explanatory variables of microhabitat structure and composition. To identify statistically significant microhabitat changes, we carried out binomial logistic regressions with past and current (0 = past, 1 = current) as the categorical dependent variables. Highly correlated variables ($|r| > 0.6$) with low R-square (R^2_N) in univariate models were eliminated in order to reduce multicollinearity (Nagelkerke 1991). The best subset of the models (Hosmer et al. 1989) was selected from all possible combinations using information-theoretic criteria to evaluate the relative effect of the various habitat variables. We ranked all models according to the second-order Akaike's information criterion (AIC_c), ΔAICc (the difference in AIC_c between each candidate model and the model with the lowest AIC_c), and Akaike weights (W_i), which provides better assessment from small sample sizes (Burnham and Anderson 2002, Wang et al. 2012). Finally, we assessed the relative importance of each variable by the sum of model weights containing this variable (Burnham and Anderson 2002). All statistical analyses were performed with STATISTICA ver. 10 (StatSoft, <www.statsoft.com/>); values are presented as mean ± SE, with statistical significance determined at the $\alpha = 0.05$ level, unless otherwise indicated.

Results

Reforestation

Overall accuracy of the 2013 land cover map was 85.5%, with kappa statistics greater than 0.80. This is an acceptable classification procedure for further analysis (Congalton 1991, Thomlinson et al. 1999). Land use/land cover change analysis revealed that the coniferous forest area increased by 214.3 ha (3.1%) in the core zones and by 1550.3 ha (11.8%) in the non-core zone, whereas coniferous forest decreased by 857.1 ha (5.7%) in the surrounding landscape outside the reserve. The broadleaf forest area decreased slightly in the core zones by 81.6 ha (1.5%) as well as by 1505.9 ha (18.7%) in the non-core zone, and increased by 2197.9 ha (16.1%) in the surrounding landscape (Table 2). As a result, both DNNR and the surrounding landscape have a slight net growth of forest areas from 2002 to 2013 (Table 3).

Between 2002 and 2013, trends of reforestation and deforestation have varied with location (Fig. 2). The land

Table 1. Class- and landscape-level metrics for habitat fragmentation pattern comparison between 2002 and 2013.

Metric	Level	Description
Number of patches (NP)	Class	Total number of coniferous patches, a homogeneous and continuous coniferous area which provided the most important habitat for Reeves's pheasant.
Largest patch index (LPI)	Class	Percentage of conifer forest type covered by largest patch.
Edge density (ED)	Class	Sum of length of all conifer forest edge segments, divided by total area for conifer forest class.
Mean patch size (MPS)	Class	Measures average patch size or area for the coniferous type.
Patch cohesion index (PCI)	Class	Measures the physical connectedness of the corresponding coniferous patches.
Mean shape index (MSI)	Landscape	Measures average complexity for a category of patch shape, compared to a square patch of identical area.
Nearest neighbor distance (NND)	Landscape	Distance (m) to the nearest neighboring patch of the same type, based on shortest edge-to-edge distance.
Contagion index (CONTAG)	Landscape	Measures the extent to which landscape elements are aggregated or clumped.
Shannon's diversity index (SHDI)	Landscape	Measures the diversity of patch types: equals 0 when the landscape contains only 1 patch and increases as the number of different patch types increases.

Table 2. Extent of land cover area and changes of Dongzhai Nature Reserve, for the core zones, non-core zone, inside the nature reserve and surrounding landscape, assessed using Landsat 5 TM and Landsat 8 OLI satellite imagery from 2002 and 2013.

Land cover	2002 (ha)	2013 (ha)	2002–2013 change (ha)	Change rate (%)
Core zones				
Coniferous	7010.4	7224.7	214.3	3.1%
Broadleaf forest	5507.5	5425.8	–81.7	–1.5%
Cultivated fields	1478.2	1361.4	–116.8	–7.9%
Wetland	1.4	14.4	13.0	900.0%
Water bodies	43.2	26.0	–17.2	–39.8%
Artificial surfaces	29.2	14.0	–15.2	–51.9%
Bare land	0.6	4.1	3.5	542.9%
Non-core zone				
Coniferous	13194.2	14744.5	1550.3	11.8%
Broadleaf forest	8057.2	6551.3	–1505.9	–18.7%
Cultivated fields	12275.6	12852.1	576.5	4.7%
Wetland	116.2	116.4	0.2	0.2%
Water bodies	983.0	336.4	–646.6	–65.8%
Artificial surfaces	545.1	496.6	–48.5	–8.9%
Bare land	74.1	148.1	74.0	99.9%
Surrounding landscape				
Coniferous	15111.9	14254.8	–857.1	–5.7%
Broadleaf forest	13667.3	15865.2	2197.9	16.1%
Cultivated fields	33796.0	33282.7	–513.3	–1.5%
Wetland	441.5	475.3	33.8	7.7%
Water bodies	2813.0	956.0	–1857.0	–66.0%
Artificial surfaces	1085.2	1489.1	403.9	37.2%
Bare land	318.1	909.7	591.7	186.0%

cover within the core zones underwent subtle changes, with less than 10% of the total area undergoing change (reforestation plus deforestation), and over 80% of the entire area was relatively stable. In contrast, the non-core zone and the surrounding landscape forest coverage were much less stable, with approximately 14% of the area experiencing change (Table 3).

Independent samples *t*-tests or Mann–Whitney *U*-tests of habitat fragmentation metrics indicated that there were no significant differences in the habitat composition pattern between 2002 and 2013 (All $p > 0.05$). Therefore, after 11 years of strict nature reserve management, the habitat pattern did not change significantly at the macrohabitat scale.

Microhabitat structure and composition changes

Microhabitat structure and composition differed significantly between 2002 and 2013 (Mann–Whitney *U*-tests, most $p < 0.01$, Table 4). In the tree stand layer, average tree height in 2013 was greater than that in 2002. Further, canopy cover in 2013 was denser than that in 2002 and the average DBH in the 2013 plots was also greater than that in 2002 (Table 4). In the shrub layer, average density and cover of shrubs was greater in 2013 than in 2002. In contrast, average height and cover of herbaceous plants in the ground layer was less in 2013 than in 2002. The depth of the leaf-litter in 2013 was thicker than in 2002 (Table 4).

A correlation matrix (Table 5) shows that we found relatively high correlations ($|r| > 0.6$) between tree height and DBH and canopy cover, between shrub density and shrub cover, and between grass height and grass cover, but also with leaf-litter depth. Using R-square (R^2_N) by Nagelkerke (1991) in univariate models, we selected canopy cover, shrub density and grass height as variables for optimum subset model building. Eight models were generated and seven were statistically significant ($p < 0.001$), the null model was the exception. Two models with $\Delta AIC_c < 3$ were selected as the candidate models (Table 6). The first one was the global model containing all three variables and the second model had canopy cover and shrub density. A Wald statistical test indicated canopy cover, shrub density and grass height had a significant effect on the detection of differences in the microhabitat between these two periods.

Discussion

Multi-scale analyses can enhance our understanding of changes in the distribution and habitat of bird species (Coreau and Martin 2007). We selected regional, macrohabitat and microhabitat scales to quantify changes in land use/land cover, habitat fragmentation, and microhabitat structure and composition under current nature reserve management practices in China. Forest coverage slightly increased both inside the DNNR and within the surrounding landscape after establishment of the nature reserve in 2001, whereas the habitat fragmentation metrics did not change significantly at the home range scale after forest rehabilitation. In particular, the secondary shrubs provided very dense cover at this early-successional stage of forest rehabilitation (Bruelheide et al. 2011, Yang et al. 2015). Our results indicated that this type of vegetation management in the non-core zone and surrounding landscape does not lead to an immediate expansion of suitable habitat for Reeves's Pheasant, likely because of this pheasant prefers habitat with tall trees and sparse undergrowth (Wu et al. 1991, 1994, Xu et al. 2007). Moreover, the land cover of the core zones of the reserve was more stable than the non-core zone and surrounding areas. With strict legislation and regulations imposed within the nature reserve aimed at reducing poaching and logging, the population of Reeves's pheasant was expected to

Table 3. Extent of area occupied by land cover changes categories of Dongzhai Nature Reserve, for the area within the core zones, non-core zone and the surrounding landscape, assessed by Landsat 5 TM and Landsat 8 OLI satellite imagery classification results between 2002 and 2013.

Change category name	Core zones		Non-core zone		Surrounding landscape	
	Area(ha)	Rate	Area(ha)	Rate	Area(ha)	Rate
Stable forest	11479.1	81.6%	16260.7	46.1%	18784.4	27.9%
Reforestation	585.7	4.2%	2517.6	7.1%	5667.8	8.4%
Stable non-forest	1552.6	11.0%	13994.0	39.7%	38453.7	57.2%
Deforestation	453.1	3.2%	2473.1	7.0%	4327.0	6.4%

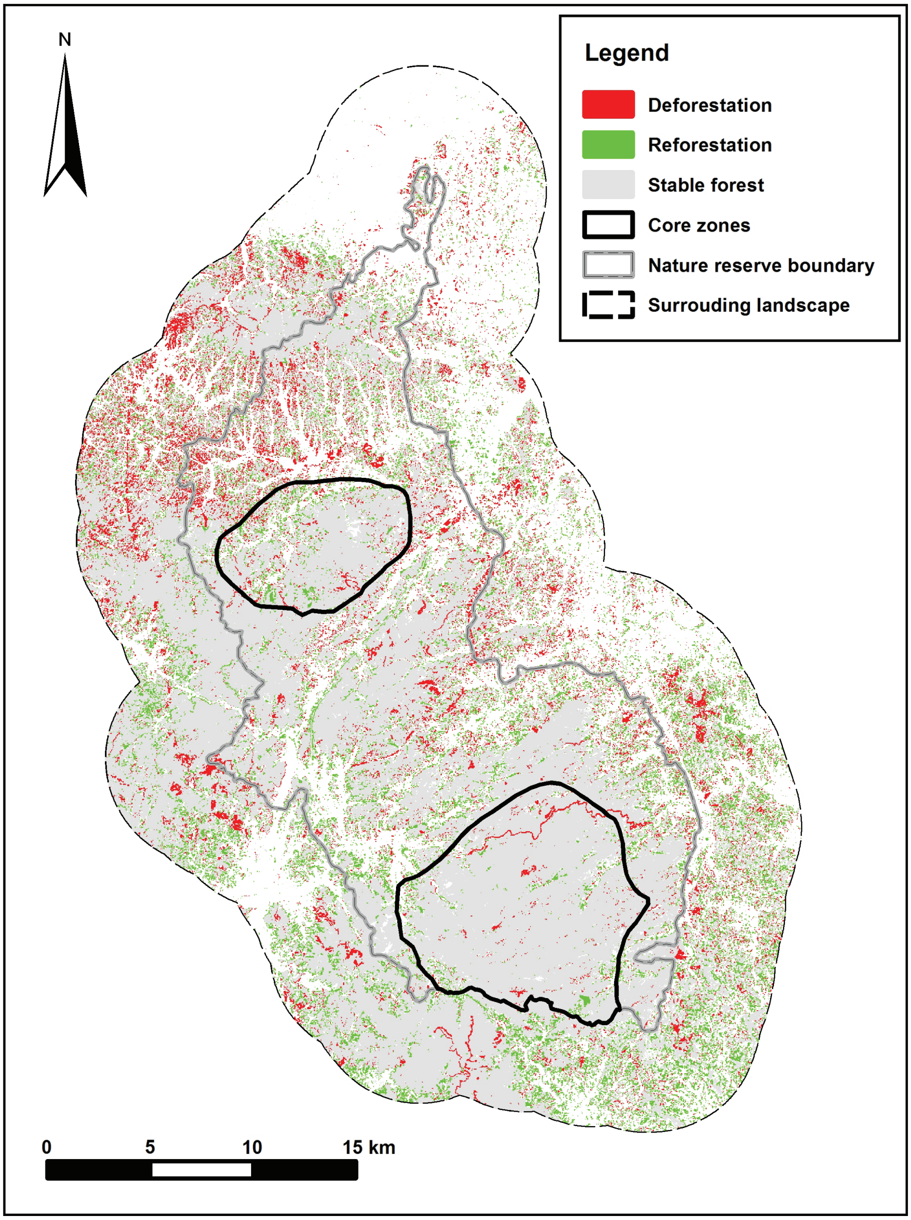

Figure 2. Land use/land cover change classes for the Dongzhai Nature Reserve core zone, non-core zone and the surrounding landscape.

Table 4. Comparison of differences in forest structure (mean ± SE) of the core zone between 2002 and 2013 by Mann–Whitney *U*-tests.

Variables	2002 (n = 25)	2013 (n = 25)	z-value	p-value
Tree diversity	3.00 ± 0.28	2.44 ± 0.27	1.30	NS
Tree density	10.00 ± 1.27	10.08 ± 1.50	0.27	NS
DBH[a](cm)	11.96 ± 1.04	19.43 ± 1.37	– 3.91	< 0.01
Tree height (m)	7.96 ± 0.60	9.38 ± 0.74	– 2.19	0.02
Canopy cover (%)	26.56 ± 3.35	60.40 ± 4.92	– 4.59	< 0.01
Shrub diversity	7.28 ± 0.51	7.68 ± 0.44	– 0.69	NS
Shrub denstiy (%)	48.40 ± 4.53	88.48 ± 11.77	– 3.42	< 0.01
Shrub height (m)	1.89 ± 0.21	2.25 ± 0.13	– 1.91	NS
Shrub cover (%)	34.72 ± 3.60	43.80 ± 2.73	– 2.32	0.02
Grass height (cm)	44.49 ± 5.17	13.43 ± 1.43	4.79	< 0.01
Grass cover (%)	52.72 ± 4.90	29.27 ± 4.25	3.34	< 0.01
Leaf-litter cover (%)	67.12 ± 4.86	80.24 ± 1.89	–1.85	NS
Leaf-litter depth (cm)	1.11 ± 0.12	3.62 ± 0.28	– 6.05	< 0.01

Notes: 1. Abbreviations for variables: [a]DBH, diameter at breast height.
2. n - sample size; NS - not significant.

Table 5. Spearman's correlation coefficients among eight forest structural variables used in modeling key factors of change in the habitat of the Reeves's pheasant.

	DBH[a]	TH	TC	SD	SC	GH	HC
TH	0.61[b]						
TC	0.82[b]	0.60[b]					
SD	0.31	0.16	0.26				
SC	0.26	0.10	0.12	0.63[b]			
GH	–0.62[b]	–0.31	–0.68[b]	–0.35	–0.32		
HC	–0.48	–0.19	–0.54	–0.24	–0.31	0.75[b]	
LD	0.72[b]	0.38	0.70[b]	0.45	0.27	–0.64[b]	–0.51

[a]Abbreviations for variables: DBH, diameter at breast height; TH, tree height; TC, tree canopy cover; SD, shrub density; SC, shrub cover; GH, grass height; HC, herb or grass cover; LD, leaf-litter depth.
[b]High correlation between variables with $|r| > 0.6$.

increase or stabilize in the core zones. However, according to our annual survey, the Reeves's pheasant population density declined from 30.5 individual birds per km^2 in 2002 to 11.0 in 2012 in the core zones in DNNR despite the creation of the reserve and legislation (Zhou et al. 2015). Therefore, the macrohabitat change analysis cannot explain this species population decline in the core zones.

Threatened species are vulnerable to changes in localized habitats that can have a significant impact on their overall population (McCulloch and Norris 2001). Recently designated or expanded nature reserves created from forest farms have led to a reduction in forest management (Xu and Melick 2007). This has resulted in secondary succession and forests have recovered to relatively undisturbed states with variable forest structures developing over time (He et al. 2002). Similarly, the reduction in logging and other direct human disturbance after the DNNR establishment in 2001 has led to significant microhabitat structure and composition changes. As a result, in the core areas, the forest canopy and shrub layer grew denser, while the herbaceous plants in the ground layer became sparser. Forest canopy is one of the major determinants of microhabitat in forest ecosystems (Jennings et al. 1999). Tall trees with a dense canopy cover provide good roost-sites for Reeves's pheasants to avoid predators (Wu et al. 1991, Xu et al. 1991), but do not provide sufficient light and warmth for the development of a field-layer and associated invertebrate communities (Jennings et al. 1999). The degradation of invertebrate communities and the increase in leaf-litter depth requires the pheasant to expend greater energy on foraging, hence reducing survival because of an increased chance of being discovered by predators (Xu et al. 2002). Moreover, denser shrub cover also impedes pheasant movement due to its relatively large body size and extremely long tail (Johnsgard 1999). Grass cover is also an important habitat component for galliformes (Simonetti 1989, Parrott 2015), thus the deterioration of grass cover increases the success of raptor hunting, reduces herb seed availability, and decreases nest success (Xu et al. 2002). These subtle changes in microhabitat structure and composition in the core zone could possibly explain the decline in Reeves's pheasant populations.

Bird species change with forestry practices and management that alter habitat structure (DeGraaf et al. 1998). However, protected areas are often treated as a single conservation management area despite the variety of habitat, reasons for their establishment, and objectives/criteria for their success (Chape et al. 2005, Geldmann et al. 2013). Current management of nature reserves restricts entrance and prohibits any extractive activities in the core and buffer zones (State Council 1994, Xu et al. 2012), resulting in secondary succession. As a result, understory shrubs grow rapidly, producing dense shrub thickets, impeding pheasant understory activity (Johnsgard 1999). Based on our survey from 2011–2012, Reeves's pheasant were extirpated from some nature reserves in China under current regulations (Zhou et al. 2015). Therefore, the current regulations for nature reserve management may not be suitable for maintaining the survival of this galliform species.

Although a legislative framework for nature reserve management has been established in China, some shortcomings exist in the legislation and enforcement (Xu et al. 2012). In particular, the rules in the Regulations of Nature Reserves in China are often too strict and inflexible with respect to nature reserve management (Xie et al. 2004). Once nature reserves are established, the exclusion of human activities in core and buffer zones often exacerbates the conflict between natural resource acquisition by local communities and nature conservation (Xu et al. 2012). In fact, many nature reserves are not pristine and have long been affected by humans and described as sites of cultural heritage (Xu and Melick 2007).

Our work highlights the important role that local forest management can play in the conservation of some threatened species in nature reserves. Without human intervention, natural succession changes the microhabitat structure and composition significantly and can affect the habitat suitability for some threatened species (Connelly et al. 2000). For instance, as a vulnerable species in need of protection, the Reeves's pheasant requires specific management interventions to ensure its continued survival (Zhou et al. 2015). Thus, it is necessary to legislate consistent and operational regulations and to modify the habitat conservation regulations of nature reserves for some threatened species. In particular, allowing the local people to thin understory shrub could be beneficial to increase the Reeves's pheasant habitat suitability in the short term.

Table 6. Model selection for identifying microhabitat changes between 2002 and 2013 in the core area of Dongzhai National Nature Reserve. Models were ranked according to ΔAIC_c, -2log(l) referred to -2log-likelihood. All models used n = 50 sampling plots.

ID	Predictors	-2log(l)	K	AIC_c	ΔAIC_c	W_i
1	Canopy cover + shrub density + grass height	23.36	4	32.25	0.00	0.74
2	Canopy cover + shrub density	28.62	3	35.14	2.89	0.17
3	Shrub density + grass height	30.48	3	37.00	4.75	0.07
4	Canopy cover + grass height	34.02	3	40.54	8.29	0.01

In addition, assessing management effectiveness of nature reserves depends largely on the monitoring that is undertaken. Current monitoring activities mainly focus on threatened species populations and habitat quantity, but often overlook habitat quality which is evaluated by microhabitat structure and composition (Ma et al. 2009). If the current spatial bias in monitoring is not resolved, then inferring future extinctions will become even more problematic (Boakes et al. 2016). Therefore, a system of multi-scale, long-term habitat monitoring and evaluation should be established, and the result should be used to modify management plans and policies.

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