Can Rural Tourism Policy and Development Influence Local Bird Diversity? A Quasi-Field Experiment

Bo Hu [a]; Yixin Wang [a]; Jiaci Lin [a]; Liuna Geng [a]

[a] School of Social and Behavioral Sciences, Nanjing University, PR China

Correspondence concerning this article should be addressed to: Liuna Geng

School of Social and Behavioral Sciences, Nanjing University, Nanjing, PR China

gengliuna@nju.edu.cn.

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Abstract

Rural tourism is increasingly promoted as a strategic tool for economic revitalization and rural sustainability, yet its compatibility with biodiversity conservation remains a subject of debate. This study investigates the causal impact of China's National Key Rural Tourism Villages policy on local bird diversity by employing a quasi-experimental design. Using a nationwide county-year panel constructed from citizen-science bird observation records, we find that the policy designation leads to a statistically significant decline in bird species richness. Event-study estimates show an immediate and front-loaded decline, a pattern consistent with early infrastructure construction and intensified human disturbance. The policy is associated with the expansion of accommodation capacity, and high-end hotel intensity is in turn negatively associated with predicted bird richness. Heterogeneity analyses further indicate that negative impacts are concentrated in ecologically sensitive areas, specifically forests, wetlands, and wild plant habitats, and in resource-constrained low-GDP counties. The study suggests that to achieve a synergy between development and conservation, tourism accreditation systems should integrate mandatory ecological planning, spatial zoning, and long-term biodiversity monitoring.

Keyword: Rural tourism; Biodiversity conservation; Bird Biodiversity; Ecological disturbance

Introduction

Biodiversity is the basis of human existence and development, but global wildlife populations are declining (Doley & Barman, 2023). Because diversity reflects the health of an ecosystem, maintaining biodiversity, and specifically bird diversity, is an important part of sustainable development (Sachs et al., 2019; C. Sekercioglu, 2006). China has made biodiversity protection a top priority with policies and action plans such as the China Biodiversity Conservation Strategy and Action Plan being developed, as well as ecological goals being incorporated into its longer-term vision. While urban biodiversity has received much attention, the rural landscape remains as critical refuge because it typically contains larger and more contiguous semi-natural habitats than peri-urban areas (Daily et al., 2001; Tscharntke et al., 2005).

Rural tourism is an important measure for promoting rural economic development. China's rural tourism industry has expanded rapidly under the national rural revitalization strategy (Gao & Wu, 2017; Y.-L. Liu et al., 2023; Su, 2011). Recent evidence also indicates that destination development can reshape rural character through changes in land use, livelihoods, and the built environment (Hong et al., 2024). Rural tourism key villages refer to rural communities that use their natural and cultural landscape resources to offer leisure, vacation, and agricultural-experience activities (Zou et al., 2023). These villages are selected using a systematic set of criteria to ensure representativeness in ecological, cultural, and industrial terms (Ministry of Culture and Tourism, 2022).

More recently, the Ministry of Culture and Tourism has implemented since 2019 the National Key Rural Tourism Villages, in which it recognizes selected villages each year as national key rural tourism villages. In four batches from 2019–2022, a total of 1,399 key villages were designated nationwide (Ministry of Culture and Tourism, 2022). The rapid influx of tourists into rural areas poses a double-edged challenge for ecological systems, especially bird communities. Excessive or uncontrolled tourism development may disrupt bird habitats and behavior (Steven & Castley, 2013). Increased human presence is typically associated with noise, light pollution, and direct disturbance, which may alter avian behavior and physiology (Pérez-González et al., 2024). Research has shown that human disturbance may increase birds' stress, reduce breeding success, and in some cases, force sensitive species away from suitable habitat (Bötsch et al., 2017; Moroni et al., 2017; Tablado & Jenni, 2017). This means that areas with high tourist activity could suffer declines in the abundance and richness of birds. Rural tourism, if it involves intensive land-use change, will fragment habitats and remove the vegetation cover that birds rely on for foraging and nesting (Gössling, 2002; Steven et al., 2011). Ecological effects depend on the scale and form of tourism and on visitor behavior. Infrastructure-intensive, mass-visitation models are likely to be more disruptive than low-density, interpretation-led ecotourism, while off-trail intrusion and the use of artificial light and sound can amplify disturbance (Pickering & Hill, 2007; Steven et al., 2011). Because recreational use often concentrates along particular routes and during peak periods, these pressures can also be spatially and temporally uneven (Costa et al., 2024).

These general pathways are refined by recreation-ecology evidence, which shows that disturbance responses vary among species and contexts. Breeding status, habitat, disturbance frequency, and the time available to compensate for lost foraging can determine whether repeated encounters produce cumulative costs (Dertien et al., 2021; Van Der Kolk et al., 2021). Quantitative thresholds are often expressed as distances from people or recreational trails, but their substantial variation cautions against applying a single universal setback distance (Dertien et al., 2021). Visitor conduct is also consequential: close approaches, off-trail movement, group size, and noise can produce different avoidance responses, including in birdwatching and nature-tourism settings (Aas et al., 2023; Avenzora et al., 2024). The ecological effects of rural tourism depend not only on infrastructure intensity but also on when, where, and how visitors use the landscape.

On the other hand, rural tourism may produce beneficial effects for bird conservation and biodiversity (R. C. Buckley et al., 2016). When tourism development prioritizes ecological values, it can provide financial resources and incentives for habitat protection (Ferraro & Hanauer, 2011; Schwoerer & Dawson, 2022). Income from tourism can be reinvested in environmental restoration, reserve management, or conservation education programs (Echeverri et al., 2022; Huynh et al., 2025). The presence of visitors interested in nature can spur local communities and authorities to value their natural heritage and maintain a cleaner, greener environment (Echeverri et al., 2022; Huynh et al., 2025). Where villages are involved in tourism, farmers and residents preserve more semi-natural landscapes (Clausen et al., 2024). Such semi-natural habitats also sustain higher bird diversity. In a South American study, Weyland and colleagues (2021) found that farmer participants in rural tourism showed a greater willingness to conserve uncultivated areas because they perceived the intrinsic and economic value of a flourishing natural environment. Rural tourism development is often accompanied by government-led infrastructure improvements, which can benefit the local ecosystem. When managed as ecotourism, tourism can raise public awareness and appreciation for biodiversity so that visitors and residents become advocates for conservation (Powell & Ham, 2008; Stronza & Pêgas, 2008). With proper planning and community participation, rural tourism could enhance bird diversity by unleashing a virtuous cycle of conservation and development.

Accordingly, tourism effects should not be generalized as uniformly negative or positive. Their direction and magnitude depend on habitat sensitivity, visitor intensity, temporal concentration, visitor behavior, and the quality of management. Recent evidence in Current Issues in Tourism is consistent with this context-dependent view: rural tourism development can alter the rural character and land-use functions of destinations (Hong et al., 2024), while mobile-app data show that recreational use of protected areas is strongly concentrated across routes, places, and times, creating opportunities for targeted zoning and monitoring (Costa et al., 2024). Behavioral interventions can also be tailored to the determinants of pro-environmental conduct rather than relying only on generic awareness messages (Povilionis et al., 2025). The present study therefore estimates the average effect of a specific Chinese accreditation-and-development pathway, not an invariant effect of all rural tourism.

Given these contrasting potential impacts, the question remains open whether the net effect of rural tourism development on biodiversity is beneficial or not in practice. Many existing studies on rural tourism focus on economic or social outcomes (He et al., 2021). The advantages of rural tourism for local entrepreneurship, employment, and economic growth are well known (Wei et al., 2024). Tourism can alter the habitat, behavior and physiology of wild animals, and change their spatial distribution and abundance (Jęczmyk et al., 2021). In the field of bird diversity, current studies have mainly focused on certain regional scales rather than a nationwide perspective. However, the distribution of birds in China is characterized by significant geographical diversity and ecological adaptability, driven by natural conditions and migration routes alike (T.-S. Ding et al., 2006). The complexity makes a nationwide assessment particularly informative. Meanwhile, conservation research frequently studies protected areas or habitat management in isolation (J. Liu et al., 2007). A need exists to connect these domains and examine how a policy seeking rural economic revitalization affects ecological indicators such as bird species diversity. Understanding this relationship is important for achieving win-win outcomes under the rural sustainability agenda: rural development policies should synergize with biodiversity conservation rather than work at cross-purposes (Y. Liu & Li, 2017).

This study aims to assess whether the establishment of rural tourism key villages in China has led to changes in local bird diversity, using a quasi-natural experimental design. We use the staggered implementation of the National Key Rural Tourism Villages (2019-2022) as an intervention and examine bird diversity effects before and after, compared to suitable control observations. We move beyond estimation of the policy effect toward a mechanism interpretation by linking the designation to the accommodation-capacity measure using hotel data. We are then able to ask which accommodation-expansion pathway is most consistent with the data. From the perspective of policy evaluation, the evaluation of the National Key Rural Tourism Villages policy can not only identify the implementation problems but also offer information for the subsequent policy optimization, as well as provide evidence to guide the rural tourism development in different regions to avoid blind replication. It also can serve to guide the design of other industrial policies under rural revitalization, facilitate efficient policy-resource allocation, and promote sustainable rural development and urban-rural integration.

Local governments follow central guidelines to plan rural tourism, but because the selection criteria are standardized and objective, the ability of local officials to gerrymander which villages are listed is limited (C. Liu et al., 2020). This study presents quasi-experimental evidence of the relationship between a rural tourism accreditation policy and biodiversity outcomes, supplementing cross-sectional, case-based, and anecdotal evidence. We also examine the relationship between a rural development program and ecological protection objectives. The study indicates whether the rural tourism program enhances or detracts from biodiversity protection (positive synergy or negative trade-off), in the context of attempting to integrate multi-dimensional policy goals. Although birds are the focal ecological indicator in this study, the proposed transmission channels are not just for birds, having broader implications for all biota and for other certified destination schemes.

Methodology

Data Sources

We assemble an unbalanced county–year panel from 2015 to 2024 to evaluate the relationship between China's National Key Rural Tourism Villages policy and bird diversity. Bird observations come from China's citizen-science bird checklist platform (www.birdreport.cn). It is the most extensive online database of bird observation records in China (Y. Ding et al., 2022). Each checklist records the submission date, geographic coordinates, observation start time and duration, the number of observers, and the species detected together with reported counts. The platform contains information on both species presence and reported abundance. However, abundance reporting follows the platform's free-form data-entry convention, and reported counts are not subject to mandatory observer-level certification. We treat the data as primarily presence- and richness-oriented and use the reported abundances only as an internal consistency check.

To create a consistent measure of local biodiversity, we aggregated individual bird checklists (date, geographic location, and species counts) to the county-year level. Due to variable observer effort, we standardize the species richness metric on the checklist level before aggregation. We specifically regress the natural logarithm of species count against the logarithm of observation time, controlling for hour-of-day fixed effects to account for diurnal patterns in bird activity. The fitted values are exponentiated to give predicted species richness that is partially adjusted for effort patterns. We use the annual mean of the predicted richness as our main outcome variable, as well as the annual mean of raw observed richness and the annual maximum of predicted richness to capture potential effects on rare or episodically detected species. To additionally address potential concerns of uneven observer participation that effort standardization at the checklist level cannot fully accommodate, we also create county-year measures of sampling effort: the natural logarithm of the number of checklists submitted, the number of unique observers, the number of distinct sampling locations, the number of months covered, and the mean observation time. The controls are entered as additional controls in the robustness checks and pre-policy baseline interacted with year fixed effects. These adjustments reduce observable effort differences but cannot fully eliminate species-specific detectability, variation in observer expertise and reporting propensity, preferential sampling of accessible or attractive locations, or changes in the composition of observers over time (Backstrom et al., 2025; Hochachka et al., 2023; Robinson et al., 2021; Scher & Clark, 2023).

The treatment variable is the staggered adoption of the National Key Rural Tourism Villages designated by the Ministry of Culture and Tourism (total of 1,399 villages, and 1,036 villages matched with bird observation records). For each county, we calculate an average location by taking the median of geographic coordinates from bird checklists submitted in 2015 or later. Each county is then assigned its nearest National Key Rural Tourism Village based on straight-line distance. The timing of the policy intervention is the year in which the nearest village was officially marked. We use this timeline to design a treatment that determines whether a particular year is on or after the designation year for a county. A panel feature of the model that influences the choice of the comparison group is that all counties in our analysis sample are eventually exposed to a designated village; that is, the panel contains no strictly never-treated county. This has less consequence for the static specification, but for the cohort-based event study, we use the last-treated cohort as the control cohort (Sun & Abraham, 2021). To control for regional socio-economic confounders, we combine county-level data on Gross Domestic Product (GDP) and cultivated land area from statistical yearbooks matched to bird observation centroids by nearest-neighbor interpolation. These variables are fixed at their base level and interact with year fixed effects.

To facilitate the mechanism analysis, we also summarize a deduplicated hotel Point of Interest (POI) corpus (according to Amap data). The source records each business by registered category. For each accommodation type (high-end hotels, apartment hotels, and homestays), we compute the county-year count and its log-transform and the pre-policy baseline mean. Because these accommodation variables are themselves determined in response to policy, we classify them as policy-induced intermediate channels rather than as exogenous regressors, and the corresponding estimates are presented as descriptive decompositions rather than as causal mediation effects.

Using village-level designations to characterize county-level exposure introduces a coarsening of the spatial unit. We address this through three complementary checks: distance-threshold restrictions and a distance interaction; Queen-contiguity spatial diagnostics and spatial-lag robustness models; and replication of the sampling effort and distance robustness on the two alternative outcomes.

Empirical Strategy

The two-way fixed effects difference-in-differences (TWFE-DID) method can be used to estimate the average treatment effect (ATT). Our data are structured around the core conditions for staggered-adoption DID: a panel with repeated observations on the same counties; asynchronous treatment timing determined by the nearest policy site’s year; meaningful untreated or not-yet-treated comparison years for each treated county; and rich fixed effects that purge confounders that are time-invariant (county FE) or common across space (year FE). The Sun–Abraham estimator addresses treatment-effect heterogeneity across cohorts/event times, which is likely here because counties join in different calendar years and tourism development may diffuse at different speeds. Considering that interventions arrive in batches and dynamic effects may vary with relative intervention time, we employ Sun–Abraham's event study in addition to the baseline TWFE to achieve robust estimates of batch-specific heterogeneous dynamics and explicit tests of pre-intervention trends. Because the panel contains no strictly never-treated county, we use the last-treated cohort as the control cohort in the Sun–Abraham event study and assess the sensitivity of this choice by re-estimating the dynamic specification after dropping the last cohort from the panel.

Our baseline specification is a staggered-adoption TWFE regression:

$$Y_{ct} = \beta Post_{ct} + \mu_c + \lambda_t + X_c' \cdot \theta_t + \epsilon_{ct}$$

where Y_{ct} denotes annual mean predicted richness, annual mean observed richness, or annual maximum predicted richness. Post_{ct} equals one for $t \geq G_c$ with G_c = first treat year. We include county fixed effects μ_c to absorb time-invariant differences in habitat, and year fixed effects λ_t to capture nationwide shocks. β is the ATT under the parallel-trends assumption: absent the policy, outcomes for treated and not-yet-treated counties would have moved in parallel after conditioning on fixed effects and control-year interactions.

A key identifying assumption of the TWFE model is parallel trends: in the absence of the tourism policy, the bird diversity trends in treated and control counties would have followed parallel paths. To rigorously test this assumption and explore dynamic treatment effects, we employ the interaction-weighted estimator (Sun & Abraham, 2021). This method is robust to potential heterogeneity in treatment effects across different adoption cohorts, which can bias static TWFE estimates. We estimate the following event-study specification:

$$Y_{ct} = \sum_{k \neq -1} \beta_k \cdot 1\{t - G_c = k\} + \mu_c + \lambda_t + \boldsymbol{X}_c' \cdot \theta_t + \epsilon_{ct}$$

where k indexes event time (years relative to first treatment), and $k = -1$ is the omitted baseline. This design produces cohort-robust β_k and a transparent pre-trend diagnostic $\beta_k: k < 0$ should be near zero and jointly insignificant if parallel trends hold.

Results

Baseline Estimates

Table 1 reports the baseline estimations for the impact of key rural tourism villages policy on bird diversity. We treat the predicted-richness specification as the primary outcome and the other two as supporting evidence, and we report sampling effort and distance robustness on all three outcomes to show that the same direction is preserved on the alternative metrics. The coefficient on the post-treatment indicator is consistently negative across specifications. For the annual mean of predicted richness, the estimated coefficient is –0.610 (p =.014). This result suggests that the designation of National Key Rural Tourism Villages leads to a significant decline in local bird species richness. Using raw observed richness yields a negative coefficient (β = –0.300; p = .445) that aligns in direction with the main result. For annual maximum predicted richness, the coefficient is larger in magnitude (β = –0.957, p = .085, approached significance). The three outcomes thus provide a consistent directional pattern; the precision differences reflect the noisier nature of raw counts (which absorb effort variation more fully) and the noisier upper-tail behavior of the annual maximum. Given that statistical precision varies across the three richness metrics employed, the strength of this conclusion rests primarily on the predicted-richness specification, and the overall pattern should be interpreted with appropriate caution.

Table 1. The Impact of Rural Tourism Villages Policy

	(1)	**(2)**	**(3)**
Rural Tourism Villages	-0.610**	-0.300	-0.957*
	(0.248)	(0.393)	(0.555)
Outcome	Predicted richness	Observed richness	Predicted richness (Max)
Observations	14,635	14,635	14,635
County FE	Yes	Yes	Yes
Year FE	Yes	Yes	Yes
Controls × Year FE	Yes	Yes	Yes
Adj. R-squared	0.165	0.297	0.275

Note: * p < 0.1, ** p < 0.05, *** p < 0.01; Standard errors clustered at the county level in parentheses.

Dynamic estimates from the Sun–Abraham event study (Figure 1) show that pre-treatment coefficients are small, fluctuate around zero, and are jointly statistically indistinguishable from zero, while post-designation coefficients drop immediately at $(k \geq 0)$ and remain negative. The immediacy of the decline points to front-loaded disturbances, likely stemming from infrastructure construction and increased human activity during the initial phase of tourism development. Pre-treatment balance, robustness, alternative-outcome, and spatial analyses are reported in the Supplementary Materials (Tables S1–S9; Figures S1–S2).

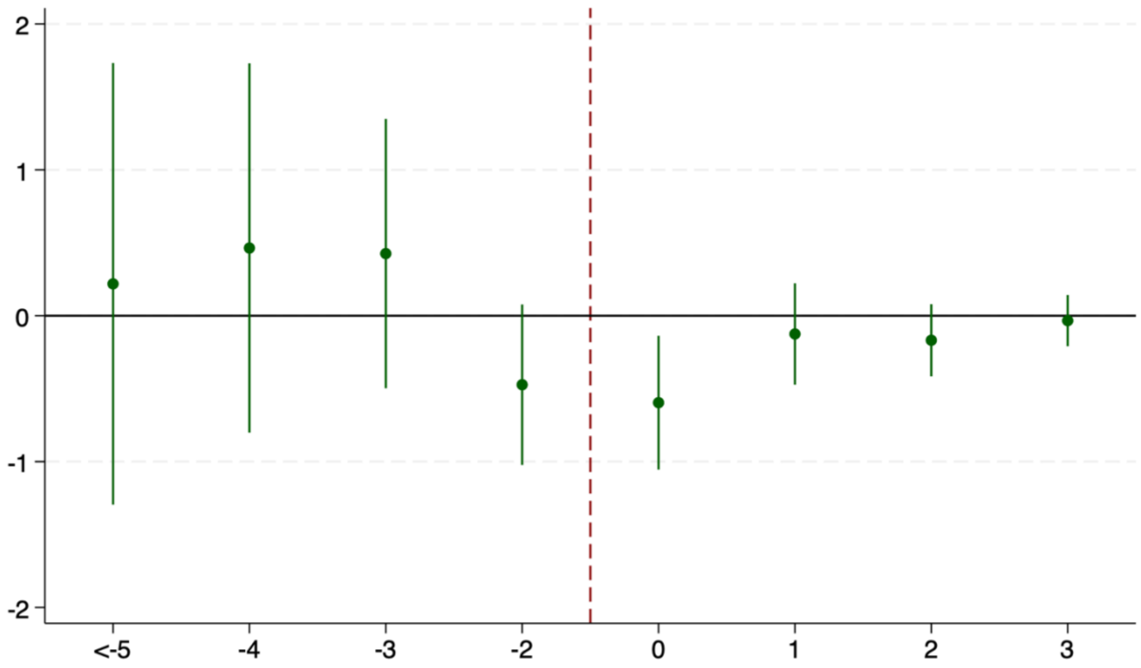

Figure 1. Event-study Estimates of the Impact of Rural Tourism Policy on Bird Diversity.

Mechanism Analysis

The policy indicator does not directly measure tourism intensity. To move toward a mechanism interpretation, we link the designation to fine-grained hotel POI data and decompose accommodation expansion. The analysis has two parts: (1) the first-stage association between the policy and each accommodation type, and (2) the conditional association between each accommodation type and bird richness (Table 2). In the first stage, the policy is associated with a weak expansion of total accommodation capacity ($\beta = 0.086$, $p = .061$), mainly through apartment hotels ($\beta = 0.064$, $p = .020$) and homestays ($\beta = 0.059$, $p = .050$). The high-star hotel estimate is positive but statistically imprecise ($\beta = 0.032$, $p = .219$). In the second stage, high-star hotel intensity is the accommodation channel most strongly associated with a decline in bird richness ($\beta = -0.181$, $p = .048$); total hotel intensity is also negatively associated ($\beta = -0.077$, $p = .091$). Apartment hotels and homestays are not significantly associated with bird richness in this specification (apartment hotels: $\beta = 0.108$, $p = .117$; homestays: $\beta = -0.001$, $p = .984$). The two stages imply that the policy mainly expands apartment hotels, homestays, and total accommodation capacity, whereas the ecological cost is most clearly related to the high-star segment. Because accommodation outcomes are themselves jointly determined with the policy, the conditional associations between hotel intensities and predicted richness should not be interpreted as formal mediation effects. We present the estimates as a descriptive consistency check connecting policy, plausible operational channels, and the bird-richness outcome.

Table 2. Hotel POI Mechanism Paths.

	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Policy	0.086*	0.064**	0.032	0.059*				
	(0.046)	(0.027)	(0.026)	(0.030)				
All hotels					-0.077*			
					(0.046)			
Apartment						0.108		
						(0.069)		
High-star hotels							-0.181**	
							(0.092)	
Homestays								-0.001
								(0.071)
Observations	12,001	12,001	12,001	12,001	12,001	12,001	12,001	12,001
Adj. R-squared	0.657	0.842	0.861	0.771	0.157	0.157	0.157	0.156
County FE	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Year FE	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Controls × Year FE	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes

Note: Columns 1–4 take the indicated hotel category (log-transformed county-year count) as the dependent variable (All hotels, Apartment, High-star hotels, and Homestays); column 5–8 takes the annual mean predicted bird species richness as the dependent variable. * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

Heterogeneity Analysis

We further explore whether this negative association is distributed evenly across different contexts. Table S9 presents the results of the heterogeneity analysis. First, we examine whether the impact varies by ecosystem type, interacting the treatment indicator with dummy variables for different ecosystem types (Forest, Wetland, Desert, Wildlife, and Wild Plants). The results reveal that the negative response in bird diversity is significant precisely in areas adjacent to ecosystems often valued for tourism: forests, wetlands, and wild plant habitats. Specifically, the designation of tourism villages is associated with significant declines in species richness in counties near forest ecosystems (β = -0.733, *p* = .012), wetland ecosystems (β = -0.765, *p* = .045), and wild plant reserves (β = -0.843, *p* = .049). In contrast, no statistically significant effects are detected for counties near desert or wildlife reserves. We also investigate heterogeneity by local economic development, categorizing counties into low- and high-GDP groups based on whether baseline GDP was below or above the sample median. The policy effect for low-GDP counties is negative and statistically significant (β = -0.792, *p* = .019). A complementary partition by pre-policy land-use composition reveals a more textured pattern than the protected-area types alone. The negative policy effect is present across all land-use partitions but is somewhat stronger in counties with below-median pre-policy ecological-land share (β = -0.712, *p* = .016) and below-median pre-policy built-up share (β = -0.740, *p* = .019), while it is also strong in counties with above-median pre-policy forest share specifically (β = -0.707, *p* = .009).

Discussion

This study aimed to assess whether the designation of the National Key Rural Tourism Villages has effects on local bird species richness. The estimates reveal a substantial decline in mean bird species richness after designation, with event-study evidence of negligible pre-trends and negative post-treatment coefficients. The mechanism analysis with hotel POIs suggests that the result is consistent with the route of expanding accommodation capacity. The land-use heterogeneity clarifies that this capacity expansion in ecologically rich settings coexists with a higher per-unit ecological cost in forest-dominated counties. Our estimates indicate that the ecological cost of rural tourism expansion, although small, is policy relevant under the institutional form of Key Rural Tourism Villages.

The literature provides countervailing expectations of tourism and biodiversity. Tourism can fund conservation, valorize natural assets, and incentivize landscape restoration, but on the other hand it can fragment habitats, increase disturbance, and introduce novel pressures (R. Buckley, 2011; Pickering & Hill, 2007). Our results provide empirical support for the latter in the short-to-medium run under the policy framework of Key Villages. The negative effect on bird richness does not mean that tourism cannot be made biodiversity-positive; it shows that when development tilts towards visible features and human presence, ecological costs materialize, even in largely rural areas. Of these hotel-like accommodations, high-star hotel intensity is the form most strongly associated with bird-richness decline, while aggregate hotel intensity is weakly negatively associated, and apartment hotels and homestays are unassociated in the same specification. High-star accommodation is the product most associated with hardscape, on-site amenities, lighting, and acoustic footprint and traffic generation, and as such may be the type whose growth most directly translates into the habitat-quality and disturbance mechanisms, even though the policy itself does not, on average, significantly drive high-star hotel expansion on the full sample. This interpretation remains context specific. Neutral or beneficial outcomes remain plausible where visitation is low-density, sensitive habitats are buffered, and disturbance is seasonally managed. The negative average effect identified here should therefore be read as evidence about the design and implementation of the Key Village policy during the study period, rather than as evidence that tourism development necessarily reduces bird diversity in every setting.

Several mechanisms are consistent with this pattern. First, designation can spur construction of visitor infrastructure and beautification that replaces scruffy, semi-natural patches with manicured lawns, hardscape, or ornamental plantings (Benton et al., 2003). For many birds, these scruffy areas provide nesting material, insect prey, and shelter. That is, even if total green cover increases, habitat quality can decline through homogenization (McKinney, 2006). Second, increased human presence increases acoustic disturbance, artificial light at night, and temporal unpredictability of movement (Francis & Barber, 2013; Longcore & Rich, 2004). Many species respond by shifting activity to suboptimal times, abandoning micro-sites, or increasing their flight initiation distance (Steven et al., 2011). Such behavioral responses can reduce detectability and local presence simultaneously, for sensitive or crepuscular taxa. Finally, early implementation often prioritizes quick visible change to meet performance and reporting timelines (Gao & Wu, 2017), which compresses the time window for ecological planning, and is consistent with the front-loaded event-study pattern we find. The hotel POI evidence is consistent with these pathways sharing a common mechanism related to driving visible-amenity-led development. Hotels and associated roads may extend these effects beyond their immediate footprint through traffic, pedestrian activity, noise, and artificial lighting, thereby reducing habitat quality and prompting avoidance of otherwise suitable patches (Dertien et al., 2021; Steven et al., 2011). Field studies in birdwatching and nature-tourism settings show that close approaches and noisy behavior can disturb birds, with responses varying by tourist distance, group size, and species sensitivity (Aas et al., 2023; Avenzora et al., 2024). Repeated displacement can also reduce foraging time, particularly for individuals with limited capacity to compensate for interruptions (Van Der Kolk et al., 2021). These mechanisms strengthen the biological plausibility of the hotel-based pathway but do not establish causal mediation.

The event-time evidence suggests two phases of operation: an early front-loaded disturbance around the designation period and a later operating regime. The sudden post-designation decline is consistent with an implementation shock in which construction, renovation, and rapid site reconfiguration occur in parallel with abrupt increases in visitor presence and vehicle flow, creating acute disruption through habitat alteration and increased unpredictability of human activity (Shannon et al., 2016; Sih et al., 2011). The subsequent attenuation or partial easing can plausibly be the result of a mix of stabilization processes as sites stabilize into new configurations and disturbance regimes become more predictable

(Clergeau et al., 2006; Vincze et al., 2016). This timing pattern, in combination with the spatial pattern of stronger effects in forest-dominated and lower-GDP counties, suggests that habitat fragmentation and the timing of intervention are at least as important to the ecological outcome as the absolute number of tourists that we end up affecting. The period immediately surrounding designation is a high-leverage window for ecological safeguards, because decisions made then, where to build, what to pave, and how to light can lock in long-run habitat quality and disturbance regimes (Kiesecker et al., 2010).

The heterogeneity patterns indicate how policy-induced expansion of tourism is most likely to collide with ecological sensitivity and governance constraints. Stronger negative responses were found in counties with forest, wetland, and wild-plant ecosystems. This implies that the trade-off is concentrated where both baseline habitat quality and tourism attractiveness are high, and where bird assemblages may be more habitat-specialized and disturbance-sensitive (Balmford et al., 2009; Şekercioğlu et al., 2002). Forests and wetlands may support species with more restrictive nesting and foraging requirements, and many of these species have strong responses to noise, edge, lighting, and humans, so that small changes in access and visitation can lead to disproportionately large ecological benefits (Reed & Merenlender, 2008). The land-use partitions add precision to this picture; the policy effect is present across both eco-poor and eco-rich counties, but is largest in absolute magnitude in counties dominated by forest cover specifically. The per-unit-ecological-cost-of-construction-led-tourism is largest where the underlying habitat is forest-dominated, while the broad expansion of the accommodation-capacity documented in the mechanism analysis is not limited to any particular land-use category. These layered channels may operate in different counties and collectively produce a broadly distributed but unevenly weighted national signal. Contrast this with null or weaker signals near deserts or wildlife-reserve contexts, which may represent lower baseline richness, different visitor modalities, or stronger pre-existing access constraints. The stronger negative response in lower income counties points to an institutional capacity channel rather than just an ecological channel. Where fiscal space and administrative capacity are limited, the marginal political return to rapid, visible tourism upgrading is high, while the capability to plan ecologically sensitive layouts is limited (Eaton & Kostka, 2014; Kostka & Mol, 2013; Tosun, 2000). This type of tourism development path tends to involve intensive land conversion, standardized beautification, and a reliance on volume-based revenue strategies that scale up ecological costs (Chok et al., 2007).

The results of this study can offer suggestions for the policy design of rural tourism. First, biodiversity outcomes should be made legible and consequential within the designation system. Ecological performance indicators should be incorporated into entry criteria and continued recognition so that villages and local governments have credible incentives to maintain habitat heterogeneity and manage disturbance. Such criteria should not be limited to broad greenness or tree-planting indicators, which can reward homogenized landscaping, but should instead prioritize habitat complexity and disturbance governance during breeding and migration periods (Dearborn & Kark, 2010; Font & Harris, 2004). Second, the program should regard the designation year as a regulated transition period with mandatory ecological planning. Before major construction or facility expansion, villages should be required to prepare a site-level habitat and disturbance management plan. The plan should include spatial zoning to protect sensitive habitats and visitor routes that minimize fragmentation, as well as standards to control important causes of disturbance (Geneletti, 2013). Route-level monitoring can reveal when and where visitor use concentrates and thereby inform targeted zoning, rerouting, or temporary access restrictions (Costa et al., 2024). Where locally validated disturbance-response evidence is available, visitor routes and setback distances should be calibrated to species and site conditions rather than a universal distance (Dertien et al., 2021). Because of the evidence that high-star hotel intensity is most negatively related to bird richness, the application process for new high-star hotel sites in ecologically sensitive locations should require enhanced ecological assessment and explicit zoning restrictions, while lower-footprint accommodation types should be encouraged. Third, operational management should explicitly regulate tourist behavior and the timing of visitation, because disturbance regimes are often more consequential for birds than are small changes in infrastructure aesthetics. Capacity-oriented tools and seasonally calibrated rules can be used to smooth peaks and tie human activity to periods of heightened ecological sensitivity (Monz et al., 2013), while behavior-change measures should be tailored to the specific beliefs and practical barriers that sustain harmful conduct among different visitor groups (Povilionis et al., 2025). This management framework could be built on community education to create a culture of stewardship, in which residents understand the reason for ecological restrictions and will voluntarily participate in disturbance mitigation. Fourth, monitoring should be institutionalized as an adaptive-management tool rather than a one-time assessment. Citizen-science checklists provide a scalable backbone for tracking biodiversity in the design of this study, but policy monitoring should incorporate multiple complementary streams, such as periodic professional surveys for validation and habitat audit via remote sensing (Conrad & Hilchey, 2011). Monitoring should not be framed as purely punitive, because villages will otherwise underreport or resist (Plummer & Fennell, 2009). Finally, the program should encourage a shift from infrastructure-led mass visitation to nature-based value creation that internalizes biodiversity as an asset (C. H. Sekercioglu, 2002). Birdwatching, soundscape tourism, and seasonal ecological interpretation can create revenue by maintaining bird presence and habitat quality, transforming conservation from a limitation to a product attribute (Pijanowski et al., 2011). Participatory carrying-capacity assessments can combine physical, ecological, managerial, and community knowledge when setting route limits for bird tourism (Orozco et al., 2026).

More broadly, although birds are our indicator taxon here, our results have implications well beyond avian diversity and beyond rural China. The transmission channels we document (visible-amenity-led construction, accommodation upgrading concentrated in ecologically rich areas, and front-loaded human disturbance during a designation window) are common features of certified-destination programs around the world and are not bird-specific. Taxa with strong site-fidelity, narrow microhabitat requirements, or sensitivity to noise and night lighting (e.g., bats, amphibians, ground-nesting insects) are likely to be affected by the same mechanisms. By the same logic, urban and peri-urban accreditation schemes that accelerate visible infrastructure investment may face analogous trade-offs even where the rural label does not apply. We interpret our findings as suggesting that tourism-accreditation systems, when implemented, should treat the designation moment itself as a high-leverage opportunity for environmental governance.

Limitation

This study still has limitations. We focus on checklist richness proxies and do not evaluate abundance, community composition, or functional diversity (Dornelas et al., 2014; Hillebrand et al., 2018). We do not report Shannon's diversity index because the raw checklist files contain checklist-level species richness. Effort standardization reduces observable differences in sampling intensity but cannot estimate species-specific detection probabilities for every checklist, fully control observer skill and reporting behavior, or remove preferential sampling of accessible and attractive sites; abrupt disruptions may also change monitoring coverage (Backstrom et al., 2025; Gillings et al., 2021; Hochachka et al., 2023; Robinson et al., 2021; Scher & Clark, 2023). Future research should therefore combine repeated-visit occupancy models, structured validation surveys, and explicit spatial sampling models. It should also examine community composition, species turnover, and functional diversity, because relatively small changes in richness may mask replacement of specialists by disturbance-tolerant generalists or changes in ecological functions (Avenzora et al., 2024; Otieno & Mutati, 2021). Additionally, this study uses the nearest Key Village's year as the treatment-onset proxy, exposing the design to an administrative-centroid mismatch with the actual area of influence (Wooldridge, 2010). If influence decays with distance and many counties have their centroid outside the core catchment, then attenuation bias is likely; the distance-restriction and spatial-lag robustness checks alleviate but do not eliminate this concern. Relatedly, our distance-based exposure robustness uses straight-line distance only; we do not have road-network distance, transit accessibility, or functional-connectivity measures, and so cannot rule out misclassification along these dimensions, although the spatial-lag specifications partially absorb that residual correlation. Last, our exposure measure is policy designation, not visitor flows, and the hotel POI mechanism evidence speaks to accommodation capacity rather than to realized tourist throughput; direct measurement of visitor volumes (e.g., gridded mobility data, ticketing records) would be needed to translate accommodation expansion into per-capita pressure. These limitations counsel interpretive restraint. In particular, the variation in statistical significance across the three richness metrics means that the directional conclusion is more robustly supported than any specific point estimate, and the overall findings should be interpreted with commensurate caution. The study provides credible quasi-experimental evidence of a negative average association on county-year bird richness proxies after designation.

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