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Optimizing key patches can effectively mitigate the negative impacts of wind turbine deployment on the functional connectivity of waterbirds

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ABSTRACT

With the rapid development of the wind power industry, the negative impacts of wind turbines (WTs) on birds have drawn increasing attention. While previous research has explored the effects of WTs on birds, few have examined the issue of functional connectivity or how to mitigate the impacts of existing WTs on bird habitats. To address this, this study selected the coastal region of the Yellow and Bohai Sea in China as the study area, which is both a key habitat for waterbirds in China and a concentrated area for WT deployment. The study first compared and analyzed the differences in functional connectivity of waterbird habitat networks along the Yellow and Bohai Sea coasts, both with and without WT deployment. Results show a 21.4% decline in functional connectivity due to WTs. The study also introduces a stepping stone site selection framework to identify key habitat patches that improve connectivity. After incorporating these stepping stones, the functional connectivity increased by 23.2% relative to the current WT scenario, reaching a level comparable to the no-WT scenario. These findings suggest that optimizing key habitat patches can effectively mitigate the negative impacts of WTs on habitat functional connectivity.

1. Introduction

Wind energy, as a critical renewable energy source, is a central driver of energy system decarbonization (Pryor et al., 2018, 2020) and plays a significant role in optimizing China's energy structure and achieving carbon neutrality goals (Liu, Wang, et al., 2022). In recent years, China's wind energy sector has experienced significant growth and aims to increase the installed capacity of wind and other renewable energy sources to over 1200 GW by 2030 (Liu, Deng, et al., 2022). However, the development of wind farms (WFs) also presents ecological challenges, including potential significant impacts on bird diversity (Dang et al., 2026; Meng et al., 2025; Miao et al., 2019), local and regional climate (Dang et al., 2025; Li et al., 2018), and vegetation and ecosystem dynamics (Su et al., 2024; Thaker et al., 2018). Therefore, understanding and mitigating the impacts of WFs is crucial for scientifically planning renewable resource development and ensuring its sustainable growth (Su et al., 2026; Xia et al., 2024; Yang, Zhang, et al., 2024).

Functional connectivity of waterbird habitat networks is vital for birds. Well-connected habitats provide more complex ecological functions, such as breeding opportunities and lower predation pressure. In contrast, reduced connectivity may hinder population migration and dispersal between habitats, affecting the exchange of matter, energy, and genes, and preventing the support of stable bird populations (Huang et al., 2025; Zhang et al., 2023). For example, Xu et al. (2019) identified a significant correlation between decreased functional connectivity and changes in bird population size. Consequently, preserving habitat functional connectivity is critical for regional bird conservation and has significant implications for maintaining bird diversity at larger scales (Studds et al., 2017; Zhang et al., 2023).

However, the construction of wind turbines (WTs) may have a negative impact on bird habitats and their connectivity. For instance, low-frequency noise and visual disturbance from WTs can affect aerial habitats, prompting some sensitive species to move from highly disturbed areas to less disturbed ones (Lehnardt et al., 2024). WTs can

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also lead to a decrease in survival within a certain range (Carrete et al., 2012; LeBeau et al., 2014), and their wake effects may drive away birds (Leroux et al., 2024; Skov et al., 2026). Studies have shown a significant decline in bird populations within a defined radius from WTs (Drewitt & Langston, 2006; LeBeau et al., 2017; Marques et al., 2020), indicating that some high-quality bird habitats are abandoned and resulting in functional habitat loss (Reusch et al., 2023). Furthermore, WFs can create a barrier effect, causing birds to alter their migratory flight paths to avoid WFs (Skov et al., 2026), disrupting connections between feeding and roosting areas (Drewitt & Langston, 2006; Watson et al., 2025). Even when birds are able to bypass WT areas, such path alterations may increase flight distance and energy expenditure (Schwemmer et al., 2023), reduce movement efficiency among habitat patches, and ultimately lead to a decline in habitat connectivity.

In this context, Katzner et al. (2025) emphasized the importance of scientific site selection, such as avoiding protected areas, high-value habitats, migratory routes, and key locations for species sensitive to WTs (including breeding grounds, wintering habitats, and habitat corridors), to mitigate the negative impacts of WTs on birds. A more pressing issue is that WTs have already been extensively deployed in coastal areas, highlighting the need to explore strategies for mitigating their impact on bird habitat connectivity within existing WF layouts. Some researchers, guided by the concept of ecological protection networks (Hilty et al., 2020), aim to optimize habitat connectivity by identifying regional ecological sources and corridors, managing ecological hotspots and barriers, or constructing stepping stones (Huang et al., 2025; Wang, Zhou, et al., 2022; Xu et al., 2024). Yet, research on improving habitat connectivity under the impact of WTs using relevant theories and methods remains limited.

While previous studies have provided valuable insights into the effects of WTs on birds, the importance of habitat connectivity as a crucial factor in bird conservation has often been neglected. The extent to which WT construction affects habitat connectivity remains unclear. Additionally, how to restore or mitigate the effects of existing WFs on birds requires further exploration. To this end, this study focuses on key waterbird habitats in the coastal region of the Yellow and Bohai Sea, an area also heavily concentrated with WT deployment in China. The study aims to analyze the impact of WTs on the connectivity of waterbird habitats, propose a stepping stone site selection framework, and assess the benefits of optimizing key habitat patches for regional habitat connectivity. This research will contribute to a deeper understanding of the ecological impacts of wind energy development and provide a scientific basis for the formulation of future wind energy policies.

2. Materials and methods

2.1. Study area

The East Asian-Australasian Flyway (EAAF) supports the migration of approximately 50 million birds annually (Tang et al., 2023). The coastal region of the Yellow and Bohai Sea in China, part of the EAAF, hosts a diverse range of wetland ecosystems, including tidal flats, estuarine deltas, aquaculture zones, and salt pans (Yang, Duan, & Yu, 2024) (Fig. 1b). These wetlands provide essential stopover and feeding sites for waterbirds, making the region a key component of the EAAF. Several waterbird species dependent on the Yellow and Bohai Sea wetlands meet the global 1% population threshold for conservation (Bai et al., 2015) and have been designated as a UNESCO World Natural Heritage site (Ma et al., 2023).

The Yellow and Bohai Sea coastline spans several provinces from north to south, including Liaoning, Hebei, Tianjin, Shandong, and Jiangsu. This region includes 20 coastal cities and is one of the most economically developed and densely populated areas in China. On the other hand, due to its well-developed and stable wind energy resources (Sun et al., 2024; Zhang et al., 2024), the Yellow and Bohai Sea coast is one of the earliest and most densely concentrated areas for WT deployment in China (He et al., 2025; Yang et al., 2025). In related provinces such as Hebei, Shandong, Liaoning, and Jiangsu, the ranking for installed capacity is 3rd, 6th, 8th, and 13th, respectively (He et al., 2025). This study focuses on a 50-km inland extension from the Yellow and Bohai Sea coastline (Yang, Duan, & Yu, 2024), covering a total area of about 132,000 km^2. By 2020, the total number of WTs in the

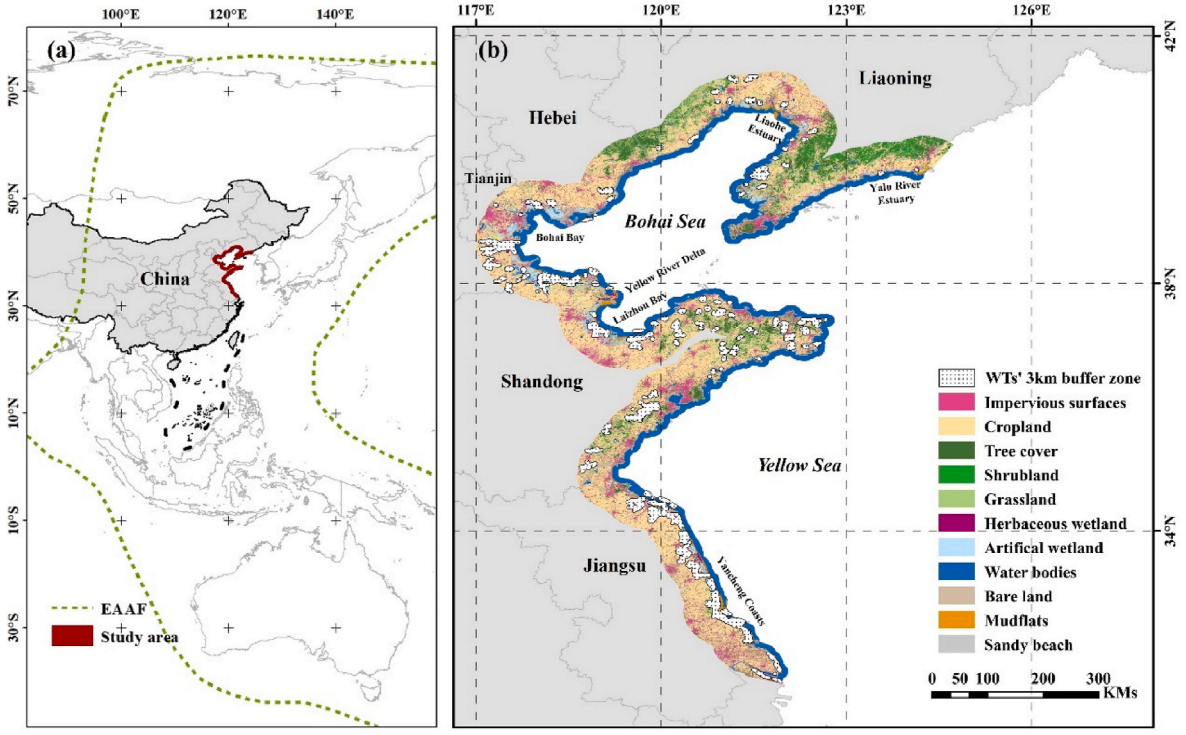

Fig. 1. Study area.

study area had reached 8579 (Fig. S1).

2.2. Data sources

The data used in this study are shown in Table S1. The WT distribution data were obtained from an interpreted dataset by the research team, which is based on Open Street Map data and Sentinel imagery. This dataset interprets the temporal and spatial distribution of onshore and offshore WTs in China from 2014, with a spatial accuracy of over 85%, meeting the needs of the study (Lyu et al., 2025). The WT distribution data for 2020 were selected from this dataset, totaling more than 140,000 WTs. This number is comparable to the number of WTs identified in other studies (He et al., 2025; Yang et al., 2025a; Yang et al., 2025b).

Land use data were sourced from Li et al. (2022), who interpreted 10-m resolution land cover data for coastal China. This data includes a buffer zone extending 10 km inland from the coastline and 20 km seaward. To address this, the study incorporated the global 10-m resolution LUCC data, GLC_FCS10 (Zhang, Liu, et al., 2025). After integrating the classification systems, the datasets were merged, resampled to a 100-m spatial resolution, and clipped to obtain the land use data for the study area.

The bird species list used in this study is primarily derived from the China Bird List v8.0 (2020) (China Bird Report, 2020) and the Checklist of Birds of China (Zheng, 2017), with additional selection of coastal waterbirds based on relevant studies (Bai et al., 2015; Huang et al., 2025; Xia et al., 2017). The study only retained bird species observed in the research area between 2015 and 2020 as the bird species list for this study (Dang et al., 2025). The bird observation records were primarily sourced from the China Birdwatching Reporting Center (http://www.birdreport.cn/).

2.3. Overview of the methodology

To assess the impact of WTs on habitat connectivity and propose mitigation strategies, this study conducted two main tasks:

(1) Analysis of functional connectivity differences between scenarios with and without WT deployment. The study simulated the habitats of four typical waterbird groups (shorebirds, wading birds, open-water waterbirds, and waterfowl) along the Yellow and Bohai Sea coast using species distribution models (SDMs) and compared the differences in habitat connectivity under the presence and absence of WTs.
(2) Site selection optimization and benefit assessment. The study developed a stepping stone site selection framework that identified patches as stepping stones based on spatial overlay analysis of ecological corridors, potential waterbird habitats, and artificial wetlands. The optimization benefits were then evaluated by comparing habitat connectivity across three scenarios: no-WTs, actual WT deployment, and with stepping stones.

2.4. Methods

2.4.1. Identification of bird habitats

Based on relevant studies (Wang, Chen, et al., 2022; Xia et al., 2017), this study covers 103 species of waterbirds. Given the differences in feeding habits, body size, migratory characteristics, and habitat preferences, waterbirds were divided into four groups: shorebirds, wading birds, open-water waterbirds, and waterfowl, representing different ecological niches (Huang et al., 2025). Shorebirds (43 species), are characterized by their preference for edge areas of inland or coastal waters. Wading birds (16 species), are generally larger in size and prefer grasslands, tidal flats, and deeper wetlands as foraging and nesting habitats. Open-water waterbirds (13 species), including the family Laridae, are mostly marine birds with strong flying abilities. They typically breed near open waters and migrate over long distances (Li et al., 2024). Waterfowl (31 species), primarily including ducks and geese, prefer reed beds near water bodies as foraging sites (Lin et al., 2022).

The MaxEnt algorithm was used to identify bird habitats, with 75% of the occurrence points used as training data and the remaining points as test data. Following relevant studies (Dang et al., 2026; Sun et al., 2022; Yang, Duan, & Yu, 2024), the environmental variables used to fit SDMs included 19 bioclimatic variables from WorldClim2 (bio1 to bio19) (Fick & Hijmans, 2017), land use data, and distance to the coastline, totaling 21 environmental variables. Land use data were encoded using one-hot encoding. To ensure the accuracy of model predictions and avoid overfitting due to multicollinearity among independent variables, a correlation analysis was conducted, and variables with correlations greater than 0.8 were excluded (Fig. S2). In the end, 16 variables were retained (9 land use factors, along with bio2, bio3, bio4, bio8, bio13, bio19, and distance to the coastline). The model results were validated using the area under the receiver operating characteristic curve (AUC). Additionally, each group was simulated five times, and the average AUC value of these five simulations was calculated. The AUC values for the models of the four bird groups ranged from 0.84 to 0.88, indicating good predictive accuracy. Based on this, a threshold of 0.75 was selected to binarize the average habitat suitability (Duan & Yu, 2024; Huang et al., 2020; Xie et al., 2025; Yang et al., 2024), and habitat patches larger than 1 km^2 were selected for further analysis (Xu et al., 2024).

Several studies have identified the impact range of WTs on bird activity, but no unified conclusion has been reached. In different study areas or for different bird species, the impact range may vary from a few hundred meters to 5 km (Drewitt & Langston, 2006; Gómez-Catasús et al., 2018; LeBeau et al., 2017; Pearse et al., 2021; Tolvanen et al., 2023). In this study, a 3 km radius around WTs was considered unsuitable for bird habitats and was excluded from the habitat analysis.

2.4.2. Quantifying functional connectivity of waterbird habitat networks

Functional connectivity in the study area is measured using the equivalent connected area (ECA) (Xu et al., 2019), with larger ECA values indicating better connectivity:

$$P_{ij} = e^{-kd_{ij}}$$

$$ECA = \sqrt{\sum_{i=1}^{n}\sum_{j=1}^{n} a_i a_j P_{ij}^{*}}$$

Here, P_{ij} denotes the dispersal probability between habitat patches i and j, and d_{ij} is the closest distance between the two patches. k is a constant defined so that dispersal probability is 50% when d_{ij} equals the median of the mean flight distance of the waterbirds, with the relevant data derived from Johnson et al. (2014). The maximum threshold was set to 32.5 km, which corresponds to the maximum foraging flight distance of the primary waterbird species (Johnson et al., 2014). In the ECA equation, a_i and a_j represent the area (km^2) of habitat patches i and j, respectively. P_{ij}^{*} represents the maximum product probability between patches i and j, corresponding to the most effective path in the habitat network when all possible direct and multi-step connections are considered. When $P_{ij}^{*} = P_{ij}$, the distance between patches i and j is close enough for individuals to migrate directly. When $P_{ij}^{*} > P_{ij}$, the optimal path consists of multiple routes, involving several intermediary patches between patches i and j.

The connectivity value can be decomposed (contribution, %) into internal, direct, and indirect parts, which represent the contributions of the habitat interior, direct connections, and the use of intermediary patches to connect different habitats, respectively (Saura et al., 2014). Specifically, internal connectivity refers to the proportion of the area within a habitat that is accessible to birds. Direct connectivity refers to the amount of connectivity between habitats without the use of other

habitats. Indirect connectivity refers to the increased connectivity resulting from the use of other habitats to enhance probabilities. Additionally, the study used dPC to measure the importance of each habitat in maintaining connectivity. The calculations for the above indices were performed using Conefor 1.1.6 (Saura & Torné, 2009).

2.4.3. Identification of ecological corridors

The habitats of the four bird groups were overlaid, and patches supporting three or more groups of waterbirds were designated as ecological sources. In constructing the resistance surface, factors such as land use, NDVI, slope, and distance from roads were considered as resistance factors, following relevant studies (Huang et al., 2025). Additionally, due to the barrier effect of WFs, which cause birds to alter their flight paths to avoid WTs (Drewitt & Langston, 2006), the distance from WTs was incorporated into the adjustment of the ecological resistance surface. Then, following relevant studies (Huang et al., 2025), resistance values were assigned to each factor, and the analytic hierarchy process was used to determine the weights of these factors, thus creating a multi-layered resistance surface evaluation system. Higher resistance values indicate greater obstacles to the migration of waterbirds. The resistance surface visually illustrates the migration patterns of waterbirds within habitats under the combined influence of multiple factors, with detailed information provided in Table S2 and Fig. S3. Based on this, ecological corridors were identified using the Linkage Mapper tool.

2.4.4. Stepping stone site selection framework

The study proposes a coastal waterbird habitat functional connectivity optimization framework, which integrates ecological corridors, SDMs, and artificial wetland distribution to identify key habitat patches as stepping stones (Fig. 2). Specifically, ecological corridors, as linear landscape features connecting different ecological sources, facilitate material cycling, energy flow, and information transfer, such as species migration and ecosystem service flow (Peng, Xu, et al., 2025). Therefore, selecting patches that intersect with ecological corridors ensures that the identified stepping stones effectively improve habitat connectivity (Huang et al., 2021; Xu et al., 2024).

Secondly, a threshold of 0.65 was further applied to delineate potential habitat. Frequency analysis showed that values around 0.65 represented a high-frequency interval in the SDM results, reflecting the dominant suitability class in the model output (Fig. S4). Accordingly, 0.65 was used as the threshold to identify potential waterbird habitat. Although areas within this range did not meet the criterion for actual habitat, they were considered to have greater potential to support waterbird habitation than other low-suitability areas. At the same time, areas within a 3-km radius of WTs were excluded, because ecological restoration in these areas could turn WFs into ecological traps, attracting birds into hazardous areas and increasing mortality risk (Peste et al., 2015).

Additionally, previous studies have shown that restoration or management measures can effectively enhance the bird habitat function of artificial wetlands, with the bird diversity supported by artificial wetlands approaching that of natural wetlands (Fan et al., 2021; Guo et al., 2025; Shimizu et al., 2025). The implementation of projects such as fish farm removal and topographic transformation in artificial wetlands has become one of the key strategies for coastal bird habitat conservation (Liu & Ma, 2024). Therefore, by combining SDM thresholds with artificial wetland distribution, the study ensures that the selected patches have high potential for supporting bird habitats.

3. Results

3.1. Changes in the functional connectivity of waterbird habitats

As shown in Fig. 3, the habitats of the four waterbird groups are primarily distributed within 5 km of the coastline, with concentrations in the Bohai Bay, Yellow River Delta, Laizhou Bay, and the Yancheng coastline in Jiangsu Province (Fig. 3a). The land use types of these habitats mainly include impervious surfaces, bare land, natural wetlands, water bodies, and artificial wetlands. All four groups of waterbird habitats are impacted by WTs, particularly in the Bohai Bay (Fig. 3b) and Yancheng coastline (Fig. 3c) regions. These areas are not only crucial for waterbird habitats but also host the highest concentration of WTs. Under the assumed 3-km WT impact buffer, the total area of the four waterbird habitats decreased by an average of 13.5%, and the ECA of the four waterbird habitats decreased by an average of 21.4%.

The impact of WTs on different groups of waterbirds varied, with open-water waterbirds being the most affected (32.7%) (Fig. 4a). The habitat characteristics derived from SDMs support this observation, with open-water waterbirds having the largest average habitat area among the four groups, reaching 49.3 km^2, and being primarily distributed in the contiguous artificial wetland areas of Bohai Bay, the Yellow River Delta, and the Yancheng coastline (Fig. S5b). However, these areas are also heavily affected by WTs, with 14.5% of their habitat area falling within the WT impact radius. This results in increased habitat fragmentation for open-water waterbirds, with the average habitat area decreasing by 13.6%, second only to shorebirds. On the other hand, the affected habitats are mostly located in regions with high dPC values

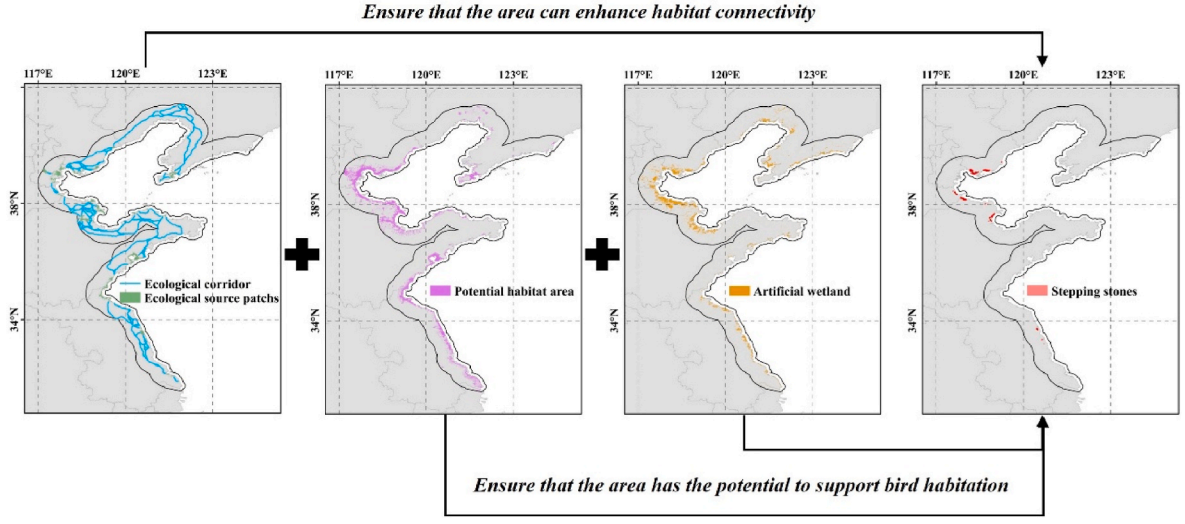

Fig. 2. Stepping stone site selection framework.

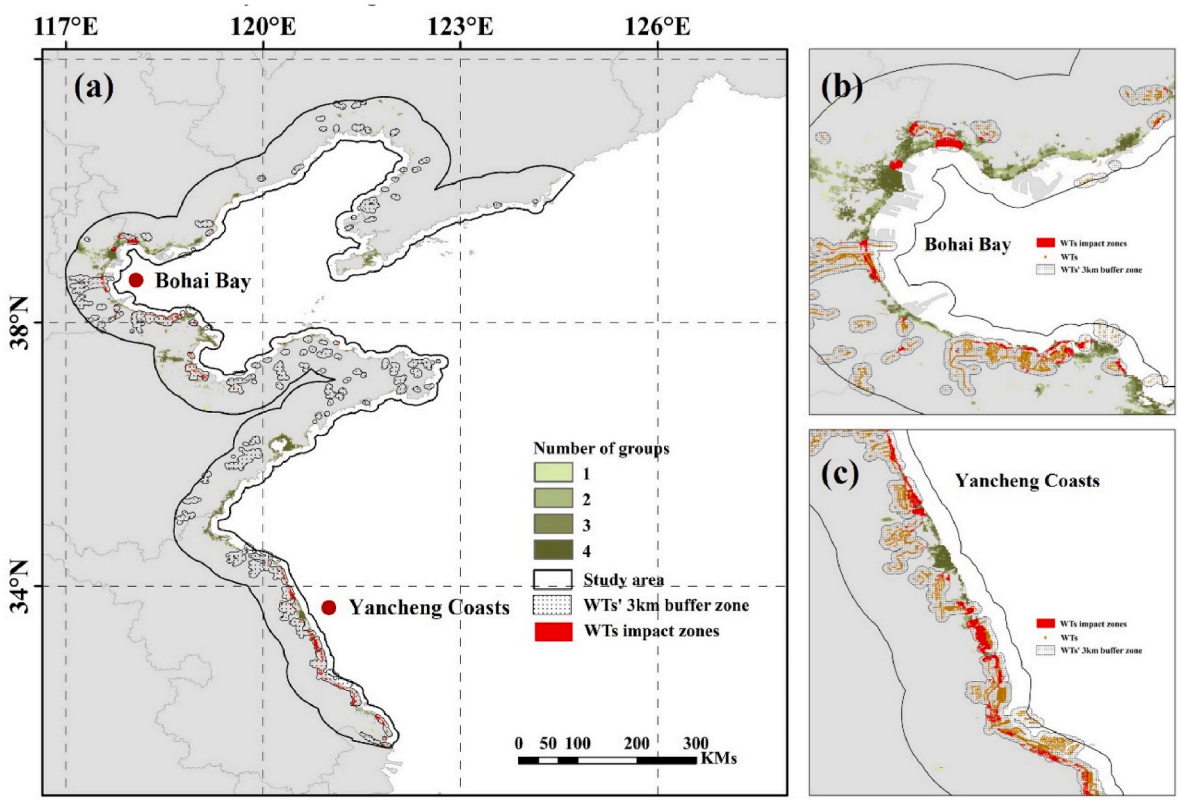

Fig. 3. The distribution of waterbird habitats, with numbers indicating the number of groups supported by each habitat.

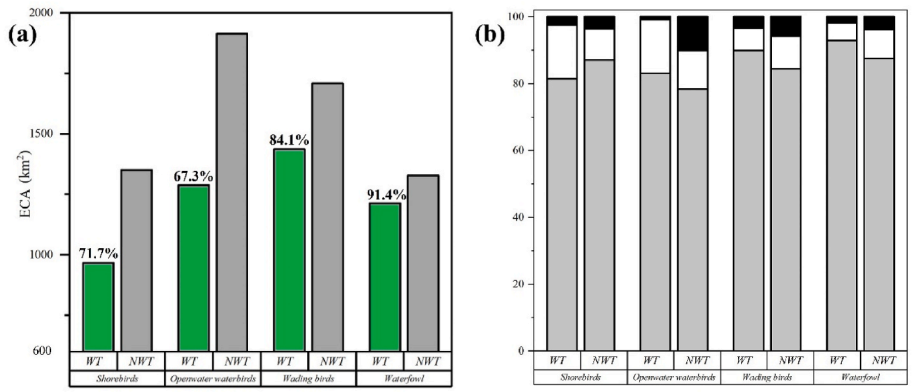

Fig. 4. Functional connectivity of waterbird habitat networks. (a) Average equivalent connected area for four waterbird groups with and without WT deployment. (b) Contribution of fractions showing intrapatch (light gray), direct (white), and step (black) connections as percentages of functional connectivity with and without WT deployment.

(Fig. S5b), indicating that these areas contribute significantly to connectivity, making the overall connectivity impact more pronounced. The impact on waterfowl is the smallest (8.6%) (Fig. 4a), primarily due to their higher adaptability to habitat changes and broader distribution. Furthermore, the habitat area of waterfowl affected by WTs is the smallest among the four groups (9.5%), and the impact on habitats with high dPC values is relatively small, resulting in the least overall impact on connectivity (Fig. S5d).

Among the four waterbird groups, internal connectivity contributes more than 78.4% to the total connectivity, followed by direct connectivity, with indirect connectivity contributing the least (Fig. 4b). Compared to connectivity without WT deployment, indirect connectivity for all four waterbird groups decreased under the influence of WTs. Notably, for open-water waterbird habitats, the proportion of indirect connectivity dropped from 10.0% to 0.8%.

3.2. Functional connectivity under different scenarios

The ecological sources, ecological corridors, and identified stepping stones within the study area are shown in Fig. 5a. After excluding the WT impacted areas (503 km^2), the remaining ecological sources consist of 89 patches, with a total area of 3356 km^2. The total area of the stepping stones, identified through the combination of potential waterbird habitats, artificial wetlands, and ecological corridors, is 436 km^2, primarily distributed in the Bohai Bay (Fig. 5b) and the Yancheng coastline (Fig. 5c).

The results of connectivity under three scenarios are presented in Fig. 6a. Specifically, under the no-WTs scenario, the total area and ECA

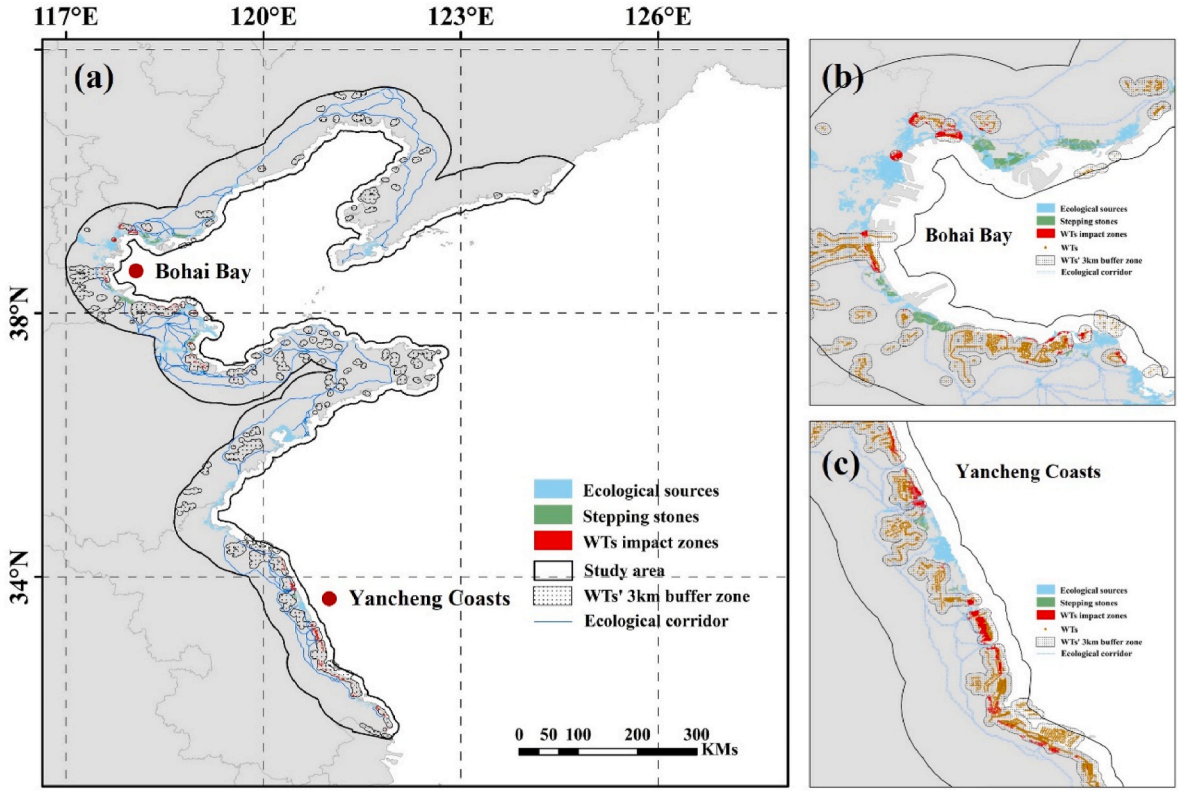

Fig. 5. The distribution of ecological sources, ecological corridors, and the selected stepping stones.

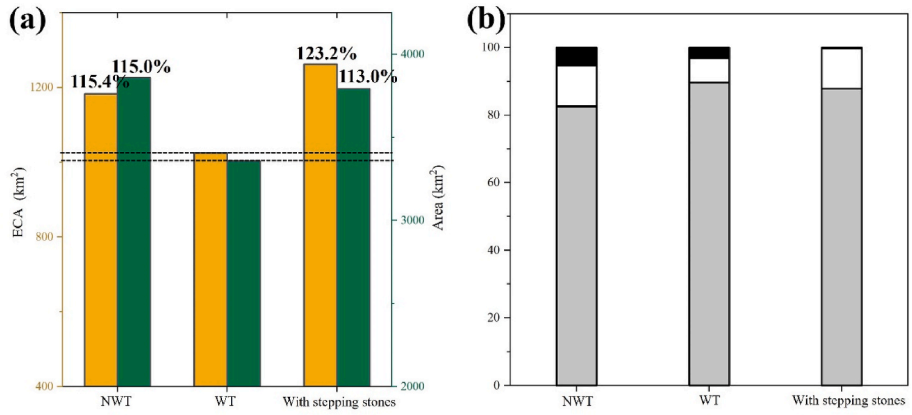

Fig. 6. (a) ECA and total area under three scenarios: no-WTs (NWT), actual WT deployment (WT), and with stepping stones. (b) Contribution of fractions showing intrapatch (light gray), direct (white), and step (black) connections as percentages of functional connectivity under three scenarios.

are 115.0% and 115.4% of the actual WT deployment scenario, respectively. After incorporating stepping stones, the total area and ECA increased by 13.0% and 23.2%, respectively. In other words, although the addition of stepping stones did not fully compensate for the area of ecological sources impacted by WTs, it significantly enhanced the connectivity.

Under the actual WT deployment scenario, the proportions of direct and indirect connectivity decreased compared to the no-WT scenario (Fig. 6b). In contrast, direct connectivity increased in the stepping stone scenario. This increase was mainly because the selected stepping stones were located along ecological corridors and therefore occupied key positions between fragmented ecological sources. By reconnecting these fragmented ecological sources, the selected stepping stones shortened effective distances, strengthened direct dispersal links, and integrated separate ecological sources into larger, more cohesive habitat units. In some areas, they may also provide stopover habitats, further facilitating waterbird movement across the habitat network.

4. Discussion

Accurately understanding and mitigating the impact of WTs on habitat functional connectivity for birds is not only a crucial foundation for protecting regional bird biodiversity but also provides essential insights for the development of related policies, thereby enhancing the sustainability of wind energy utilization (Agha et al., 2020; McElwee et al., 2025). Existing research primarily focuses on the effects of WTs on birds, including aspects such as bird behavior and diversity (Dohm et al., 2019; Meng et al., 2025; Pearse et al., 2021; Watson et al., 2025). However, research on how to mitigate these impacts is relatively limited. The primary approach in the relevant studies is to reduce

the potential negative effects of WTs on birds through the scientific siting of WFs (Katzner et al., 2025; McElwee et al., 2025). To address this issue, this study first compares and analyzes the differences in functional connectivity of waterbird habitat networks along the Yellow and Bohai Sea coastlines under scenarios with and without WT deployment. Next, WT impact sources are integrated into a resistance surface, and, based on this, key habitat patches are identified as stepping stones by combining ecological corridors, SDMs, and artificial wetland distributions. The results show that optimizing certain habitat patches can effectively mitigate the decline in habitat connectivity caused by WTs in the region. Furthermore, the proposed stepping stone site selection framework is potentially transferable and could be applied to improve habitat connectivity in future WF or photovoltaic power plant projects.

4.1. Mitigating connectivity loss through strategic stepping stone selection

Ecological corridors represent low-resistance pathways that easily connect adjacent ecological sources and are most conducive to species dispersal (Xie et al., 2025; Xu et al., 2024). Therefore, by extracting stepping stones based on the ecological corridor range, dispersal links between ecological sources and stepping stones can be ensured, thus enhancing functional connectivity. This framework may have broader applicability to other renewable energy infrastructure. Research has indicated that not only the construction of WFs but also the deployment of photovoltaic power plants can negatively affect wildlife connectivity. For instance, Leskova et al. (2022) found that the construction of photovoltaic power plants in Florida posed an additional threat to the endangered Florida panther's habitat and migration corridors. While some scholars have suggested that vegetation management around photovoltaic power plants could create habitats for pollinators, such as bees and butterflies, thereby improving habitat connectivity and even increasing crop yields (Hernandez et al., 2019; Walston et al., 2018), from a cost-efficiency perspective, this solution may be less suitable for protecting the connectivity of large animal and bird habitats. Therefore, scientific site selection and subsequent ecological engineering measures are necessary for effective habitat connectivity conservation.

4.2. Management implications for bird conservation in coastal WFs

Scientific restoration should be implemented for the identified stepping stone areas to effectively enhance their function as bird habitats. Drawing from the experiences of existing coastal wetland ecological restoration projects, and incorporating nature-based solutions, the restoration measures should include topographical modifications, vegetation restoration, fish and benthic habitat recovery, and hydrological regulation (Liu et al., 2026; Peng, Zhu, et al., 2025; Shimizu et al., 2025). Restoration design should account for the ecological requirements and behavioral traits of potential target bird species, creating diverse waterbird habitats that meet the needs of different bird groups.

Katzner et al. (2025) highlight that WFs may impact birds both directly and indirectly in various ways. Among these, bird collision events are the most direct impact of WTs on birds. To reduce the likelihood of bird collisions, a "monitoring + management" strategy can be implemented. Specifically, an integrated monitoring platform should be established to accurately track bird density within WF areas, and laser bird deterrent devices can be installed to reduce bird activity near WTs and minimize collision risks. If bird abundance in the WF area exceeds a certain threshold, temporary WT shutdowns should be considered (Zhang, Wan, et al., 2025).

In addition, to reconcile the conflict between coastal WF development and bird habitat connectivity, a government-led policy framework should be established to include constraints and compensation mechanisms. The current project-based environmental impact assessment model may not fully consider the impact on regional habitat connectivity. Therefore, habitat connectivity indicators should be introduced to assess the impact of WF layouts on the ecological networks. For WF projects that, after scientific evaluation, are deemed necessary but still cause habitat losses and impacts, the developers must take responsibility for ecological restoration. A government-led ecological compensation mechanism should be established (Agha et al., 2020), with compensation funds sourced from compensation fees paid by companies and dedicated to the construction and maintenance of the identified key optimized patches.

4.3. Limitations and future research directions

There are certain uncertainties in the study. Firstly, we assume that the impact range of WTs on bird habitats is 3 km. However, the influence radius of WTs may vary across different regions and bird species (Drewitt & Langston, 2006; Tolvanen et al., 2023), which could lead to either an overestimation or underestimation of the impact of WTs on habitat connectivity. Another source of uncertainty lies in the spatio-temporal coupling between wind energy resources and bird activity, particularly whether the seasonal activity patterns of birds coincide, in both time and space, with periods of highly efficient WTs operation. This calls for more detailed research on the effects of WTs on bird behavior in future studies. Studies have indicated that changes in functional connectivity can significantly affect bird population sizes (Xu et al., 2019). In the case of the Yellow Sea and Bohai Sea coasts, our research focused only on the changes in habitat connectivity caused by WTs. Future studies should further analyze the relationship between these connectivity changes and bird population sizes in the region. Additionally, research has pointed out that offshore WFs can also create barrier effects for waterbirds, causing them to alter their flight paths to avoid WFs. This increases energy expenditure and disrupts migration or foraging routes. Some waterbird species have reduced or abandoned their foraging and breeding sites near WFs after they were operational, leading to a decline in local populations (Watson et al., 2025). Therefore, an important direction for future research is to develop an optimization framework that simultaneously improves habitat connectivity for both coastal and offshore waterbirds.

5. Conclusion

In this study, we compared and analyzed the differences in the functional connectivity of waterbird habitat networks along the Yellow and Bohai Sea coasts with and without WT deployment. Using ecological corridors, SDMs, and the distribution of artificial wetlands, we developed a stepping stone site selection framework. The results indicate that WT deployment led to an average 21.4% decrease in the ECA of waterbird habitats, suggesting that the construction of coastal WTs over the years may be driving the reshaping of waterbird habitat patterns in the region. However, optimizing certain key habitat patches could effectively mitigate the decline in regional habitat connectivity. This study enhances the understanding of the ecological impacts of wind energy development and provides valuable insights for the formulation of renewable energy policies, promoting the sustainable development of renewable energy utilization.

CRediT authorship contribution statement

Dongliang Dang: Writing – original draft, Methodology, Formal analysis, Data curation, Conceptualization. **Xiaobing Li:** Writing – review & editing, Supervision, Funding acquisition. **Xin Lyu:** Writing – review & editing, Investigation. **Shiliang Liu:** Writing – review & editing, Conceptualization. **Wanyu Cao:** Writing – review & editing, Methodology. **Siyu Liu:** Investigation. **Ning Su:** Methodology. **Shiyi Xie:** Writing – review & editing. **Chenhao Zhang:** Investigation.

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

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