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The development of wind power leads to a decline in bird species richness

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A R T I C L E I N F O

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A B S T R A C T

In recent years, the rapid growth of the wind power industry has increasingly drawn attention to its negative effects on bird species. Although some studies have assessed the impact of wind turbines (WTs) on birds at local scales, comprehensive analyses at the national level remain scarce, hindering the ability to identify broader, systematic effects. This study focuses on China, using county-level administrative units as the research units to examine the impact of WTs on bird species richness. Specifically, we first conducted a comparative analysis of bird species richness between counties with and without WTs. Then, based on the potential distribution patterns of 1024 bird species, we calculated the intensity of WT impacts on bird distribution at the county level. Finally, we employed generalized additive models to further characterize the patterns of species richness change in response to increasing WT impact intensity. The results revealed that the richness of all bird, resident and migratory species, was significantly lower in counties hosting WTs compared to non-WT counties, and species richness began to decline rapidly once the intensity of WT impacts exceeded 0.2. This study enhances our understanding of the ecological and environmental consequences of wind power development and provides scientific evidence for future wind farm site selection and policy formulation, with the aim of balancing renewable energy development and biodiversity conservation.

1. Introduction

As the largest carbon emitter globally, China's transition to low-carbon energy is pivotal for achieving the 2 °C temperature rise limitation target outlined in the Paris Agreement (Liu et al., 2022b). Wind energy, as a key renewable resource, is central to decarbonizing the energy system (Pryor et al., 2020; Pryor et al., 2018) and plays a crucial role in optimizing China's energy structure and achieving carbon neutrality (Liu et al., 2022a). Against this backdrop, China's wind energy industry has made substantial progress in recent years. By 2022, the total installed capacity reached 365.96 GW, accounting for 40.7 % of the global wind energy capacity, solidifying China's position as the world leader (IRENA, 2023). In line with China's "dual carbon" targets (UN, 2020), the country plans to expand the installed generation capacity of wind and other renewable energies to over 1200 GW by 2030 (Liu et al., 2022b). While wind energy plays a vital role in mitigating climate change and reducing greenhouse gas emissions (Barthelmie and Pryor, 2014), its development has also led to various ecological and environmental challenges. For example, the operation of wind farms (WFs) can result in bird and bat fatalities from collisions (Loss et al., 2013), noise pollution (Nazir et al., 2020), soil erosion (Xia et al., 2025), and significant impacts on local and regional climates (Liu et al., 2023; Miller and Keith, 2018), vegetation, and ecosystem dynamics (Qin et al., 2022; Su et al., 2024; Wu et al., 2023; Dang et al., 2025a). Therefore, a more comprehensive understanding of the multifaceted ecological and environmental impacts of renewable energy development is essential for the scientific planning of resource utilization and ensuring its sustainable growth (Xia et al., 2024).

Among these impacts, the effect of wind turbines (WTs) on bird populations can be categorized into direct and indirect effects (Miao et al., 2019; Zhu et al., 2016). Direct effects refer to bird fatalities caused by collisions with WTs, such as the study by Carrete et al. (2012) in southern Spain, which found a significant increase in griffon vulture mortality near WTs in densely populated areas. Indirect effects include

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noise pollution, habitat fragmentation, and bird avoidance behavior resulting from the construction and operation of WTs. For example, Pearse et al. (2021) found that in the Great Plains of the United States, whooping cranes avoided areas within 5 km of WTs and were 20 times more likely to nest in areas farther from them. Therkildsen et al. (2021) noted that the presence of WTs caused birds to alter their flight direction and altitude when approaching. Moreover, some studies have suggested that the impact of WTs on birds is closely related to the ecological characteristics of the construction areas. For instance, Heuck et al. (2019) found that in areas with high habitat suitability, the impact of WTs on birds was more pronounced, while Rehling et al. (2023) observed that the effect of forest-based WTs on bird populations was closely related to forest structure, with forest bird communities being more sensitive to forest quality than to the presence of WTs. Additionally, large-scale studies on these patterns are scarce. Some researchers have used fixed effects models to find that increased WT development leads to a decrease in bird numbers and richness at the national scale (Meng et al., 2025; Miao et al., 2019). However, these studies have not adequately considered the ecological characteristics of the WT construction areas and their role in supporting bird diversity, which could lead to inaccurate assessments of the impacts of WTs.

In conclusion, while existing research has provided valuable insights into the impact of WTs on birds, these studies are limited by their scale and cannot reveal broader, systematic patterns. Moreover, few studies have explored the potential nonlinear relationship between WTs and bird richness, and even fewer have considered differences in the behaviors of bird species (resident vs. migratory birds). Therefore, this study, based in China, aims to investigate whether WTs affect the richness of all bird, threatened, resident, and migratory species, and to analyze the relationship between WT impact intensity and species richness. To achieve this, we compared richness between counties with and without WTs and then characterized the patterns of species richness change in response to increasing WT impact intensity. This study will contribute to a deeper understanding of the ecological and environmental impacts of wind power development and provide scientific evidence for future WF site selection and related policy formulation.

2. Materials and methods

2.1. Data sources

The data used in this study are presented in Table S1. The WT distribution data were sourced from a dataset independently interpreted by the research team, which is based on OpenStreetMap data and Sentinel imagery. This dataset provides the annual spatiotemporal distribution of offshore and onshore WTs in China since 2014, with a spatial accuracy exceeding 85 %, meeting the research requirements (Lyu et al., 2025). For analysis, the WT distribution data for 2020 selected from this dataset, with a total of over 140,000 WTs. This number is close to the number of WTs identified in other studies (He et al., 2025; Yang et al., 2025a; Yang et al., 2025b). The distribution of counties with more than 10 WTs is shown in Fig. 1a, which covers 784 county-level administrative units.

The bird species list used in this study primarily derives from the China Bird List v8.0 (2020) (China Bird Report, 2020) and the China Bird Classification and Distribution List (Zheng, 2017), with additional updates based on data from GBIF and eBird (Sun et al., 2022). Only bird species observed in mainland China between 2015 and 2020 were included in the list used for this study. The bird observation records come from the China Birdwatching Record (http://www.birdreport.cn/). Each record includes the species name, survey location (including county name and coordinates), and survey date. Species with fewer than 50 observation records were excluded (Sun et al., 2022). In total, 1024 bird species were included in this study, from 2542 county-level administrative units (Fig. 2).

Threatened bird species are those listed as Critically Endangered (CR), Endangered (EN), and Vulnerable (VU) in the International Union for Conservation of Nature (IUCN) Red List, totaling 45 species. The classification of resident and migratory birds is based on the China Bird Classification and Distribution List (Zheng, 2017), with mixed habitat species excluded from the analysis. In total, there are 426 resident bird species and 393 migratory bird species (Fig. 2).

2.2. Methods

2.2.1. Overview of the methodology

To quantitatively assess the impact of wind power on bird richness,

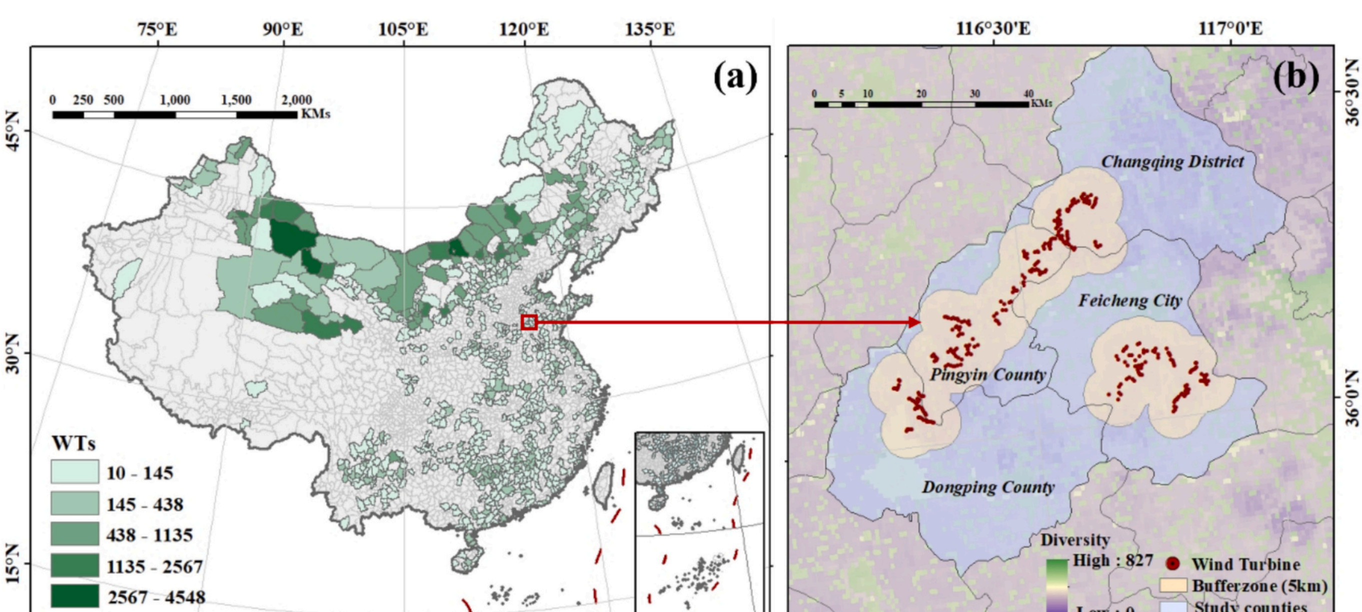

Fig. 1. Study area: (a) 784 counties with 10 or more WTs; (b) Spatial example of WTs and their 5 km buffer zone.

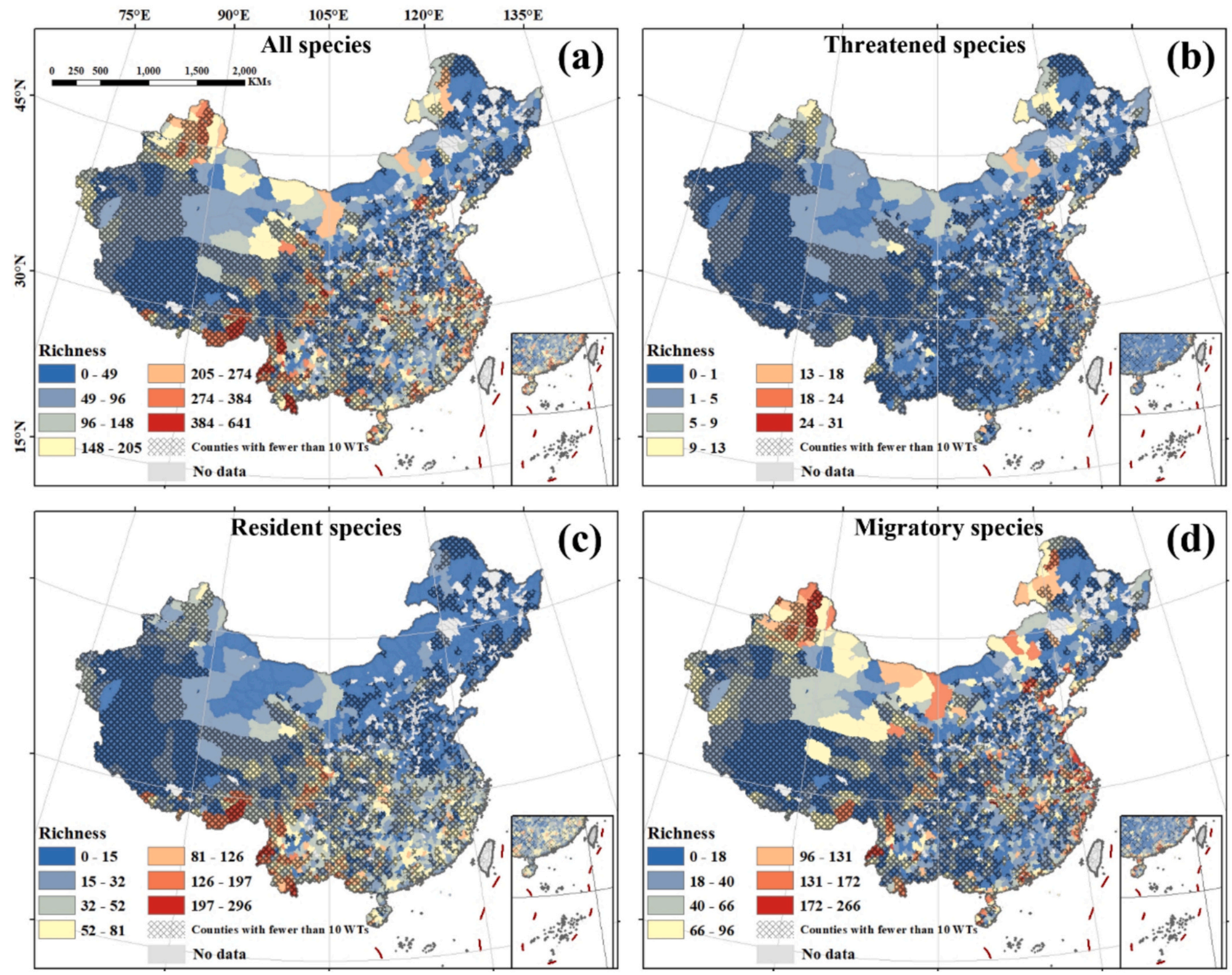

Fig. 2. Bird species richness at the county scale for four bird categories: (a) all species, (b) threatened species, (c) resident species, and (d) migratory species.

this study conducted two analyses:

(1) Differential analysis between counties with and without WTs.

First, based on WT distribution, counties with 10 or more WTs were categorized as counties with WTs, totaling 784 counties (Fig. 1), while the remaining counties were categorized as counties without WTs, totaling 1758 counties. Second, using bird observation records from the China Birdwatching Record Center, bird richness at the county level was calculated for all bird, threatened species, resident birds, and migratory birds (Fig. 2). To enhance the robustness of the analysis and mitigate potential imbalances in citizen science data, this study adopts a spatial moving window approach. Specifically, each analysis unit comprises a focal county and its 10 geographically nearest neighboring counties, forming an 11-county moving window. Within each window, the average bird species richness is calculated separately for counties with and without WTs. Windows composed entirely of either WT or non-WT counties are excluded from the subsequent analysis. This approach enables full coverage of all counties and the construction of matched paired samples. Furthermore, as the data did not follow a normal distribution, the study employs the Wilcoxon signed-rank test on the paired samples to assess differences in richness of all bird, threatened, resident, and migratory species, between counties with and without WT.

(2) Patterns of species richness change in response to increasing WT impact intensity.

To evaluate the impact of WTs on bird distribution, the ratio of the WT impact area to the county-level administrative unit was introduced as a measure. Specifically, previous observational studies have indicated that after the construction of wind farms, bird activity within a 5 km radius around the WTs was significantly affected. (Gómez-Catasús et al., 2018; Pearse et al., 2021; Tolvanen et al., 2023; Kelly et al., 2025). Therefore, the 5 km buffer zone around WTs was defined as the WT impact area (Fig. 1b). Next, using the species distribution modelling (SDM) to predict the potential distribution of birds, the total bird diversity within the WT impact area for each county was calculated, as well as the total bird diversity for the entire county. The ratio of these two values was then defined as the intensity of WT impact on bird distribution at the county scale (Fig. 5). Compared to directly using the number of WTs as an independent variable (Meng et al., 2025), this metric more accurately reflects the ecological background of WT construction areas and their capacity to support bird diversity, providing a quantitative basis for assessing the impact of WTs on bird diversity. Based on this, the study applies a “space-for-time” approach (Wu et al., 2022; Wu et al., 2025), for the 784 counties with WTs, bird species richness is used as the dependent variable, and the WT impact intensity is treated as the core independent variable. Generalized additive model

(GAM) is employed to describe the nonlinear relationship between WT impact intensity and bird richness.

2.2.2. Species distribution modelling

This study used the MaxEnt algorithm from the "SSDM" package to predict bird distribution, with 75 % of the occurrence points used as training data and the remaining points as testing data. Based on relevant studies (Sun et al., 2022; Yang et al., 2024), the environmental variables used to fit the SDMs included 19 bioclimatic variables (bio1 to bio19) from the WorldClim2 dataset (Fick and Hijmans, 2017), along with elevation, slope, roughness (Xiao et al., 2018), NDVI, and land use data (Yang and Huang, 2021), totaling 24 environmental variables. To ensure model accuracy and avoid overfitting due to multicollinearity among the independent variables, principal component analysis (PCA) was performed on the original 24 variables. By extracting 95 % of the variance, 12 composite variables were generated.

The model's performance was validated using the Area Under the Curve (AUC) from the Receiver Operating Characteristics (ROC) curve. Additionally, for each species, five repeated simulations were conducted, and the mean habitat suitability and AUC values of these simulations were calculated to serve as the predicted distribution probabilities and the evaluation metrics of model accuracy. The equal training sensitivity and specificity logistic threshold was selected to binarize the mean habitat suitability, with 1 representing potential distribution areas and 0 indicating no distribution. The AUC values of the models for 1024 bird species ranged from 0.79 to 1.00, with an average AUC of 0.95, indicating that the majority of the models exhibited good predictive accuracy (Yang et al., 2024) (Fig. S1). Finally, the binarized results were spatially overlaid to output the potential distribution patterns of all bird, threatened species, resident birds, and migratory birds (Fig. 3).

2.2.3. Generalized additive model

A GAM was used to describe the nonlinear relationship between WT impact intensity and bird richness, with calculations and analyses performed using the "mgcv" package in R. Previous studies have shown that bird richness is influenced by various factors, such as regional ecological environment, agricultural land, degree of urbanization, and proximity to coastlines (Sun et al., 2022; Yang et al., 2024). Additionally, bird observation data are affected by factors like county area, population density, and economic development level. To account for spatial heterogeneity in bird species richness and mitigate the imbalance inherent in citizen science data, eight control variables were included in the model: county area, potential bird diversity (Fig. 3), distance to the coastline, impervious surface ratio, nighttime light intensity, GDP per capita, and population density, the proportion of farmland in the county area (Liao et al., 2024; Sun et al., 2022; Yang et al., 2024). The relevant data sources are listed in Table S1. After calculating the correlation coefficients among variables, we found that the impervious surface ratio

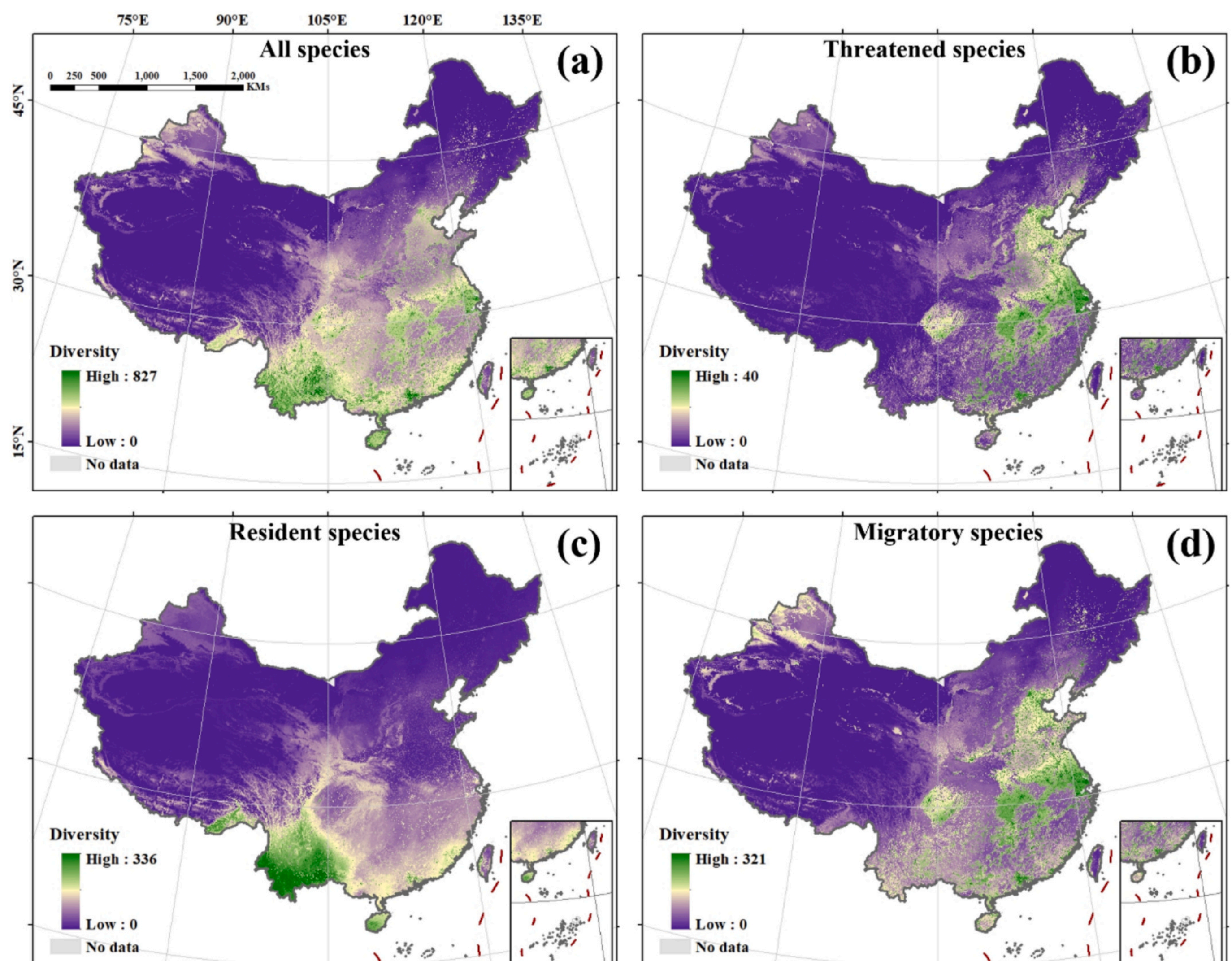

Fig. 3. Potential distributions of four bird categories: (a) all species, (b) threatened species, (c) resident species, and (d) migratory species.

was strongly correlated with other variables ($r > 0.7$, $p < 0.01$). Therefore, this variable was excluded from the subsequent analyses.

To reduce data noise, eliminate local outliers, and address uneven data distribution, a moving window method was employed to smooth the data (Dang et al., 2025b). The data were sorted by WT impact intensity in ascending order, and the average value of variables within the moving window was calculated with a step size of 1, resulting in data points for subsequent analysis. This procedure also effectively disrupted the spatial continuity among counties and substantially reduced the potential influence of spatial autocorrelation on the results. To test the effect of window size, 11 different window sizes ranging from 30 to 130 (at intervals of 10) are applied. After excluding variables with a variance inflation factor (VIF) greater than 10, GAMs are fitted. Across all window sizes, the richness trends remain consistent, with thresholds observed at a WT impact intensity of 0.2 for all, resident, and migratory species (Fig. S2). Among these, the model using a window size of 100 yields a relatively low mean squared error (MSE) across the four bird richness categories (Table S2) and incorporates a greater number of contributing variables. This suggests that, under this window size, prediction error is minimized and model performance is optimized; therefore, a window size of 100 is selected. Additionally, all GAMs report adjusted R^2 values greater than 0.9, indicating strong model fit.

To test whether the effect of WT impact intensity on bird species richness differs from random expectations, the observed results are compared with 100 GAM models fitted to randomized datasets. In these randomizations, bird species richness and control variables for each county are held constant, while WT impact intensity values are randomly reassigned (Felipe-Lucia et al., 2020). This procedure allows us to determine whether the observed patterns are truly associated with WT impact intensity or could occur by chance (Wu et al., 2022). The comparison reveals a clear divergence between the observed trends and those derived from the 100 randomizations (Fig. S3), indicating that the effect of WT impact intensity on bird richness is not random, but strongly associated with the presence of WTs.

3. Results

3.1. The differences in bird richness between counties

The differences in bird richness between counties with and without WTs are shown in Fig. 4. The average richness of all bird species in counties with WTs was 106.8, which was significantly lower than the 109.2 observed in counties without WTs ($p < 0.01$). Similarly, the average richness of both resident and migratory species was significantly lower in counties with WTs compared to those without ($p < 0.01$). However, no significant differences were observed in the richness of threatened species between the two types of counties.

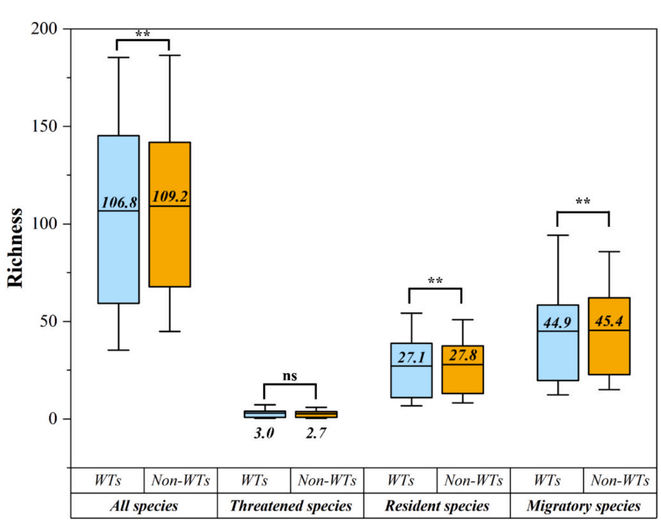

Fig. 4. Differences in bird species between counties for four bird categories ($^{ns}p > 0.05$, $^{**}p < 0.01$).

3.2. The intensity of WT impact on bird distribution

The intensity of WT impact on bird distribution is illustrated in Fig. 5. In terms of spatial distribution, the impact intensity of WTs on all bird, threatened, resident, and migratory species follows a relatively consistent spatial pattern. All four categories of impact intensity exhibit significant spatial clustering ($p < 0.001$), with hotspot areas concentrated in northern and eastern China, and a low-high clustering pattern observed (Fig. S5). Cold spot areas are located in Xinjiang and southern China (Fig. S4). According to the SDM, the hotspots of climatically based potential ranges of threatened and migratory species are found along the eastern coastal regions (Fig. 3b, d), which largely overlap with the hotspots of WT impact intensity. This spatial overlap suggests that regions with high climatic suitability for these species also experience greater WT impact intensity (Fig. 5b, d). In terms of magnitude, the average WT impact intensity across the 784 counties was 0.19 ± 0.16 for all bird, 0.18 ± 0.16 for threatened species, 0.20 ± 0.16 for resident species, and 0.19 ± 0.16 for migratory species.

3.3. The pattern between WT impact intensity and bird species richness

The GAM results are shown in Fig. 6. WT impact intensity significantly affects the richness of all bird, threatened, resident and migratory species ($p < 0.001$). Since variables with a VIF greater than 10 were excluded, the variables included in the GAM models for the four bird categories are not consistent. In general, in addition to WT impact intensity, control variables such as distance to the coastline, nighttime light intensity, population density, and GDP per capita were incorporated into the GAM fitting. The proportion of farmland area was not included in the construction of any of the models. Different variables exhibited certain nonlinear relationships with bird richness (Fig. S6).

Focusing specifically on WT impact intensity, the richness of threatened species did not show a clear trend with increasing WT impact intensity. However, the richness of all bird, resident and migratory species fluctuated with increasing WT impact intensity, but declined sharply once the intensity exceeded 0.2. Among these, resident species exhibited a more pronounced decline compared to migratory species (Fig. 6).

4. Discussion

Accurately understanding the impact of WTs on bird diversity provides important insights for WF site selection and wind energy policies, which can enhance the sustainability of wind energy utilization (Miao et al., 2019). Existing research has mainly focused on the impact of specific WFs on birds (Dohm et al., 2019; Pearse et al., 2021), but such studies may be difficult to generalize due to the influence of factors like WF type, local bird richness, and ecological conditions (Sovacool, 2013). In contrast, some large-scale studies have used fixed-effects models to analyze the impact of WTs on birds (Meng et al., 2025; Miao et al., 2019), but these studies overlook the ecological background of WT construction areas and their ability to support bird diversity. In this study, we tested the differences in bird richness between counties with and without WTs, and based on the potential distribution patterns of birds and WT buffer zones, we defined the intensity of WT impact on bird distribution at the county scale across China. We further characterized the changes in bird richness under increasing WT impact intensity using GAM. Compared to the previously mentioned studies, this research advances the understanding of WT impacts on bird richness at a large scale. It spatially reveals the patterns by which WTs affect bird richness and reveals that the effect of WTs on bird richness follows a nonlinear pattern with an impact threshold. Moreover, the analytical

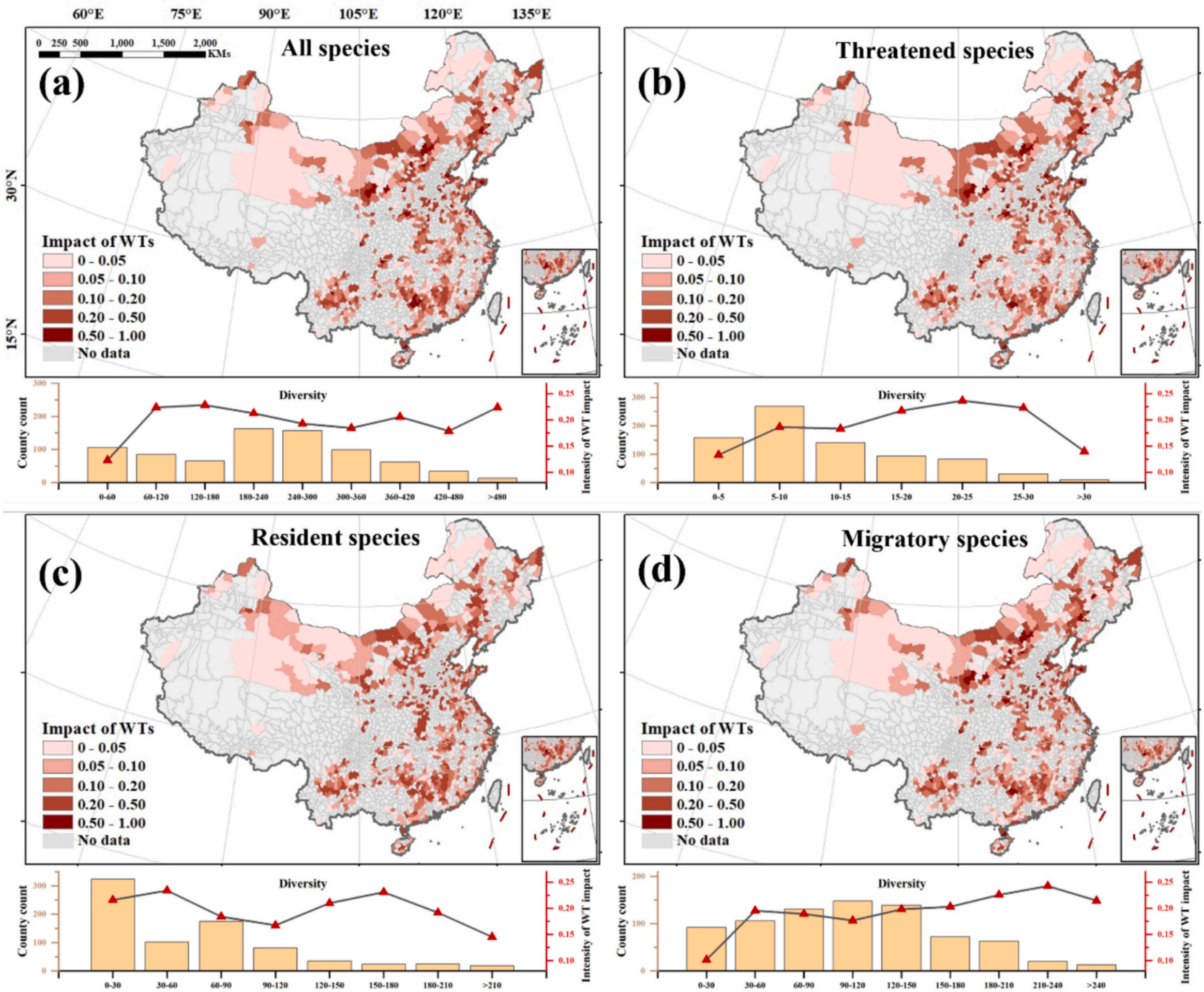

Fig. 5. Intensity of WT impact on the distribution of four bird categories. The histograms below represent the county-level statistics of climatically-based potential ranges for each bird category. The lines indicate the average intensity of WT impact across counties: (a) all species, (b) threatened species, (c) resident species, and (d) migratory species.

framework proposed in this study can be refined and applied to future research on the impact of WTs or photovoltaic plants on species richness.

4.1. Analysis of the impact of WTs on bird richness

The study found that the richness of all bird, resident and migratory species in counties with WTs was significantly lower than in counties without WTs (Fig. 4), indicating that WTs negatively impact bird richness across China. This result is consistent with the findings of Meng et al. (2025), who noted that for every standard deviation increase in the number of WTs in China, bird species richness decreases by 12.2 %. The impact of WTs on resident birds is particularly pronounced, likely due to the fact that most birds killed by WTs are resident species. For example, Zhang et al. (2022) found in their study of WFs along the Chinese coastline that the birds killed by WTs were mainly resident species or those that had already bred in the study area. Similarly, Karen et al. (2009) observed in their study of WFs in the Netherlands that most of the birds killed by WTs were diurnal resident species. Additionally, Barrios and Rodríguez (2004) found in their study on the Iberian Peninsula that most of the birds killed by WTs were resident species. Migratory birds, on the other hand, are particularly sensitive to environmental threats (Burger and Gochfeld, 1991), can detect the presence of WTs, and alter their flight paths when approaching them. As a result, their population numbers are not significantly impacted (de Lucas et al., 2004). However, the updraft areas that migratory birds commonly rely on often overlap with areas where WTs are concentrated, which could lead to the loss of their functional habitats (Barbosa et al., 2020). Moreover, birds are sensitive to noise, and prolonged exposure to high-noise environments may cause them to avoid these areas, affecting their population stability (Carral-Murrieta et al., 2020). For all bird species, this may result in the abandonment of originally suitable habitats, further contributing to a decline in richness.

Further analysis revealed that when the WT impact intensity reached 0.2, the richness of all bird, resident and migratory species showed a decreasing trend, while the richness of threatened birds exhibited fluctuating changes (Figs. 6). This result aligns with the differential analysis, which showed that the richness of all bird, resident and migratory species in counties with WTs was significantly lower than in counties without WTs, while no significant differences were found between the two types of counties for threatened species (Fig. 4). The appearance of a

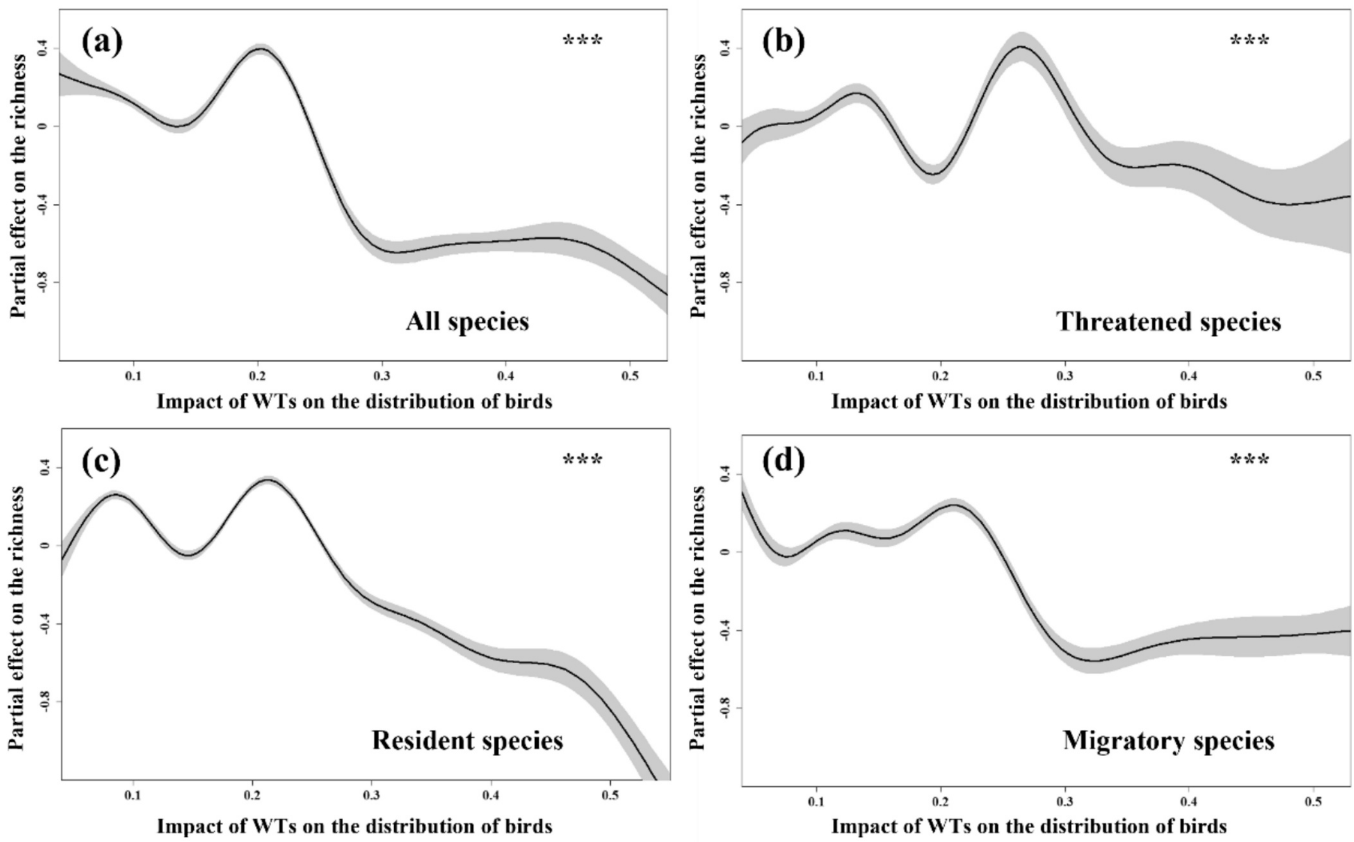

Fig. 6. Effects of WT impact intensity on bird richness for four bird categories: (a) all species, (b) threatened species, (c) resident species, and (d) migratory species.

threshold point in the curve suggests that in the early stages of increasing WT impact intensity, bird richness did not directly decrease. Rather, it was only when the intensity reached 0.2 that the negative impact on bird richness sharply increased. A study conducted in Greece lends further support to this conclusion, Bounas et al. (2025) found that if no additional WTs were constructed, populations of both Cinereous vultures and Griffon vultures would steadily grow. However, introducing 85 new WTs would likely drive the Cinereous vulture population to local extinction within 18 years and significantly suppress the growth rate of the Griffon vulture population. Moreover, under scenarios involving further WT expansion, both species could face complete extinction within just two to five years. This phenomenon can be explained by findings from other studies. For instance, Desrochers et al. (2011) found in Ontario, Canada, that as natural areas were converted to human-dominated land cover, changes in bird species richness followed a peak relationship. When the loss of natural areas did not exceed 44 %, bird richness actually increased. This may be because, as natural areas decline, habitat richness initially increases as human-dominated habitats introduce new habitat types that provide shelter for new species, leading to an increase in species richness. As a result, the relationship between bird richness and habitat loss is not linear (Banks-Leite et al., 2012). However, continued transformation of natural areas and increasing habitat fragmentation degrade the ecological functionality of habitats, ultimately resulting in reduced species richness (Desrochers et al., 2011).

For other control variables, the species richness of all bird—as well as that of threatened, resident, and migratory species—was significantly negatively correlated with distance from the coastline (Fig. S7). This may be attributed to the presence of typical wetland ecosystems in eastern coastal China, particularly along the Yellow Sea and Bohai Sea coasts, such as coastal salt marshes and estuarine wetlands. These areas serve as critical stopover and foraging sites for migratory birds along the East Asian–Australasian Flyway (Wang et al., 2022), and are also among the regions with the highest bird species richness in China (Sun et al., 2022). Bird species richness was also significantly positively correlated with nighttime light intensity (Fig. S7), consistent with findings by Sun et al. (2022), who reported that bird distribution hotspots often overlap with highly urbanized areas. In addition, significant positive correlations were found between bird richness and both population density and economic level (Fig. S7), which may be explained by underreporting of bird observations in counties with lower population density and weaker economic development.

4.2. Policy recommendation

Based on the previous interpretation of the relationship between WT impact intensity and bird richness, we propose the following two recommendations. First, our study revealed the existence of a threshold point at 0.2. According to the impact intensity assessment for 784 counties (Fig. 5), 36.1 %, 39.0 % and 33.7 % of the counties have an impact intensity greater than 0.2 for all bird, resident and migratory species, respectively. This suggests that in these counties, an increase in WTs will have a stronger negative effect on bird richness. Therefore, future WF site selection should avoid these counties or important resident bird habitats within counties (Li et al., 2020). Secondly, our study revealed that the hotspot regions of climatically suitable potential ranges for threatened and migratory bird species are mainly concentrated along the eastern coastal areas of China (Fig. 3b). However, these areas also coincide with the main development zones for coastal wind farms, where WT impact intensity is high (Fig. S4). Therefore, the establishment of new wind and solar energy facilities in these regions should be approached with greater caution (Zhang et al., 2025).

Additionally, since WTs can affect aerial habitats for birds or trigger avoidance behaviors (Katzner et al., 2025), this may negatively impact habitat connectivity along migratory routes. To mitigate this, we recommend implementing habitat restoration projects, such as "retiring fishing for wetland restoration," in areas significantly impacted by WTs. These initiatives would create additional high-tide roosts that support bird habitat functions (Fan et al., 2021; Guo et al., 2025), thereby enhancing habitat connectivity. In summary, developing innovative and environmentally friendly renewable energy strategies is essential for achieving a balance between renewable energy development and biodiversity conservation, and for strengthening the synergies among the Sustainable Development Goals (Xing and Wang, 2021; Xu and Peng, 2024; Zhang et al., 2024).

4.3. Limitations

There are several limitations in our study. First, the bird observation data used in this study come from the CBR dataset, which is a citizen science dataset composed of user-reported data and may be affected by potential sampling issues and biases. The dataset we used contains over 6.7 million observation records, and large sample sizes help reduce bias and improve accuracy (Dickinson et al., 2010). Additionally, the CBR data platform has strict data quality control standards, ensuring a certain level of data accuracy (Gu et al., 2024). Furthermore, to ensure robust statistical analysis, we excluded bird species with fewer than 50 observation records. However, this approach may have disproportionately affected threatened species, as they are often less frequently observed due to their rarity or specialized habitat requirements. As a result, the responses of these species to WT distribution may be underrepresented, and the patterns observed in our study should be interpreted with caution. Future research should conduct more detailed studies on threatened bird species, such as tracking changes in migration routes under the influence of WTs, to verify the responses of these species and provide a more comprehensive understanding of the impacts of WTs on biodiversity.

Second, we assumed that the impact of WTs on bird richness is spatially independent, but the impact of WTs may have spillover effects, causing bird richness to exhibit spatial non-independence (Meng et al., 2025). Additionally, some studies have indicated that the impact of WFs on birds decreases over time (Dohm et al., 2019; Farfán et al., 2017). This study overlooked these effects, which may introduce uncertainty into the results. To address these challenges, future research should consider integrating multi-source datasets to construct continuous time series of bird observations and WT development. A Before-After-Control-Impact design may be adopted to systematically evaluate both the long-term and lagged effects of WTs on biodiversity. In addition, spatial regression models should be employed to more accurately account for spatial autocorrelation in WT effects. Together, these approaches are expected to offer deeper insights into the mechanisms by which WTs influence diversity.

We evaluated the impacts of WTs on bird richnes at the national scale. However, we contend that the mechanisms by which WTs affect birds represent a critical yet relatively underexplored area of research. Given the strong variability in birds' behavioral responses to WTs (Katzner et al., 2025), accurately understanding these mechanisms is essential for establishing the theoretical foundation of future risk assessments and model development. Furthermore, a study in India by Thaker et al. (2018) revealed that wind farm construction led to a decline in bird density, effectively creating predator-free environments. This, in turn, triggered cascading responses across lower trophic levels, resulting in changes to regional biodiversity and ecosystem stability. Such localized disturbances may even generate broader ecological consequences through telecoupled effects on distant ecosystems (Liu, 2017). In other words, the sustained influence of WTs may gradually transform ecosystems both within and beyond their boundaries, potentially driving them toward new ecological states or regimes. Future research should prioritize the exploration of these cascading effects and work toward developing a comprehensive analytical framework grounded in systems thinking to better understand and capture the complexity of such dynamics.

5. Conclusions

This study tested the differences in bird richness between counties with and without WTs, and further characterized the changes in richness under increasing WT impact intensity. The results show that the richness of all bird, resident and migratory species was significantly lower in counties with WTs than in those without. Moreover, we observed that bird species richness exhibited a steep decline when the impact intensity exceeded 0.2, and the degree of impact varied among bird groups with different ecological traits. This study contributes to large-scale assessments of the effects of WTs on bird species richness, enhancing understanding of the ecological and environmental consequences of wind power development. It also provides important scientific evidence to inform wind farm site selection and policy formulation, thereby supporting the sustainable development of wind energy. Moreover, the analytical framework proposed in this study is adaptable and can be applied to future research on the impacts of wind power or photovoltaic power stations on biodiversity.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.eiar.2025.108212.

Data availability

Data will be made available on request.

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