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Species–area relationships in human-modified landscapes: insights from urban and managed golf courses in Karnataka, India

Nag K. S. Chetan*, T. Prathusha and S. J. Amith

Centre for Urban Ecology, Bio-Diversity, Evolution and Climate Change (CUBEC) JAIN (Deemed-to-be UNIVERSITY) #319, 17th Cross, 25th Main, J P Nagar 6th Phase, Bengaluru Karnataka, India- 560078
**Corresponding author*: *ks.chetan@jainuniversity.ac.in*

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Abstract

Urban environments are increasingly recognized for their potential to contribute to both local- and landscape-scale biodiversity conservation; however, the ecological functioning of managed green spaces remains insufficiently understood. This study investigated whether the classical species–area relationship (SAR) persists in anthropogenically modified urban landscapes using two golf course ecosystems in Karnataka, India, as model systems: Zion Hills Golf County (ZHGC), representing a larger, less urbanized, and structurally heterogeneous landscape, and Bangalore Golf Club (BGC), representing a smaller, spatially constrained site embedded within a densely urbanized matrix. Faunal diversity was assessed using species richness, abundance, evenness, and the Shannon–Wiener and Simpson diversity indices. Statistical analyses included Welch's two-sample t-test, Pearson's chi-square test, Pearson correlation, and linear regression. ZHGC supported higher observed species richness (104 species) than BGC (59 species), consistent with differences in habitat area and ecological heterogeneity. Although differences in species abundance were not statistically significant (Welch's two-sample t-test, $P > 0.05$), strong and significant linear relationships between habitat area and species richness were observed for ZHGC ($n = 12$, $R^2 = 0.8956$, $F_{1,10} = 85.76$, $P = 3.20 \times 10^{-6}$) and BGC ($n = 12$, $R^2 = 0.9587$, $F_{1,10} = 231.9$, $P = 3.03 \times 10^{-8}$). Strong positive relationships between habitat area and species richness were observed at both sites. Although these managed ecosystems cannot replace natural forests, the findings suggest that larger, structurally complex urban green spaces can play an important role in maintaining biodiversity within metropolitan environments. The persistence of SAR patterns in these modified systems highlights the potential value of well-designed and ecologically managed urban green spaces as supplementary habitats supporting biodiversity in rapidly expanding cities.

Key words: Biodiversity conservation, spatial heterogeneity, species–area relationship, species richness, urban ecology

Introduction

Species–area relationships (SARs), also known as species–area curves, are fundamental scaling tools in biodiversity research (He et al., 2022). The species–area relationship describes the increase in species richness with increasing sampling area and is widely regarded as one of the fundamental ecological patterns (MacArthur and Wilson, 1967; Lomolino, 2000). As a powerful framework for interpolating and extrapolating biodiversity across spatial scales (He et al., 2022), SARs play a crucial role in advancing biogeographic theory and informing biodiversity conservation strategies (Matias et al., 2014).

Large habitats tend to accumulate more species from the surrounding species pool, a phenomenon explained by the passive sampling hypothesis (Connor and Simberloff, 1979). In addition, larger habitats often encompass a greater diversity of habitat types, thereby increasing environmental heterogeneity (MacArthur, 1965; Power, 1972). Together, these mechanisms are considered key drivers of species–area relationships (Deng et al., 2022). Consequently, SARs have broad practical applications, including estimating regional species richness, predicting biodiversity loss, identifying biodiversity hotspots, and informing the design and optimal size of protected areas (Simberloff and Abele, 1976; Bramson et al., 1996; Losos and Schluter, 2000).

Anthropogenic disturbance influences species richness across a wide range of ecosystems, including urban environments, by altering ecological processes and community structure (Yuan et al., 2014). Disturbance, as defined by Sousa (1984), includes any factor that modifies ecosystem structure, alters competitive interactions among species, or changes resource availability. Previous studies have demonstrated that the effects of disturbance on species richness vary among taxonomic groups. High levels of disturbance have been associated with reduced species richness in mammals (Fox and Fox, 2000), whereas intermediate levels of disturbance may support higher species richness in birds (Lepczyk et al., 2008) and insects (Jew et al., 2015). These contrasting responses suggest that taxonomic groups differ in their sensitivity to disturbance regimes. In addition, intermediate levels of disturbance have been proposed to support the highest levels of species diversity (Vera y Conde and Rocha, 2006). Furthermore, Tittensor et al. (2007) suggested that human disturbance may influence species richness either directly or through its interactions with spatial heterogeneity and habitat patchiness.

Urban habitats play a vital role in supporting biodiversity, including endemic species, yet their ecological importance has often been underestimated despite their significance at both local and global scales (Ives et al., 2016). Biodiversity surveys in urban landscapes are essential for guiding ecological restoration and the planning of green spaces (Kowarik, 2011). Such surveys provide valuable insights into community structure, population dynamics, and life-history traits, thereby supporting effective management and conservation of urban biodiversity. As urbanization continues to accelerate, integrating ecological knowledge into the planning and management of urban green spaces has become increasingly important (Lizée, 2012; Helden, 2004; Murgui, 2007; Ferenc et al., 2014).

Managed urban ecosystems, including golf courses, botanical gardens, and urban parks, provide important refuges for wildlife by supporting both aquatic and terrestrial habitats and can partially complement conventional protected areas (Franklin, 1993; Freeman, 1999; Colding et al., 2006). These urban green spaces support diverse assemblages of insects, birds, mammals, amphibians, and reptiles, thereby enhancing urban biodiversity.

Golf courses represent important urban green spaces that can contribute to biodiversity conservation. By incorporating a variety of habitats, including ponds, meadows, and woodlands, they support diverse plant and animal communities. Appropriate management practices, such as habitat restoration, sustainable water management, and the use of native plant species, can further enhance biodiversity at both local and landscape scales. Consequently, golf courses may provide valuable refuges and ecological corridors in urban areas where natural habitats are scarce or highly fragmented, thereby contributing to the ecological integrity and resilience of urban ecosystems (Colding and Folke, 2009).

The construction and management of golf courses often create a mosaic of habitat types, including wetlands, grasslands, and wooded areas (Colding and Folke, 2009; Forman and Godron, 1981). These habitats can support diverse wildlife despite occurring within human-dominated landscapes. Their ecological value is further enhanced by relatively low levels of human disturbance and low road density, which may reduce recreational impacts and wildlife mortality associated with road traffic (Colding and Folke, 2009; Guzy et al., 2013).

Despite numerous studies evaluating different groups of flora and fauna in urban ecosystems, species–area relationships (SARs) have rarely been investigated in golf course habitats. Recent studies have highlighted the importance of managed urban green spaces and golf courses in supporting biodiversity and providing valuable ecological functions within urban landscapes (Hu and Lima, 2024; Howell et al., 2024; Ormiston and Cristol, 2025; Howard et al., 2025). However, despite these advances, the applicability of species–area relationships (SARs) to managed golf course ecosystems remains poorly understood. To address this knowledge gap, we conducted a comparative ecological study using two golf course ecosystems in Karnataka (Figs. 1 and 2). One site, Bangalore Golf Club (BGC), is located within a highly urbanized environment surrounded by dense city infrastructure, whereas the other, Zion Hills Golf County (ZHGC), is situated adjacent to an emerging township experiencing comparatively lower urbanization pressure. The study quantified species richness, abundance, and evenness across the two managed urban ecosystems and characterized their faunal composition. Specifically, we examined whether species–area relationships were evident in managed golf course habitats and compared patterns of species richness and community composition between two ecosystems differing markedly in area, habitat heterogeneity, and degree of urbanization.

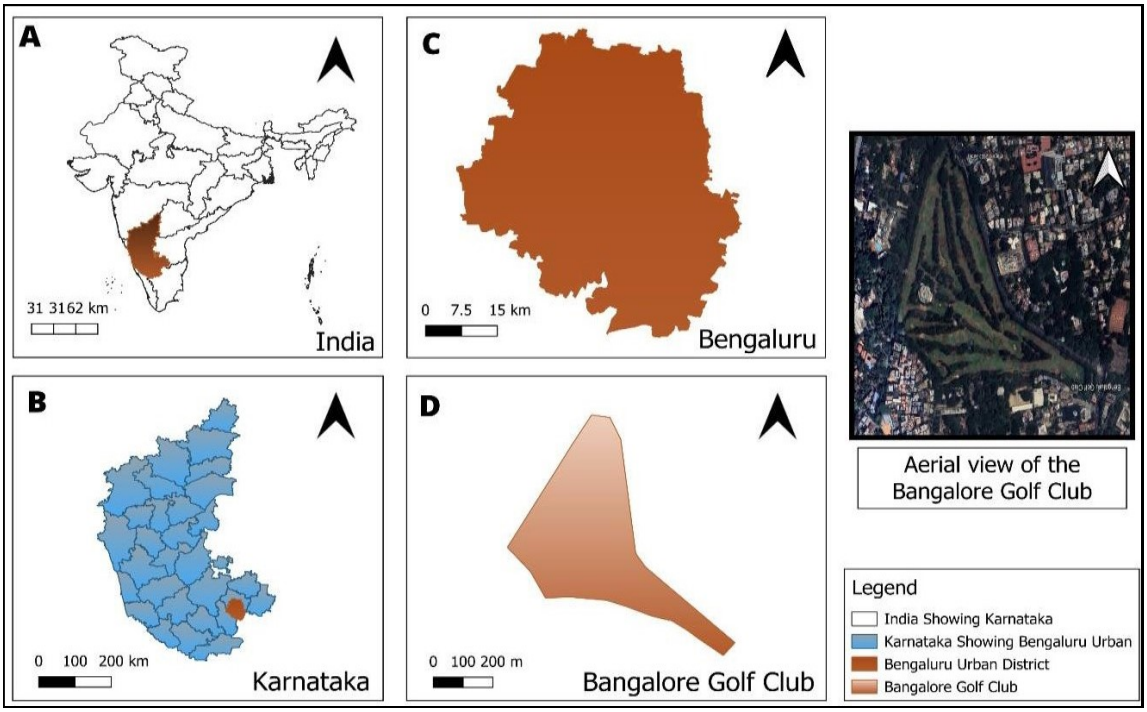

Figure 1: Geographic context of Bangalore Golf Club (BGC), Karnataka, India. The golf course covers approximately 60 acres. (A) Karnataka State within India, (B) Bengaluru Urban District within Karnataka, (C) Bengaluru Urban District, and (D) Bangalore Golf Club study site. Maps were prepared by the authors using QGIS version 3.40.3. Climatic data (long-term mean annual rainfall and temperature) are based on Rajashekar (2004).

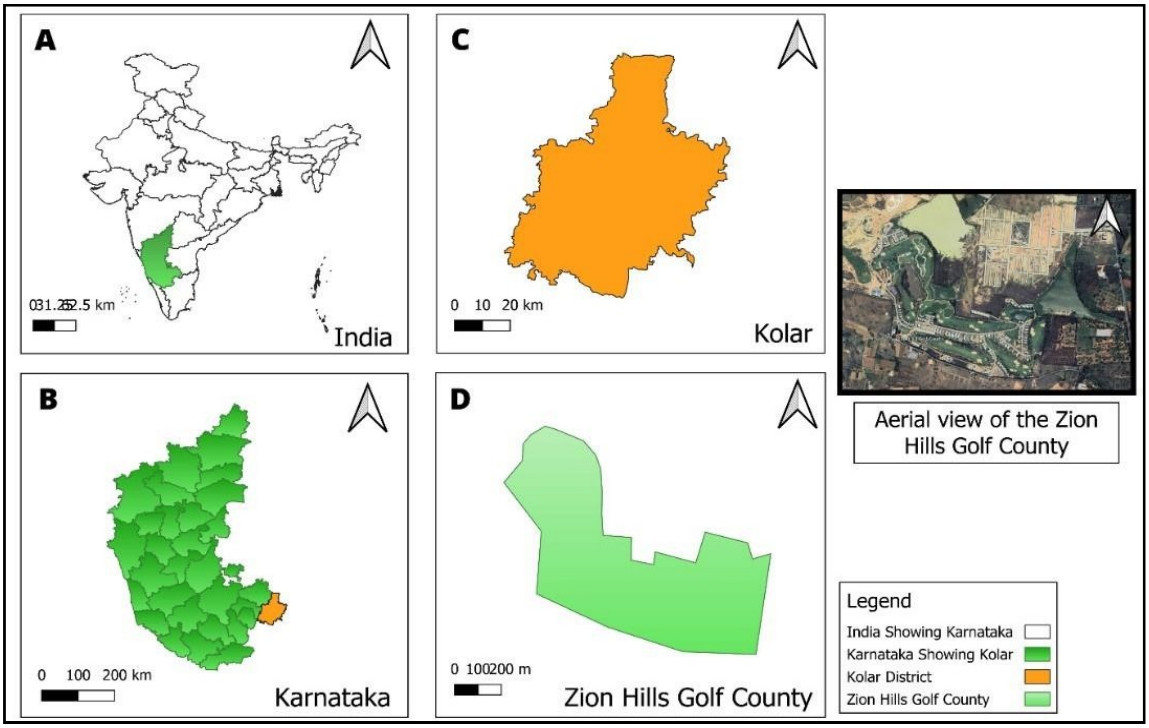

Figure 2: Geographic context of Zion Hills Golf County (ZHGC), Kolar District, Karnataka, India. The property covers approximately 250 acres, including approximately 120 acres of actively maintained golf course playing area used for biodiversity surveys and species–area relationship (SAR) analyses. (A) Karnataka State within India, (B) Kolar District within Karnataka, (C) Kolar District, and (D) Zion Hills Golf County study site. Maps were prepared by the authors using QGIS version 3.40.3. Climatic data (long-term mean annual rainfall and temperature) are based on Ahmed and Arpitha (2022) and Harishnaika et al. (2023).

Material and Methods

Study area

The study was conducted between 1 August and 30 September 2023 at two golf course ecosystems in Karnataka, India, which differed in size, landscape setting, and degree of urbanization.

Bangalore Golf Club (BGC)

Bangalore Golf Club (BGC) is located in Bengaluru, Karnataka, India (12.98633° N, 77.58846° E; 920 m above sea level [a.s.l.]). The golf course occupies approximately 60 acres and comprises planted tree stands, intensively managed turfgrass, and artificial water bodies incorporated into the course design. The site receives a long-term mean annual rainfall of approximately 1,500 mm and a long-term mean annual temperature of approximately 28°C (Rajashekar, 2004).

Zion Hills Golf County (ZHGC)

Zion Hills Golf County (ZHGC) is located in Kolar District, Karnataka, India (13.05261° N, 78.14135° E; approximately 920 m a.s.l.). The property covers approximately 250 acres, of which approximately 120 acres comprise the actively maintained golf course playing area. All biodiversity surveys and species–area relationship (SAR) analyses presented in this study were based on this 120-acre actively managed playing area, while the remaining area consists of associated infrastructure and non-playing landscape features. The landscape comprises managed turfgrass, established tree stands, and constructed water bodies embedded within a semi-urban environment. The site receives a long-term mean annual rainfall of approximately 748 mm and a long-term mean annual temperature of approximately 29°C (Ahmed and Arpitha, 2022; Harishnaika et al., 2023). Zion Hills Golf County is located approximately 77 km by road from Bangalore Golf Club, with the distance determined using Google Maps.

Study design and biodiversity survey

Three key predictor variables—habitat area, anthropogenic disturbance, and golf course characteristics—were evaluated using a combination of GIS analyses, field observations, and information obtained through discussions with golf course managers and staff. GIS analyses were used to quantify habitat size and landscape characteristics, whereas field observations documented vegetation structure, habitat features, and human activity. Information regarding maintenance practices and visitor intensity was obtained through interviews with golf course personnel. To minimize observer bias, all surveys were conducted by trained observers using standardized sampling protocols. Independent observations were cross-checked to ensure consistency in species identification and data recording.

A comprehensive vegetation survey was conducted across all major habitat types within each golf course, including grasses, sedges, herbs, climbers, lianas, shrubs, and trees. Plant species were identified in the field, and GPS coordinates of trees were recorded throughout the study area.

Butterflies were surveyed using the Pollard Walk method (Pollard, 1977) between 08:00 and 10:00 h. Individuals encountered along predetermined survey routes were identified visually and photographed for confirmation.

Insects were sampled using 1 × 1 m quadrats placed across representative habitat types, including ponds, open lawns, flowering vegetation, golf holes, and fountain areas. A total of 30 randomly distributed quadrats were surveyed at Bangalore Golf Club (BGC) and 50 randomly distributed quadrats at Zion Hills Golf County (ZHGC). Quadrat locations were randomly selected within representative habitat types for each survey to ensure adequate spatial coverage. Insects were collected using hand-picking, sweep nets, and pitfall traps where appropriate. Species identity, abundance, behavior, and habitat associations were recorded for each quadrat.

No live specimens were collected during the study. Amphibian surveys were conducted between 19:00 and 23:30 h using acoustic surveys and opportunistic visual encounter surveys following Rödel and Ernst (2004). All individuals seen or heard within a 25-m radius during 15-minute survey periods were recorded following Lips et al. (2001). Individuals were photographed for subsequent identification based on external morphology and breeding-site observations.

Birds and mammals were surveyed using presence-only observations (Gelfand and Shirota, 2019). Surveys were conducted during periods of peak activity (07:00–09:00 h and 16:00–18:00 h) while maintaining a constant walking speed of approximately 1 km h^{-1}. Birds were identified using Olympus 10 × 42 binoculars, and bird survey procedures followed standard bird census approaches (Bibby et al., 2000); species identification followed Grimmett et al. (2012).

All other major taxonomic groups included in Table 1 were surveyed using the standardized or taxon-specific field methods described above. Mollusca were the exception and were included only as opportunistic records obtained during routine biodiversity surveys.

Biodiversity surveys were conducted at both golf courses between 1 August and 30 September 2023, providing identical seasonal coverage. Due to site access restrictions, Bangalore Golf Club was surveyed on Mondays, whereas Zion Hills Golf County was surveyed on Saturdays and Sundays. Despite these logistical differences, identical sampling protocols and standardized survey procedures were applied at both study sites to ensure valid comparisons.

The total survey effort comprised 283.5 person-hours (94.5 person-hours at BGC and 189 person-hours at ZHGC) and represented the combined sampling

effort for all taxonomic groups included in the study. During each survey, observations of plants, butterflies, insects, amphibians, reptiles, birds, mammals, and opportunistically encountered molluscs were recorded simultaneously using the appropriate sampling protocols for each taxonomic group.

Data analysis

Species diversity was assessed using species richness (R), the Shannon–Wiener diversity index (H′), Simpson's Dominance Index (D), and Pielou's evenness index (J). Simpson's Dominance Index (D) was calculated as (D = \sum_{i=1}^{S} p_i^2), where (p_i) is the proportional abundance of the (i^{th}) species, calculated as (p_i = n_i/N), (n_i) is the number of individuals of the (i^{th}) species, (N) is the total number of individuals of all species, and (S) is the total number of species. Higher values of D indicate greater dominance by one or a few species within the community. Shannon–Wiener diversity, Simpson's Dominance Index, and Pielou's evenness were calculated using species abundance data in R version 4.4.1 (R Core Team, 2024) with the vegan package. Alpha diversity was defined as the observed species richness within each golf course, whereas gamma diversity represented the total number of unique species recorded across both study sites, with shared species counted only once. Beta diversity was calculated using Whittaker's beta-diversity measure (Whittaker, 1960), expressed as $\beta = \gamma/\bar{\alpha}$, where γ represents the total species richness across both golf courses and $\bar{\alpha}$ represents the mean alpha diversity (species richness) of the two study sites. Alpha, beta, and gamma diversity metrics were calculated using species presence–absence (species richness) data in R (version 4.4.1; R Core Team, 2024). For inferential comparisons of abundance between the two golf courses, the observational unit was the recorded species, with cumulative abundance calculated for each species across the survey period. Mean cumulative abundance per recorded species was compared between Bangalore Golf Club (BGC) and Zion Hills Golf County (ZHGC) using Welch's two-sample t-test, while the Mann–Whitney U test was used as a complementary non-parametric analysis. Species richness was treated as a descriptive biodiversity metric and reported as the total number of unique species recorded at each golf course.

Prior to statistical analyses, the assumptions of normality and homogeneity of variance were evaluated using the Shapiro–Wilk test and Levene's test, respectively. Although the abundance data deviated from normality, Welch's two-sample t-test was retained because of its robustness to moderate departures from normality and unequal sample sizes. The Mann–Whitney U test was used as a complementary non-parametric analysis.

Pearson's chi-square test was used to compare the distribution of individuals across six microhabitat categories between Bangalore Golf Club (BGC) and Zion Hills Golf County (ZHGC). The six microhabitat categories were Trees, Plants, Flowers, Ground Surface, Pond, and Rock, with golf course (BGC versus ZHGC) as the grouping variable.

Pearson correlation and linear regression analyses were performed to evaluate relationships between biodiversity metrics and habitat characteristics. For the species–area analysis, linear regression was used separately for each golf course to examine the relationship between habitat area and species richness. The regression dataset comprised 12 area–species-richness observations for each golf course. Habitat area was expressed in acres, with values ranging from 5 to 60 acres for Bangalore Golf Club (BGC) and from 10 to 120 acres for Zion Hills Golf County (ZHGC). Each observation represented a specified habitat-area value paired with the corresponding species richness recorded for that area. Relative faunal richness was calculated as the proportion of the total recorded species represented by each major taxonomic group within each golf course.

Mean cumulative abundance per recorded species was calculated as the average number of individuals recorded per species across the entire survey period for each golf course based on cumulative abundance records obtained during field surveys. Statistical significance was assessed at $\alpha = 0.05$, and all statistical analyses were performed in R version 4.4.1 (R Core Team, 2024).

Results

Species richness

Species richness was higher at ZHGC (104 species) than at BGC (59 species). Because these values represent single cumulative site-level richness estimates, species richness was treated as a descriptive biodiversity measure rather than subjected to inferential statistical testing.

Species abundance

Mean cumulative abundance per recorded species did not differ significantly between the two golf courses (Welch's two-sample t-test: $t = -0.189$, $df = 159.7$, $P = 0.851$).

Microhabitat distribution

The distribution of individuals across the six microhabitat categories differed significantly between BGC and ZHGC (Pearson's $\chi^2 = 4525.8$, $df = 5$, $P < 2.2 \times 10^{-16}$). The analysis included Trees, Plants, Flowers, Ground Surface, Pond, and Rock as microhabitat categories, with golf course (BGC versus ZHGC) as the grouping variable.

Pearson correlation

Pearson correlation analysis indicated a very weak positive relationship between the paired observations from BGC and ZHGC ($r = 0.019$, $P = 0.8873$), indicating no statistically significant linear association.

Linear regression

Linear regression revealed strong positive relationships between habitat area and species richness for both golf courses. For ZHGC, the relationship was significant (n = 12, $R^2 = 0.8956$, adjusted $R^2 = 0.8851$, $F_{1,10} = 85.76$, $P = 3.20 \times 10^{-6}$). A similarly strong relationship was observed for BGC (n = 12, $R^2 = 0.9587$, adjusted $R^2 = 0.9545$, $F_{1,10} = 231.9$, $P = 3.03 \times 10^{-8}$).

Species richness and abundance

ZHGC exhibited higher observed species richness, with 104 species recorded, compared with 59 species recorded at BGC. Mean cumulative abundance per recorded species was 184.93 individuals at BGC and 211.60 individuals at ZHGC; this difference was not statistically significant (Welch's two-sample t-test: $t = -0.189$, $df = 159.7$, $P = 0.851$; 95% CI for the difference = −305.78 to 252.45).

Diversity indices

Diversity indices calculated separately for each major taxonomic group indicated that insects exhibited the highest species richness and diversity at both golf courses. Birds, amphibians, and reptiles exhibited higher species richness and diversity at Zion Hills Golf County (ZHGC) than at Bangalore Golf Club (BGC), whereas mammals showed comparable diversity between the two sites. Overall, diversity values were generally higher at ZHGC across most taxonomic groups (Table 1).

The alpha-diversity values, representing observed species richness, were 59 at BGC and 104 at ZHGC. A total of 26 species were shared between the two study sites. Gamma diversity, representing the total number of unique species recorded across both sites, was therefore 137 species (59 + 104 − 26). Using Whittaker's beta-diversity measure ($\beta = \gamma/\bar{\alpha}$), where γ represents total species richness and $\bar{\alpha}$ represents the mean alpha diversity of the two sites, beta diversity was calculated as $\beta = 137/[(59 + 104)/2] = 1.68$. This value represents overall species turnover between the two study sites (Fig. 3).

The Shannon–Wiener diversity index indicated slightly higher species diversity at Zion Hills Golf County (ZHGC) (3.64) than at Bangalore Golf Club (BGC) (3.17). Simpson's Dominance Index (D) was slightly higher at Bangalore Golf Club (0.06) than at Zion Hills Golf County (0.04), indicating marginally greater species dominance at BGC (Fig. 4).

ZHGC exhibited slightly higher species evenness (0.784) than BGC (0.776), indicating only a small difference in the distribution of individuals among species. Species richness was higher at ZHGC (104 species) than at BGC (59 species) (Fig. 5).

Statistical analysis

Species richness was higher in ZHGC (104 species) than in BGC (59 species). Because these values represent single cumulative richness estimates for each golf course, species richness was treated as a descriptive biodiversity measure rather than subjected to an inferential comparison. Mean cumulative abundance per recorded species did not differ significantly between the two golf courses (Welch's two-sample t-test: $t = -0.189$, $df = 159.7$, $P = 0.851$). The distribution of individuals across the six microhabitat categories differed significantly between the two golf courses (Pearson's $\chi^2 = 4525.8$, $df = 5$, $P < 2.2 \times 10^{-16}$). Strong positive linear relationships between habitat area and species richness were observed for both ZHGC ($R^2 = 0.8956$, $P < 0.0001$) and BGC ($R^2 = 0.9587$, $P < 0.0001$).

Table 1: Diversity indices and species richness of taxa recorded at Bangalore Golf Club (BGC) and Zion Hills Golf County (ZHGC). Taxa other than Mollusca were surveyed using the taxon-specific standardized field methods described in the Materials and Methods. Mollusca were not subjected to a dedicated sampling protocol; records were obtained opportunistically during routine biodiversity surveys and were included in the overall faunal inventory and diversity calculations.

Taxa / Overall indices	Richness (BGC)	Shannon (BGC)	Simpson (BGC)	Evenness (BGC)	Richness (ZHGC)	Shannon (ZHGC)	Simpson (ZHGC)	Evenness (ZHGC)
Overall species abundance	5548	–	–	–	11109	–	–	–
Overall species richness	59	–	–	–	104	–	–	–
Amphibians	1	0.000	0.000	–	7	1.570	0.716	0.809
Birds	17	1.710	0.752	0.605	49	3.450	0.961	0.888
Fishes	2	0.686	0.492	0.989	2	0.679	0.486	0.980
Insects	30	2.730	0.916	0.804	35	2.610	0.897	0.735
Mammals	3	0.395	0.219	0.360	3	0.088	0.030	0.080
Mollusca	1	0.000	0.000	–	2	0.693	0.500	1.000
Reptiles	4	1.270	0.691	0.918	6	0.562	0.241	0.314

Table 2: Comparison of mean cumulative abundance per recorded species between Bangalore Golf Club (BGC) and Zion Hills Golf County (ZHGC) using parametric and non-parametric statistical tests.

Test	Statistic	df	P-value	95% CI	BGC mean	ZHGC mean	Conclusion
Welch's two-sample t-test	t = −0.189	159.7	0.851	−305.78 to 252.45	184.93	211.60	No significant difference
Mann–Whitney U test	W = 2919.5	—	0.6091	—	—	—	No significant difference

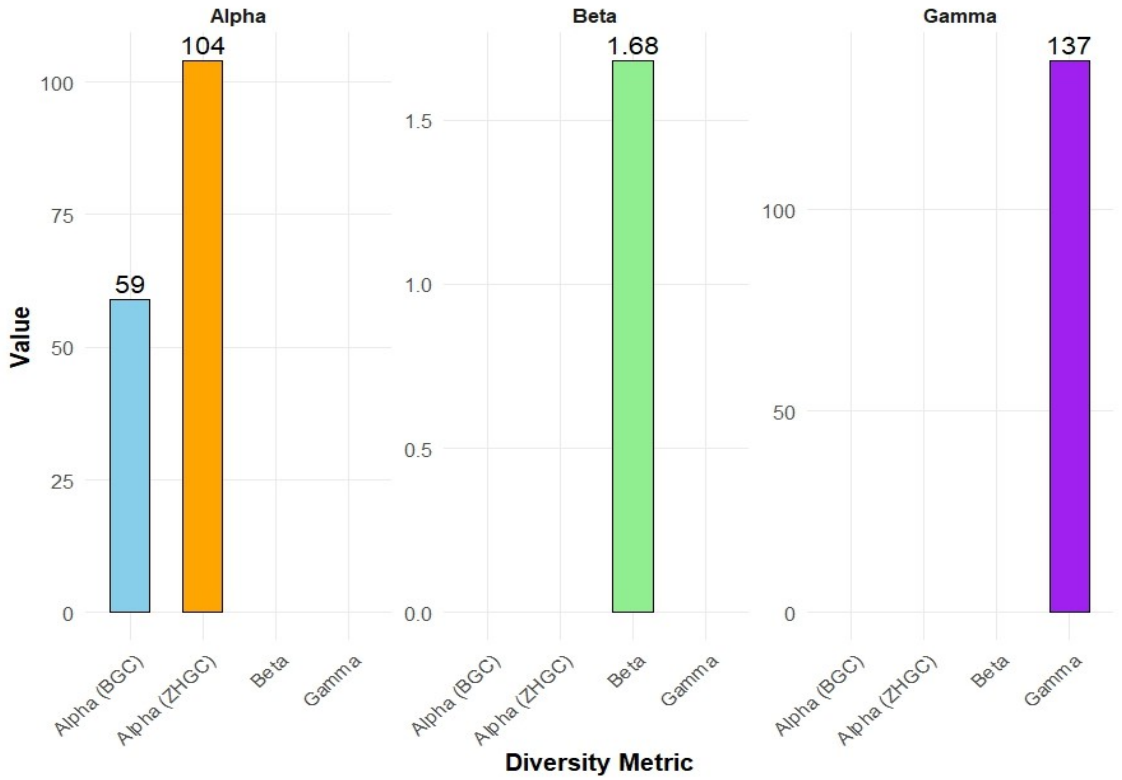

Figure 3: Alpha, beta, and gamma diversity at Bangalore Golf Club (BGC) and Zion Hills Golf County (ZHGC). Alpha diversity represents observed species richness at each golf course, gamma diversity represents the total number of unique species recorded across both sites, and beta diversity was calculated using Whittaker's measure ($\beta = \gamma/\bar{\alpha}$), where $\bar{\alpha}$ is the mean alpha diversity of the two golf courses. Diversity metrics were calculated from species presence–absence data using R version 4.4.1.

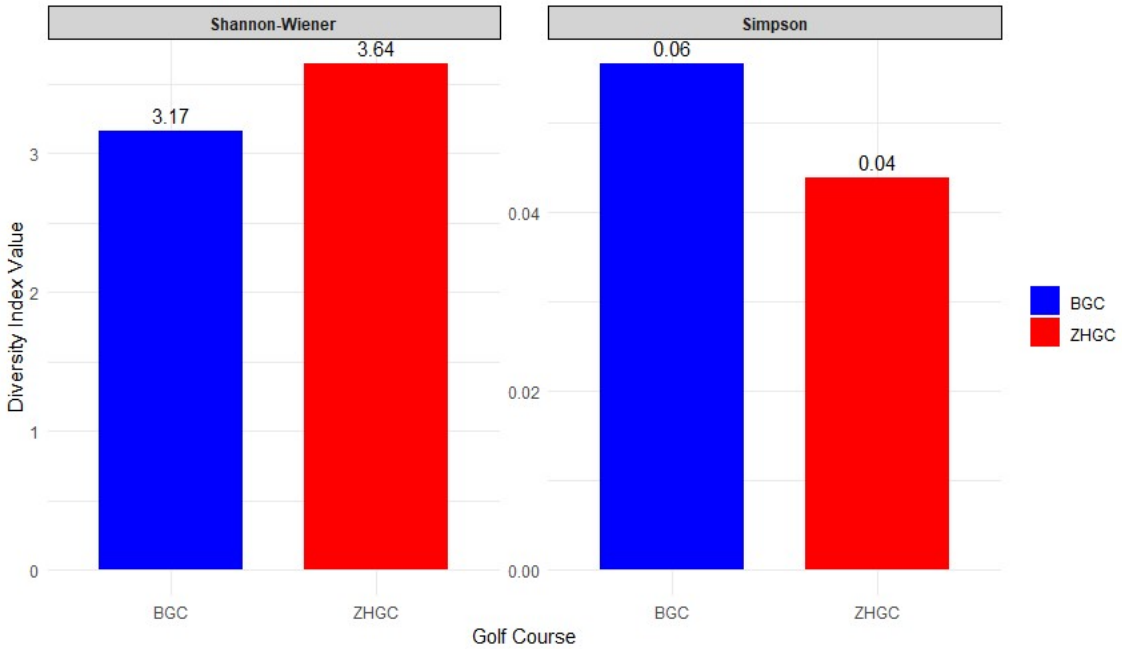

Figure 4: Shannon–Wiener diversity and Simpson's Dominance Index (D) at Bangalore Golf Club (BGC) and Zion Hills Golf County (ZHGC).

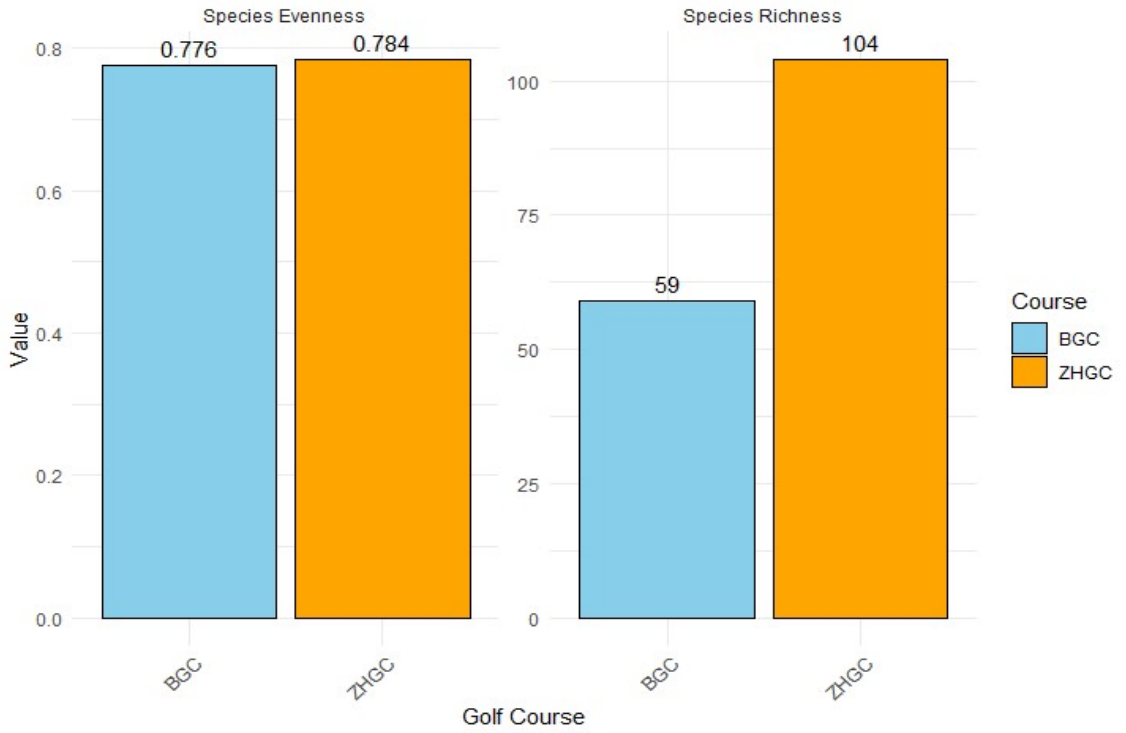

Figure 5: Species evenness and richness at Bangalore Golf Club (BGC) and Zion Hills Golf County (ZHGC). ZHGC exhibited higher species evenness (0.784) and species richness (104) than BGC (0.776 and 59, respectively).

To evaluate differences in the distribution of individuals across the six microhabitat categories between the two golf courses, Pearson's chi-square test was performed. The distribution of individuals across the six microhabitat categories differed significantly between BGC and ZHGC (Pearson's $\chi^2 = 4525.8$, df = 5, $P < 2.2 \times 10^{-16}$). The analysis included Trees, Plants, Flowers, Ground Surface, Pond, and Rock as microhabitat categories, with golf course (BGC versus ZHGC) as the grouping variable. The results are presented in Table 3.

In addition, a Welch's two-sample t-test was performed to compare mean cumulative abundance per recorded species between the two golf courses. The analysis revealed no statistically significant difference between BGC (mean = 184.93) and ZHGC (mean = 211.60) (t = −0.189, df = 159.7, P = 0.851; 95% CI = −305.78 to 252.45). Although ZHGC exhibited slightly higher mean cumulative abundance per recorded species, the difference was not statistically significant.

Graphical analysis

To visually illustrate the distribution of individuals across the six microhabitat categories between the two golf courses, the results of the Pearson's chi-square analysis are presented in Figure 6.

Discussion

General observations

The observed differences in faunal assemblages between Zion Hills Golf County (ZHGC) and Bangalore Golf Club (BGC) appear to reflect variation in habitat characteristics, golf course size, and the intensity of anthropogenic disturbance (Figs. 6–8). ZHGC, which encompasses a larger actively managed area and experiences comparatively lower levels of human disturbance, supported a greater abundance of ground-dwelling fauna than BGC. Similar patterns have been reported in managed urban green spaces, where larger and less disturbed habitats generally support greater biodiversity by providing increased habitat availability, structural complexity, and ecological resources (Hu and Lima, 2024; Lepczyk et al., 2008). Although habitat area was positively associated with species richness in the present study, differences between the two golf courses likely reflect the combined influence of habitat area, vegetation structure, and local management practices rather than any single environmental factor.

In contrast, Bangalore Golf Club supported a relatively greater proportion of arboreal fauna. This pattern may be associated with the highly urbanized landscape surrounding the golf course, where remnant tree cover provides important refuge and foraging habitat for species adapted to urban environments. Urban forests and mature tree stands are widely recognized as important habitat elements that support arboreal vertebrates within fragmented landscapes (Beninde et al., 2015; Lepczyk et al., 2008; Hu and Lima, 2024).

Table 3: Pearson's chi-square test comparing the distribution of individuals across six microhabitat categories between Bangalore Golf Club (BGC) and Zion Hills Golf County (ZHGC).

Test	Chi-Square Statistic (X^2)	Degrees of Freedom (df)	p-value	Conclusion
Chi-Square Test	4525.8	5	$< 2.2 \times 10^{-16}$	Highly significant difference

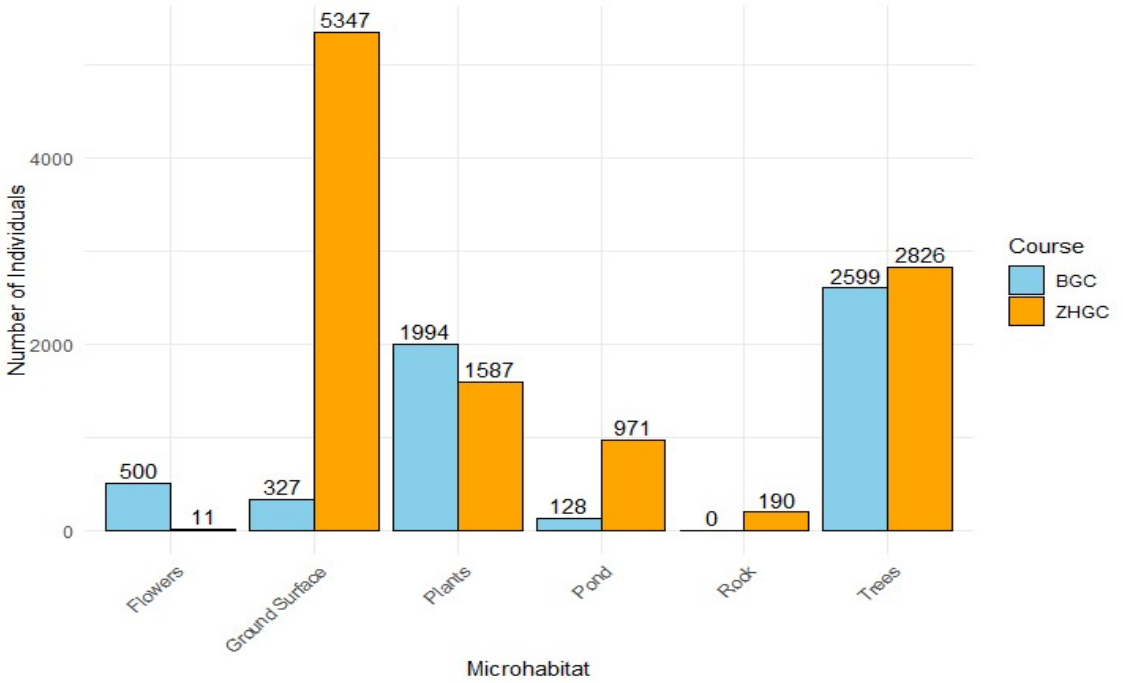

Figure 6: Pearson's chi-square test comparing the distribution of individuals across six microhabitat categories between Bangalore Golf Club (BGC) and Zion Hills Golf County (ZHGC). The microhabitat categories were Trees, Plants, Flowers, Ground Surface, Pond, and Rock. The analysis indicated a significant difference in the distribution of individuals across microhabitats between the two golf courses ($\chi^2 = 4525.8$, df = 5, $P < 2.2 \times 10^{-16}$).

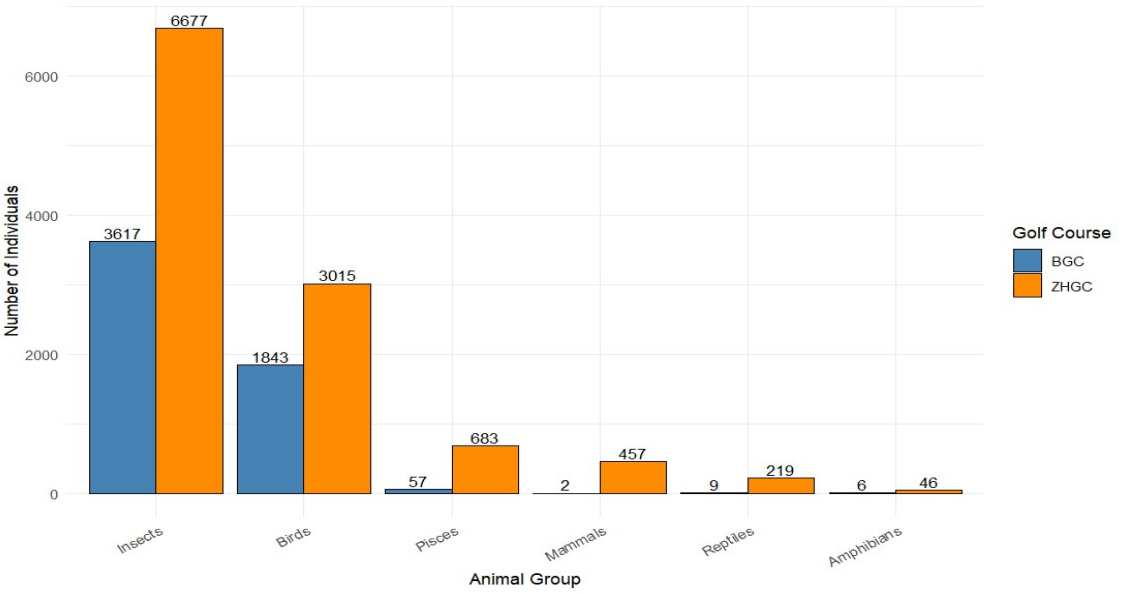

Figure 7: Abundance of individuals across major faunal groups at Bangalore Golf Club (BGC) and Zion Hills Golf County (ZHGC). Insects and birds accounted for the largest proportions of individuals at both sites, with higher overall abundance recorded at ZHGC.

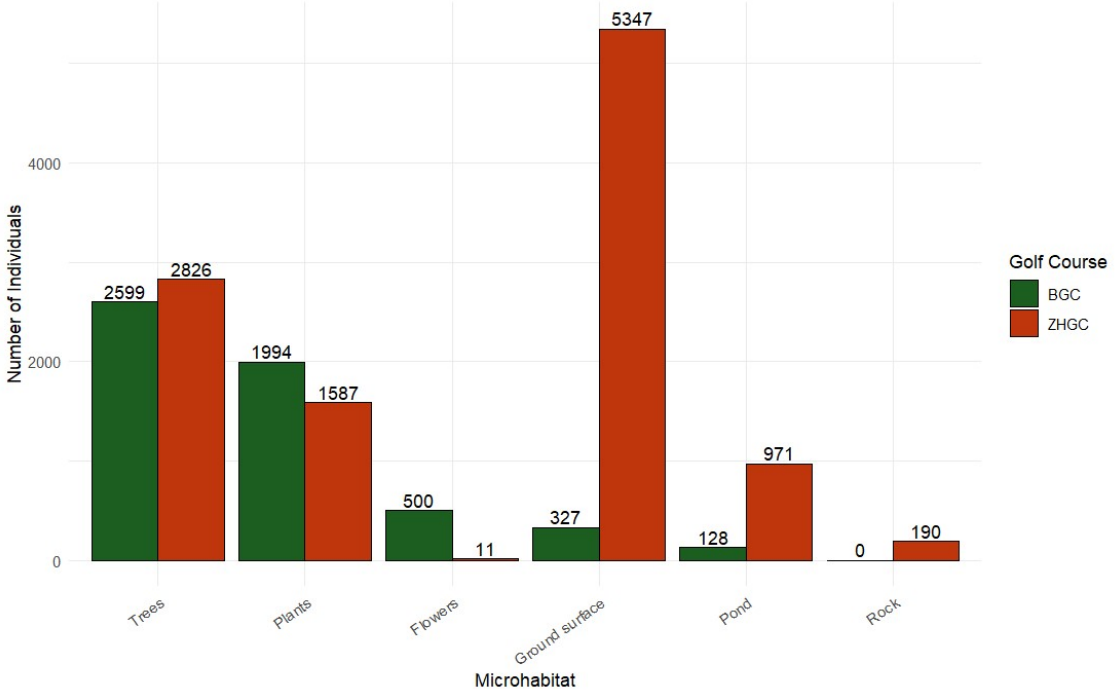

Figure 8: Abundance of individuals across microhabitats at Bangalore Golf Club (BGC) and Zion Hills Golf County (ZHGC). Ground-surface and tree microhabitats supported the highest numbers of individuals, with ZHGC showing higher overall abundance, particularly in the ground-surface microhabitat.

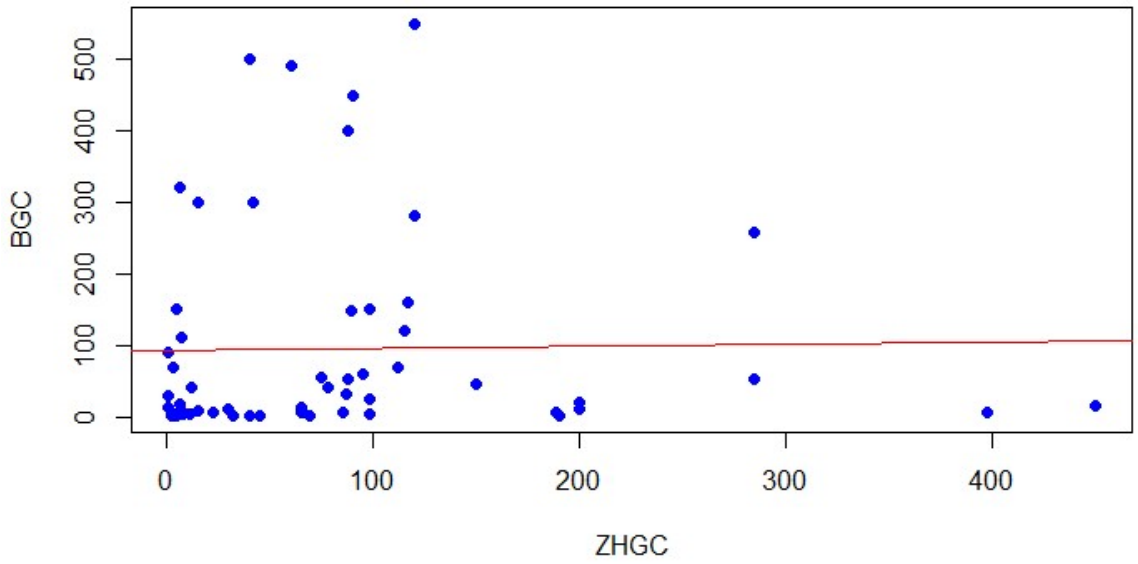

Figure 9: Pearson correlation between the measured variables at Bangalore Golf Club (BGC) and Zion Hills Golf County (ZHGC). The correlation coefficient (r = 0.019) indicated a very weak positive correlation, suggesting little or no linear relationship between the measured variables.

Pearson correlation analysis indicated a very weak positive relationship between the paired abundance observations from BGC and ZHGC (r = 0.019, P = 0.8873), indicating no statistically significant linear association. This weak relationship suggests that abundance patterns across the paired observations were not strongly correlated between the two golf courses. Differences in community structure may instead reflect variation in habitat characteristics, species composition, and local environmental conditions between the two sites.

Amphibians and other aquatic-associated taxa were comparatively underrepresented at both golf courses. This pattern likely reflects their dependence on stable freshwater habitats, suitable breeding sites, and favorable microclimatic conditions. Amphibians are widely recognized as sensitive indicators of environmental change because they respond rapidly to habitat alteration, hydrological modification, and changes in water quality (Guzy et al., 2012). Although both golf courses contained artificial water bodies, intensive landscape management and limited availability of natural aquatic habitats may have reduced habitat suitability for these taxa.

The overlap in species identity between the two golf courses indicates that some taxa were able to persist under differing levels of urbanization and human disturbance. This finding highlights the potential conservation value of appropriately managed golf courses as components of urban green infrastructure. Recent studies have increasingly recognized that well-managed golf courses can contribute to biodiversity conservation by providing supplementary habitats, enhancing landscape connectivity, and supporting diverse faunal communities within urbanized environments (Hodgkison et al., 2007; Colding and Folke, 2009; Howell et al., 2024; Howard et al., 2025). Nevertheless, such habitats should be regarded as complementary to, rather than substitutes for, natural ecosystems.

Alpha, beta, and gamma diversity:

A comparative analysis of diversity patterns demonstrated that Zion Hills Golf County (ZHGC) supported higher observed alpha diversity than Bangalore Golf Club (BGC) (Fig. 3).

The greater species richness recorded at ZHGC is likely associated with its larger actively managed area (120 acres) and greater habitat heterogeneity, including the presence of tree stands, managed turfgrass, water bodies, and semi-natural vegetation within the study area. These habitat features may provide a greater variety of resources and ecological niches for terrestrial and aquatic-associated fauna.

Numerous studies have demonstrated that larger and structurally heterogeneous habitats generally support higher local species richness by increasing the diversity of available ecological niches and resources (MacArthur and Wilson, 1967; Kang et al., 2015; Schütz and Schulze, 2015; Hu and Lima, 2024). The presence of diverse habitat features, including tree stands, managed turfgrass, water bodies, and semi-natural vegetation, may therefore have contributed to the higher alpha diversity observed at ZHGC.

In contrast, Bangalore Golf Club is situated within a densely urbanized landscape characterized by intensive surrounding infrastructure and high levels of human activity. Habitat fragmentation and reduced landscape connectivity associated with urban development are widely recognized as factors that reduce local species richness and alter community composition by limiting dispersal opportunities and increasing environmental disturbance (Lepczyk et al., 2008; Hu and Lima, 2024). These landscape characteristics may have contributed to the comparatively lower alpha diversity recorded at BGC.

Beta diversity was calculated as a single Whittaker beta-diversity value ($\beta = 1.68$) for the combined two-site dataset and therefore represents overall species turnover between the study sites rather than a value attributable to either golf course. Species turnover in managed urban landscapes can be influenced by habitat heterogeneity, environmental filtering, landscape context, and anthropogenic disturbance (Anderson et al., 2011; Tuomisto, 2010; Tews et al., 2004).

Gamma diversity, representing the cumulative unique species richness recorded across both golf courses, was 137 species. This value reflects the combined contribution of species recorded at the two study sites after accounting for 26 species shared between them. The higher gamma diversity relative to either site-level alpha diversity illustrates the complementary contribution of the two golf-course ecosystems to the regional species pool. Similar patterns have been reported in urban ecosystems, where landscape-scale biodiversity is enhanced when habitats vary in vegetation structure, management intensity, and environmental conditions (Colding and Folke, 2009; Hu and Lima, 2024).

Recalculation of diversity indices for individual taxonomic groups further demonstrated that insects and birds contributed most strongly to the overall biodiversity patterns observed in the present study. Several taxonomic groups, including birds, amphibians, and reptiles, exhibited higher diversity at ZHGC than at BGC, suggesting that larger and more structurally diverse habitats provide a wider range of ecological niches and resources capable of supporting a greater variety of species. These findings are consistent with the species–area relationship (SAR), which predicts increasing species richness with increasing habitat area, while also emphasizing the importance of habitat heterogeneity in determining biodiversity patterns (MacArthur and Wilson, 1967; Drakare et al., 2006; Rosenzweig, 1995; Kang et al., 2015).

Long-term studies examining changes in alpha, beta, and gamma diversity across broader spatial and temporal scales would provide valuable insight into the mechanisms through which urbanization, habitat management, and landscape connectivity influence biodiversity within managed urban ecosystems. Such investigations would further improve understanding of how golf courses contribute to urban biodiversity conservation and complement surrounding natural habitats (Newbold et al., 2015).

Species abundance, richness, and evenness

The present study demonstrated a positive association between golf course area and species richness, consistent with the widely recognized

species–area relationship (SAR), whereby larger habitats generally support greater numbers of species (MacArthur and Wilson, 1967; Rosenzweig, 1995; Kang et al., 2015; Schütz and Schulze, 2015). Although both golf courses exceeded the 10-ha site-area recommendation proposed by Chamberlain et al. (2007) for maximizing bird species richness, ZHGC, with its 120-acre actively managed playing area, supported substantially higher observed species richness than BGC. In addition to habitat area, factors such as habitat heterogeneity, vegetation complexity, availability of aquatic habitats, and lower levels of anthropogenic disturbance may collectively influence biodiversity patterns within managed golf course ecosystems (Colding and Folke, 2009; Hu and Lima, 2024; Howell et al., 2024).

The Shannon–Wiener diversity and Pielou's evenness indices (Figs. 4 and 5; Table 1) further supported the observed biodiversity patterns. Although evenness differed only slightly between the two golf courses, ZHGC consistently supported higher species richness and greater diversity across several taxonomic groups. These findings suggest that larger, structurally diverse golf courses containing a mosaic of habitat types—including mature trees, managed grasslands, water bodies, and semi-natural vegetation—can provide suitable habitats for a wide variety of taxa within urban landscapes (Colding and Folke, 2009; Howard et al., 2025).

The present findings further support recent studies indicating that well-managed golf courses can function as valuable components of urban green infrastructure by providing supplementary habitat for wildlife within human-dominated landscapes. However, biodiversity conservation within golf courses should emphasize the maintenance of habitat heterogeneity, native vegetation, and ecological connectivity rather than increasing habitat area alone. Future studies incorporating detailed analyses of vegetation composition, habitat quality, and species interactions would provide a more comprehensive understanding of the mechanisms shaping biodiversity in managed urban ecosystems (Haas et al., 2020).

Statistical interpretation and ecological implications

Zion Hills Golf County (ZHGC) supported greater observed species richness than Bangalore Golf Club (BGC), with 104 and 59 recorded species, respectively. Because these values represent single cumulative richness estimates for each golf course, the difference was treated descriptively rather than subjected to inferential statistical testing. Similarly, comparisons of mean cumulative abundance per recorded species using Welch's two-sample t-test revealed no significant difference between the two golf courses. These findings suggest that differences in raw species counts and cumulative abundance alone may not fully capture broader biodiversity patterns within managed urban ecosystems.

In contrast, the distribution of individuals across microhabitat categories differed significantly between the two golf courses (Table 3), indicating that individuals were distributed differently among the six recorded microhabitats. Such patterns highlight the importance of evaluating multiple components of biodiversity, as communities with similar numbers of species may differ substantially in species identity and ecological composition (Whittaker, 1960; Tuomisto, 2010). The observed variation in species composition between the two golf courses likely reflects differences in habitat characteristics, vegetation structure, landscape context, and management intensity, all of which influence the occurrence of individual species within urban environments.

Although total abundance and cumulative species richness were numerically greater in ZHGC, the absence of a significant difference in mean cumulative abundance per recorded species suggests that the species–area relationship observed in the present study is primarily associated with increased species accumulation in the larger habitat rather than a proportional increase in the average abundance of individual species. This interpretation is consistent with ecological theory, which proposes that larger habitats support greater species richness through increased habitat heterogeneity, greater environmental variability, and reduced extinction probability rather than uniformly higher abundances of all species (MacArthur and Wilson, 1967; Rosenzweig, 1995).

The significant difference in the distribution of individuals across microhabitat categories between the two golf courses further emphasizes that biodiversity assessments should extend beyond simple measures of species richness or abundance. Community composition provides important information about habitat quality, ecological integrity, and the distribution of specialist and generalist species within managed landscapes. Recent studies have similarly emphasized that assessments integrating species richness, diversity indices, and community composition provide a more comprehensive understanding of biodiversity responses to urbanization than any single metric alone (Lepczyk et al., 2008; Hu and Lima, 2024; Howard et al., 2025).

Overall, the findings of the present study demonstrate that well-managed golf courses can contribute meaningfully to urban biodiversity conservation by supporting diverse faunal communities within human-dominated landscapes. However, maximizing their conservation value requires management strategies that maintain habitat heterogeneity, native vegetation, aquatic habitats, and ecological connectivity while minimizing unnecessary habitat disturbance. Such approaches are likely to enhance the role of golf courses as complementary components of broader urban green infrastructure rather than isolated recreational landscapes (Colding and Folke, 2009; Howell et al., 2024).

Conclusions

The present study demonstrates that managed golf course ecosystems can contribute to urban biodiversity conservation by providing structurally diverse habitats within highly modified landscapes.

ZHGC, which had a larger actively managed playing area and greater habitat heterogeneity than BGC, supported higher observed species richness, with 104 species compared with 59 species at BGC. These values represent cumulative site-level richness estimates and were therefore interpreted descriptively rather than subjected to inferential statistical testing.

Mean cumulative abundance per recorded species did not differ significantly between the two golf courses (Welch's two-sample t-test: $t = -0.189$, $df = 159.7$, $P = 0.851$), whereas the distribution of individuals across microhabitat categories differed significantly between sites. These findings indicate that biodiversity assessment in managed urban ecosystems should consider multiple complementary metrics rather than species richness alone.

Although the study represents a short-term comparison of two golf courses and does not directly assess long-term population dynamics, functional connectivity, or demographic processes, the findings suggest that well-designed and ecologically managed golf courses can function as complementary components of urban green infrastructure. Maintaining habitat heterogeneity, native vegetation, aquatic habitats, and ecological connectivity may enhance their value for urban wildlife. Further studies involving additional golf courses, broader geographic coverage, and long-term monitoring are needed to determine the mechanisms underlying these biodiversity patterns.

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Author contributions

Nag K. S. Chetan: Conceptualization, study design, methodology, supervision, field investigation, data curation, validation, interpretation of results, visualization, original draft preparation, review and editing, project administration, and overall coordination of the study. Amith S J: Formal analysis, data analysis, visualization, interpretation of results, writing – review and editing. Prathusha: Field investigation (data collection), data curation, and writing – review and editing.

Conflict of interest

The authors declare that there are no conflicting issues related to this research article.

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