Freshwater fish diversity and assemblage patterns in reforested streams of the Forest Research Institute Malaysia

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Abstract. Streams within urban forest or surrounded by developments may support important freshwater biodiversity. This study assessed freshwater fish diversity, assemblage structure, and environmental relationships in three streams located within a reforested landscape and recreational forest in Selangor, Malaysia. Fish sampling was conducted monthly from June 2022 to July 2023 using backpack electrofishing along standardized transects. A total of 19 species representing 11 families were recorded, with Cyprinidae being the dominant family. Species composition differed among streams, with one stream supporting several unique taxa and a distinct assemblage structure. Diversity analysis indicated the highest diversity in the conservation-buffered stream (C-BS) (H'=1.582), while the recreationally impacted stream exhibited lower evenness and greater dominance by opportunistic species. Abundance–biomass comparison analysis revealed differences in community stability among streams, indicating varying levels of ecological disturbance. Water-quality measurements showed significant variation only in total dissolved solids among sites. Multivariate ordination suggested that environmental variables explained limited variation in assemblage structure, highlighting the role of habitat complexity and hydrological connectivity. These findings demonstrate that streams within urban forest can support diverse freshwater fish assemblages and contribute to biodiversity conservation within human-dominated landscapes.

Keywords: fish assemblage structure, biodiversity refugia, riparian restoration, tropical river ecology.

Introduction. Urbanization is widely recognized as one of the most significant drivers of biodiversity loss worldwide, primarily through habitat modification, fragmentation, and degradation of aquatic ecosystems (Paul & Meyer, 2001; Walsh et al., 2005). Rapid expansion of urban landscapes alters hydrological regimes, increases pollutant loads, and simplifies habitat structure, leading to substantial changes in freshwater community composition and ecosystem functioning (Walsh et al., 2005; Vietz et al., 2016). These processes collectively contribute to the phenomenon known as the Urban Stream Syndrome, which describes consistent ecological degradation in streams draining urban catchments, including reduced biodiversity, altered trophic structure, and dominance of tolerant species (Walsh et al., 2005).

Despite these pressures, certain urban habitats can function as refugia, providing conditions that allow species persistence within human-dominated landscapes (Chester & Robson, 2013; Gentili et al., 2024). Urban refugia are typically represented by green infrastructure such as city parks, forest reserves, grasslands and semi-natural green spaces that maintain structural habitat complexity and ecological connectivity (Haase, 2021; Gentili et al., 2024). Such environments may buffer ecological disturbances and provide essential habitat for wildlife populations (Hale et al., 2019; Haase, 2021). Recent studies have emphasized that urban refugia can support significant biodiversity when natural habitat structure and environmental heterogeneity are maintained (Chester & Robson, 2013; Gentili et al., 2024). Within these systems, aquatic habitats play a particularly important role because streams and rivers act as ecological corridors, facilitating species movement and maintaining connectivity among fragmented habitats (Fagan, 2002; Chester & Robson, 2013). Streams within and flowing through the urban forest may therefore represent critical components of biodiversity refugia within cities (Chester & Robson, 2013; Hale et al., 2019; Gentili et al., 2024). Riparian vegetation can regulate microclimatic conditions, stabilize stream banks, reduce sediment inputs, and maintain water quality, all of which influence freshwater community structure (Naiman & Décamps, 1997; Sweeney & Newbold, 2014). However, aquatic ecosystems within urban environment are still subject to varying levels of anthropogenic disturbance, particularly where recreational activities and infrastructure development occur (Walsh et al., 2005). Differences in riparian integrity, canopy cover, and human access can produce substantial variation in environmental conditions among streams even within the same urban forest landscape. Understanding how such environmental heterogeneity influences freshwater fish assemblages is therefore essential for evaluating the conservation value of urban forest ecosystems. The Forest Research Institute Malaysia (FRIM) provides a unique setting to investigate these dynamics. Located approximately 16 km northwest of Kuala Lumpur, FRIM represents one of the most successful long-term reforestation initiatives in Southeast Asia. Established in 1926 on degraded tin-mining land, the site has been transformed into a mature secondary dipterocarp forest covering more than 500 ha (Abdullah et al., 2015; Shahidan & Md-Khalid, 2019). Within this reforested landscape, several stream systems persist under varying degrees of human influence, ranging from recreationally accessed streams to more protected forest sections. These streams provide an opportunity to examine whether long-term reforestation within an urban matrix can support freshwater biodiversity and maintain stable fish assemblages despite surrounding urban pressures (Lamb et al., 2005).

Freshwater fish assemblages are widely used as indicators of ecological condition because they respond sensitively to habitat structure, hydrological variation, and water quality (Karr, 1981; Fausch et al., 1990). Changes in species composition, dominance structure, and biomass distribution can reveal early signals of environmental stress in aquatic ecosystems (Karr, 1981; Clarke & Warwick, 2001). In particular, the abundance–biomass comparison (ABC) method provides a sensitive indicator of disturbance by comparing abundance and biomass patterns within communities (Warwick, 1986; Clarke & Warwick, 2001). When combined with diversity indices and multivariate analyses, these approaches offer valuable insights into the ecological status of freshwater systems and the mechanisms structuring fish communities.

In this study, fish assemblages across three streams located within an urban forest landscape and an adjacent recreational forest area were examined to evaluate the role of reforested urban streams as biodiversity refugia. Specifically, the following hypotheses were tested:

H1: Fish assemblages in conservation-buffered streams within the reforested landscape exhibit higher diversity and greater community stability than those in recreation-impacted streams;

H2: Streams with more intact riparian structure and canopy cover support more balanced fish assemblages with higher evenness and lower dominance by opportunistic species;

H3: Local environmental conditions, particularly physicochemical water parameters and habitat characteristics, play a stronger role in structuring fish assemblages than urban proximity alone.

To address these hypotheses, fish diversity, community composition, and biomass distribution across streams with contrasting disturbance regimes were quantified and their relationships with environmental variables were examined. By integrating diversity metrics, abundance–biomass comparisons, and multivariate analyses, this study provides insights into how long-term reforestation and local habitat conditions influence freshwater biodiversity within urban landscapes.

Material and Method

Study area. Taman Eko Rimba Komenwel is located within the Bukit Lagong Forest Reserve in Selangor, Peninsular Malaysia, about 20 km northeast of Kuala Lumpur. The area is covered by lowland tropical forest and serves as a recreational destination for both local and international visitors. The eco park is managed by Selangor Forestry Department and equipped with various facilities including campsites, public toilets and open spaces for recreational activities. A stream originating from the hills of Bukit Lagong Forest Reserve, flows through the eco park before joining the Mayong River, which subsequently drains into the Selangor River. The Selangor River system forms part of an important river basins that flows through highly urbanized and densely populated areas. Although the stream is located in within the protected area, the park is primarily managed for recreational purposes, resulting in varying levels of human influence and riparian disturbance. Consequently, the site provides a valuable opportunity to study freshwater fish assemblages in an urban forest environment subjected to moderate anthropogenic pressure. The FRIM, located about 16 km northwest of Kuala Lumpur, is one of the most successful long-term reforestation initiatives in Southeast Asia. Its stream network, comprising the Kroh and Chemubong rivers, represents a mosaic of habitats varying in canopy cover, substrate, and hydrological stability. Prior to 2019, Kroh River was a popular recreational area for visitors to FRIM. In recent years, however, recreational use has declined as visitor access is managed through guided tours and organized visits. The river flows through areas with minimal to moderate infrastructure, including trails and visitor facilities and is therefore exposed to some degree of human activity. In contrast, the Chemubong River is located within research plots and conservation buffer zone, where no infrastructure has been developed along the river. The area is limited to any visitors, and the riparian vegetation remains largely intact, providing a less disturbed stream environment.

At the stream level, the streams in this study were classified into two categories: recreation impacted stream (RIS; Kroh and Komenwel Rivers) and conservation-buffered stream (C-BS; Chemubong River). RIS included streams with publicly accessible reaches and evident recreational use, resulting in partial riparian disturbance. C-BS was located predominantly within restricted forest areas with limited human access and relatively intact riparian vegetation (Figure 1, Table 1). Although only one C-BS was available, it represents a rare long-term reforested reference surrounded an urban matrix. A key limitation of this study is the small sample size (n=3 streams), which may limit inferential strength and reduce the generalizability of the findings.

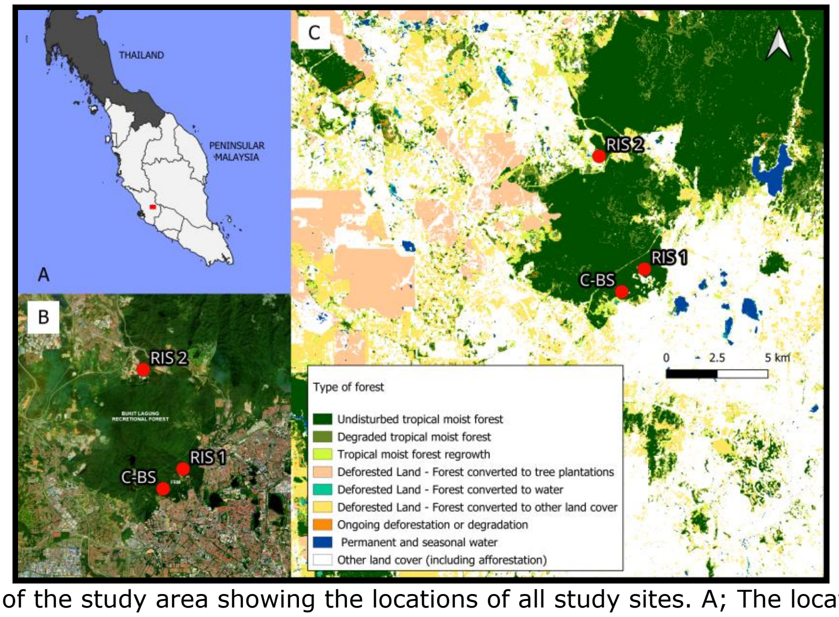

Figure 1. Map of the study area showing the locations of all study sites. A; The location of the site in Peninsular Malaysia, B; Satellite image of urban matrix surrounding the study site, and C; type of forest within the study site. C-BS and RIS 2 is located within FRIM compartment.

Table 1

The environmental parameters of three streams in this study

Stream	*Stream category*	*Disturbance regime*	*Width (m)*	*Depth (m)*	*Dominant substrate*	*Canopy cover (dominant)*
Kroh (RIS 1)	Recreation impacted stream	Moderate (public recreation zones)	7.4	0.48	Boulder–sand with gravel patches	Variable (open closed)
Komenwel (RIS 2)		High (public recreation zones)	3.8	0.29	Boulder–sand	Predominantly open
Chemubong (C-BS)	Conservation-buffered stream	Low (limited public access)	5.4	0.46	Sandy–gravel	Closed dominant

Fish sampling and identification. Fish samplings were carried out once a month from June 2022 to July 2023. A standardized electrofishing technique was employed using a Root-Smith LB24 backpack electro-fisher, supplemented with scoop nets (3 mm mesh) to collect stunned individuals along each 200 m transect. Electrofishing was conducted by a trained operator assisted by three to four collectors. Captured fish were immediately transferred to aerated buckets containing river water for processing. Each specimen was identified to species level in the field whenever possible using taxonomic keys (Zakaria-Ismail et al., 2019). Most specimens were released unharmed following enumeration and biometric measurement, while one voucher specimen per species was preserved in 10 % buffered formalin and later transferred to 70% ethanol for curation in the Zoological Collection of FRIM. Species biomass was determined using a digital balance (±0.01 g), and abundance was recorded for each stream. All sampling procedures complied with FRIM's ethical guidelines for wildlife research and followed best practices to minimize ecological disturbance. This approach ensured reliable quantitative data for assessing community structure while maintaining the integrity of the reforested habitats.

Water quality measurements. Physicochemical parameters were measured *in situ* concurrently with fish sampling in each of the three streams. Measurements were taken using a Hanna HI98194 multiparameter probe, which was calibrated before each sampling event according to manufacturer specifications. The probe was immersed approximately 10–20 cm below the surface and allowed to stabilize for about 30 seconds prior to recording. The variables monitored included pH, temperature (°C), dissolved oxygen (DO %, mg L^{-1}), total dissolved solids (TDS, ppm), and turbidity (FNU). Measurements were taken at three replicate points, and mean values were used for subsequent analysis.

Data analysis. Quantitative analyses were conducted to evaluate spatial patterns in fish diversity and community structure among the three study sites. Venn diagrams were generated using the online tool (https://bioinformatics.psb.ugent.be/webtools/Venn/) to visualize shared and unique species across sites. Six standard biodiversity indices were calculated using PAST v4.03 (Hammer et al., 2001) to assess species diversity, richness, and evenness: Shannon (H′) and Simpson (1–D) for species diversity; Margalef (R_1) and Menhinick (R_2) for species richness; and Equitability (J) and Evenness (E) for species evenness. Shannon (H′) and Simpson (1–D) diversity indices were calculated for each sampling event to quantify diversity patterns across the study sites. Diversity t-tests were then performed to evaluate statistical differences among streams based on these indices, with monthly sampling events treated as temporal replicates within each stream. Prior to statistical analysis, data distributions were examined for normality and homogeneity of variances to ensure that assumptions of the t-test were satisfied (Table 2). Although diversity indices may be influenced by sample size and species abundance structure, they remain widely used for comparative analysis of fish assemblages in freshwater ecosystems

Table 2

The formula and general value interpretation for each index used in this study

Indices	*Formula*	*Description*	*Value interpretation*
Shannon (H′)	$H' = -\Sigma (p_i \ln p_i)$	Measure species diversity considering both abundance and evenness	Lower values indicate low richness; higher values indicate greater species richness
Simpson (1-D)	$1 - D = 1 - \Sigma (p_i^2)$	Represents the probability that two individuals belong to different species	Values closer to 1 indicate higher diversity
Margalef (R1)	$R_1 = (S - 1) / \ln(N)$	Estimates species richness adjusted for sample size	Lower values indicate low richness; higher values indicate greater species richness
Menhinick (R2)	$R_2 = S / \sqrt{N}$	Measure species richness relative to total individuals	
Equitability (J)	$J = H' / \ln(S)$	Indicates how evenly individuals are distributed among species	Values closer to 1 indicate more even distribution of individuals among species
Evenness (E)	$E = (e^{H'}) / S$	Alternative measure of species evenness	
Variables	S = number of species; N = total individuals; $p_i = n_i / N$	n_i = individuals of species i; ln = natural logarithm; Σ = summation	

To examine assemblage similarity, fish abundance data were analyzed using the Bray–Curtis's similarity index, and hierarchical clustering (group-average linkage) was performed in PRIMER v6. Taxonomic distinctness of fish assemblages was calculated to evaluate community structure beyond species richness. The average taxonomic distinctness (Δ^+) was computed following Clarke and Warwick (1998) using species abundance data and a hierarchical taxonomic tree. These indices account for the relatedness among species and are robust to differences in sample size, providing additional insight into community diversity and phylogenetic structure. To evaluate disturbance levels for each stream, abundance–biomass comparison (ABC) curves were constructed following Warwick (1986), comparing cumulative biomass and abundance distributions for each site. The degree of disturbance was quantified using the W statistic (Clarke & Warwick, 2001), where values near +1 indicate undisturbed conditions, values around 0 reflect moderate stress, and negative values denote disturbed assemblages dominated by small-bodied species. This approach is particularly sensitive to pollution and habitat alteration in tropical freshwater ecosystems (Meire & Dereu, 1990).

Physicochemical parameters of water, including temperature, pH, dissolved oxygen, turbidity, and total dissolved solids, were measured at each site. Differences among sites were statistically tested using one-way ANOVA, followed by Tukey's post hoc test to identify pairwise differences, with significance set at $p<0.05$. Canonical correspondence analysis (CCA) was used to examine relationships between fish assemblage structure and measured environmental variables. Environmental variables were standardized to zero mean and unit variance to ensure comparability. The CCA was performed using PAST v4.03, and the significance of species–environment relationships were tested using a Monte Carlo permutation test with 999 permutations. Ordination biplots were used to visualize patterns of association between sampling sites, species, and environmental gradients. Because only three streams were included in this study, the statistical analyses were intended to detect site-specific ecological patterns rather than to provide regional generalizations.

Results. A total of 19 species of freshwater fish from 11 families were recorded from three study sites including two introduced species (*Poecilia reticulata* and *Oreochromis niloticus*) from two families (Poeciliidae and Cichlidae) (Table 3). The Cyprinidae family was the most dominant family with four species, followed by Channidae with three species, Danionidae and Clariidae with two species each, and other families with single species each (Figure 2). RIS 1 contained one unique species, whereas RIS 2 contained seven, and both sites shared four species. RIS 1 shared nine species with the C-BS, whereas four species were shared between RIS 2 and C-BS. Three species were recorded across all study sites, and no unique species were recorded in C-BS (Figure 3).

Table 3

List of freshwater fish species recorded from three streams on the study site

No.	*Species / Family*	*RIS 1*	*RIS 2*	*C-BS*
	Cyprinidae			
1	*Barbodes sellifer* (Kottelat & Lim, 2021)	1		1
2	*Barbonymus schwanefeldii* (Bleeker, 1854)		1	
3	*Mystacoleucus obtusirostris* (Valenciennes, 1842)		1	
4	*Neolissochilus soroides* (Duncker, 1904)		1	
5	*Poropuntius deauratus* (Valenciennes, 1842)	1	1	
	Danionidae			
6	*Danio albolineatus* (Blyth, 1860)	1		1
7	*Rasbora vulgaris* (Duncker, 1904)	1		1
	Bagridae			
8	*Hemibagrus capitulum* (Popta, 1906)		1	
	Sisoridae			
9	*Glyptothorax schmidti* (Volz, 1904)		1	
	Clariidae			
10	*Clarias batrachus* (Linnaeus, 1758)	1	1	1
11	*Clarias leiacanthus* (Bleeker, 1851)	1		1
	Mastacembelidae			
12	*Mastacembelus unicolor* Cuvier, 1832		1	
	Synbranchidae			
13	*Monopterus javanensis* (Lacepede, 1800)	1	1	1
	Osphronemidae			
14	*Betta pugnax* (Cantor, 1849)	1	1	1
	Channidae			
15	*Channa limbata* (Cuvier, 1831)	1		1
16	*Channa striata* (Bloch, 1793)		1	1
17	*Channa lucius* (Cuvier, 1831)	1		
	Poeciliidae			
18	*Poecilia reticulata* (Peters, 1859) *	1		1
	Cichlidae			
19	*Oreochromis niloticus* (Linnaeus, 1758) *		1	
	Total species	12	11	10

* Introduced species; presence (1) species occurrence in each stream; conservation-buffered streams (C-BS); recreation-impacted streams (RIS).

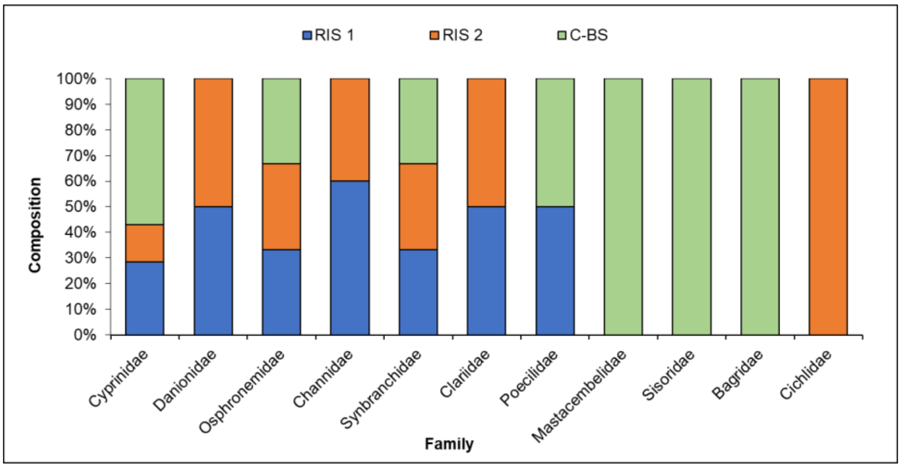

Figure 2. Family composition of freshwater fish across the study sites.

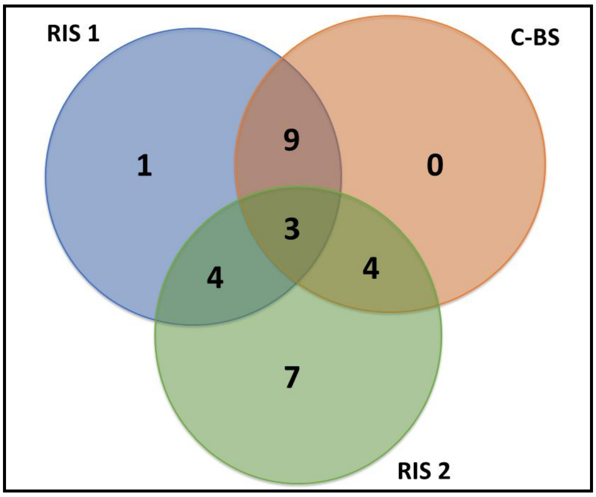

Figure 3. Venn diagram illustrates shared and unique fish species among the study sites.

The rank–abundance curves indicate that *Poropuntius deauratus*, *Barbodes sellifer*, and *Rasbora vulgaris* consistently dominated fish assemblages across all streams (Figure 4), highlighting their ecological generalism and ability to persist under a wide range of environmental conditions. C-BS exhibited the highest diversity (H’=1.582; 1–D=0.71). RIS 2, while hosting the highest richness (Margalef=1.75), displayed the lowest evenness (E=0.21). RIS 1 showed intermediate diversity (H’=1.18) but the greatest total abundance (1,352 individuals). Both species richness estimators (Chao-1 and ACE) indicate that no additional species are likely to be recorded with increased sampling effort (Table 4). Diversity t-test comparisons of Shannon and Simpson indices showed significant differences among all streams ($p<0.05$).

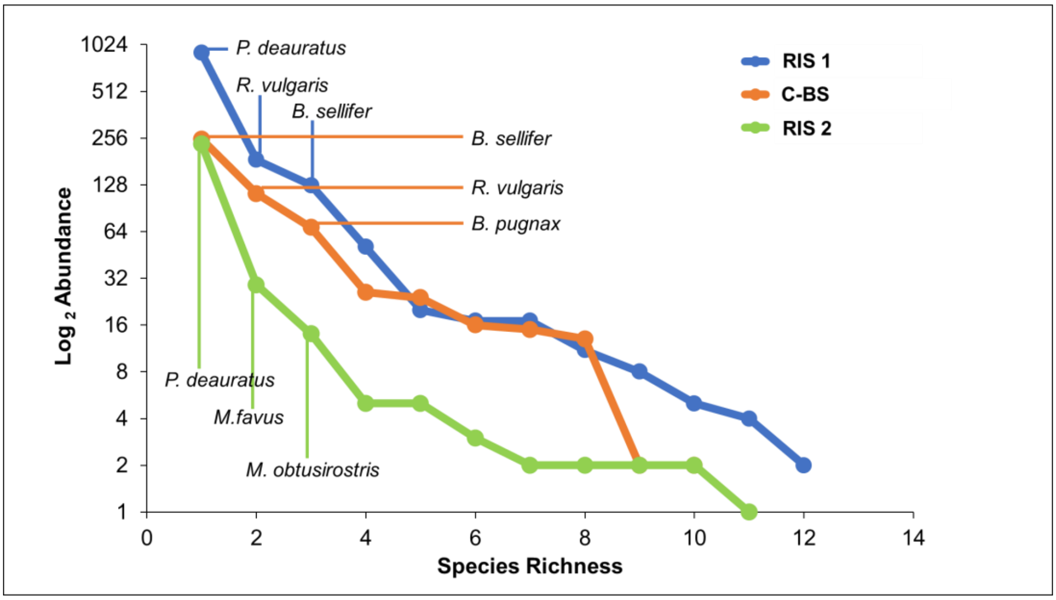

Figure 4. Rank abundance curve for all streams studied.

Table 4

Diversity measurement of freshwater fish across three study sites

Parameters	*RIS 1*	*RIS 2*	*C-BS*
Taxa_S	12	11	10
Individuals	1352	300	530
Simpson_1-D	0.53	0.38	0.71
Shannon_H	1.18	0.912	1.582
Evenness_e^H/S	0.27	0.21	0.50

Parameters	*RIS 1*	*RIS 2*	*C-BS*
Equitability_J	0.47	0.37	0.69
Menhinick	0.33	0.64	0.43
Margalef	1.53	1.75	1.44
Chao-1	12	11.12	10
ACE	12	11.38	10

Conservation-buffered streams (C-BS), recreation-impacted streams (RIS).

Cluster analysis revealed two major assemblage clusters at approximately 30 % similarity consisting of Group 1 (RIS 1 and C-BS) and Group 2 (RIS 2). These results indicate that the fish community in RIS 2 is more distinct than those in the other two systems, whereas RIS 1 and C-BS exhibit stronger similarity in assemblage composition (Figure 5). However, the analysis of average taxonomic distinctness (Δ^+) showed that fish communities from all study sites fell within the 95% probability funnel, indicating that the observed taxonomic breadth of the fish assemblages did not differ significantly from the expected values. This suggests that, despite differences in species composition and richness, the overall taxonomic structure of the assemblages remained within the range predicted for a random subset of the master species list. No river exhibited taxonomic over- or under-dispersion, reflecting a relatively consistent level of taxonomic relatedness across the study sites (Figure 6).

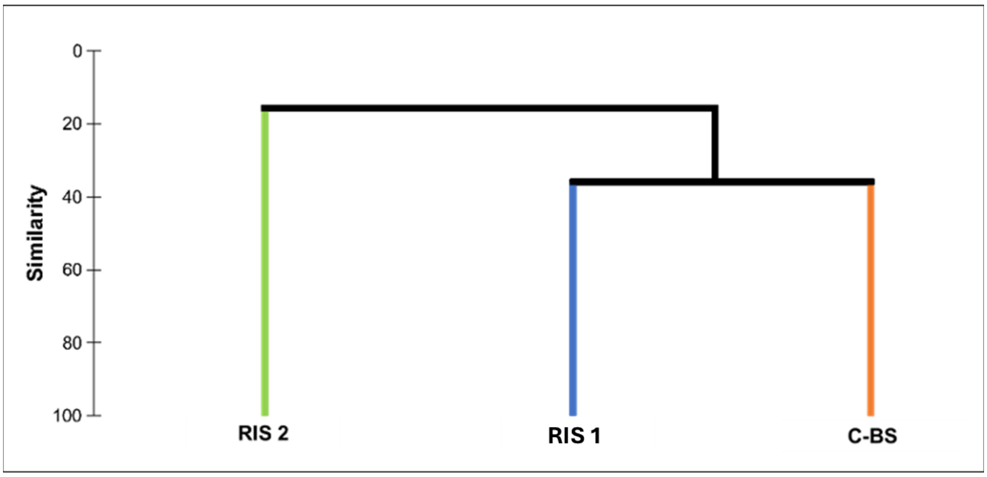

Figure 5. Cluster dendrogram based on Bray–Curtis similarity of fish assemblages across study sites.

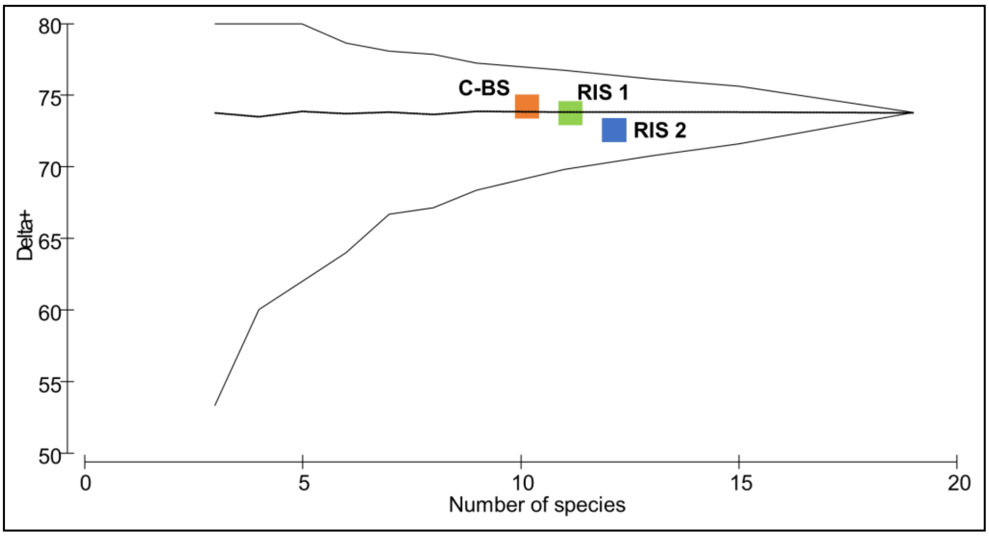

Figure 6. Taxonomic distinctness (Δ^+) of taxonomic breadth of the fish assemblages among study sites.

The ABC analysis revealed clear differences in community stability among the three rivers, reflecting variation in habitat conditions and disturbance intensity. In RIS 1, the close overlap between abundance and biomass curves (W=0.017) indicates a relatively stable assemblage structure, where biomass is not disproportionately dominated by either large-bodied or small-bodied species. In contrast, C-BS exhibited a pronounced separation between abundance and biomass curves, with abundance consistently exceeding biomass (W=−0.173), which would traditionally indicate dominance by small-bodied species and a disturbed community structure. However, despite having less human disturbance and comparatively intact riparian vegetation, C-BS unexpectedly had the lowest W-value. Instead of being a sign of ecological degradation, this trend could indicate the numerical dominance of naturally small-bodied fish species that live in structurally complex forest-stream ecosystems. Dense canopy cover, woody debris, and diverse microhabitats can support high abundances of small fish, raising abundance relative to biomass. Furthermore, the ABC technique should be interpreted carefully in tropical freshwater fish communities and supplemented with habitat assessments, diversity metrics, and species composition data because it was first established for marine benthic assemblages. Following these C-BS characteristics, RIS 2 displayed intermediate characteristics, with a slightly negative W value (−0.022), suggesting moderate disturbance but a more balanced assemblage compared to the traditional interpretation of C-BS (Figure 7).

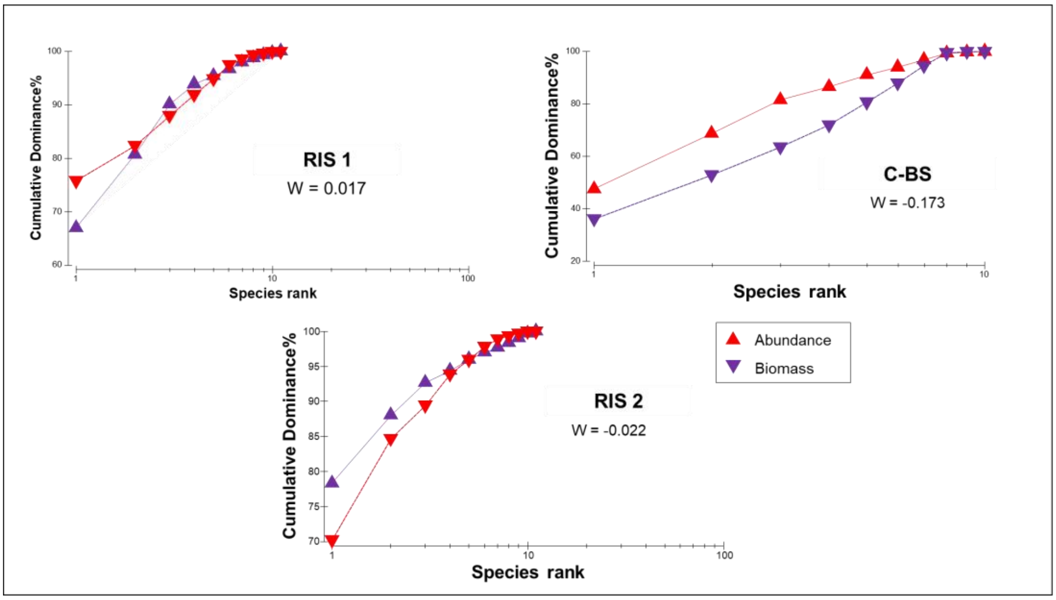

Figure 7. Abundance–biomass comparison (ABC) curves for fish assemblages in all streams. Curve relationships indicate relative community stability: (a) RIS 1 (W=0.017, stable), (b) C-BS (W=−0.173, disturbed), and (c) RIS 2 (W=−0.022, moderately disturbed).

Among the six measured physicochemical parameters—pH, temperature, dissolved oxygen (DO%, DO ppm), turbidity, and total dissolved solids (TDS)—only TDS differed significantly among the three streams (ANOVA, $p<0.05$), while all other variables showed no statistically significant difference. Post-hoc analyses revealed that TDS values differed significantly between all river pairs, highlighting spatial variation in dissolved solids among the three systems (Figure 8). The CCA ordination illustrates the pattern of association between fish assemblages and measured environmental variables, but the overall species–environment relationship was not statistically significant ($p>0.05$). This indicates that the selected environmental variables explain only a limited proportion of the variation in fish community structure. The non-significant CCA result implies that fish community composition is likely shaped by additional factors not captured in the analysis, such as habitat complexity, hydrological connectivity, stochastic colonization processes, or biotic interactions (Figure 9).

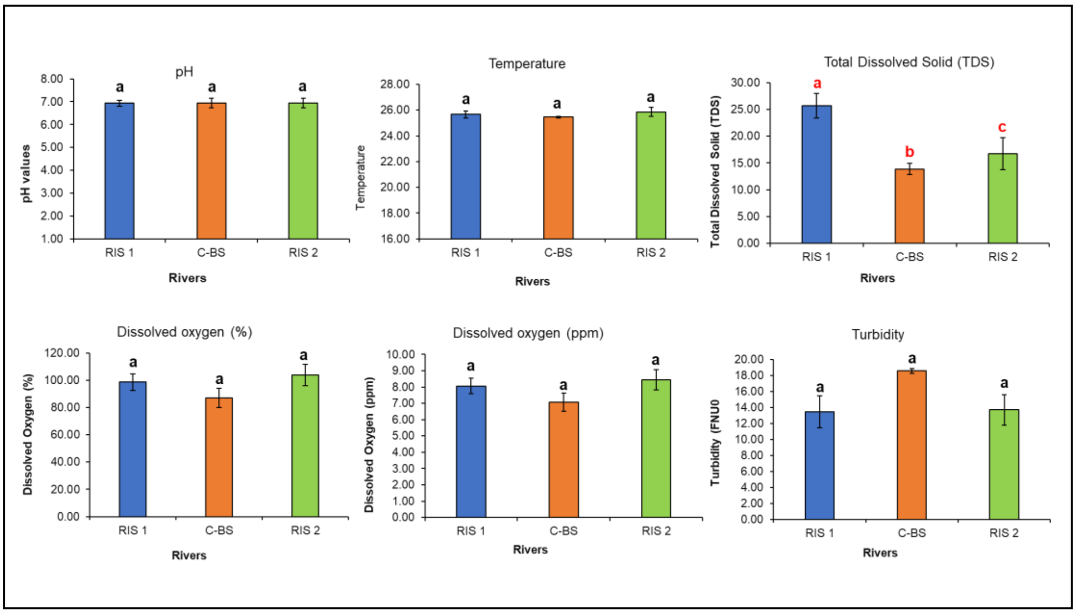

Figure 8. The average values of six water parameters measured and significant difference among the parameters. Similar letters indicated no significant difference (p>0.05).

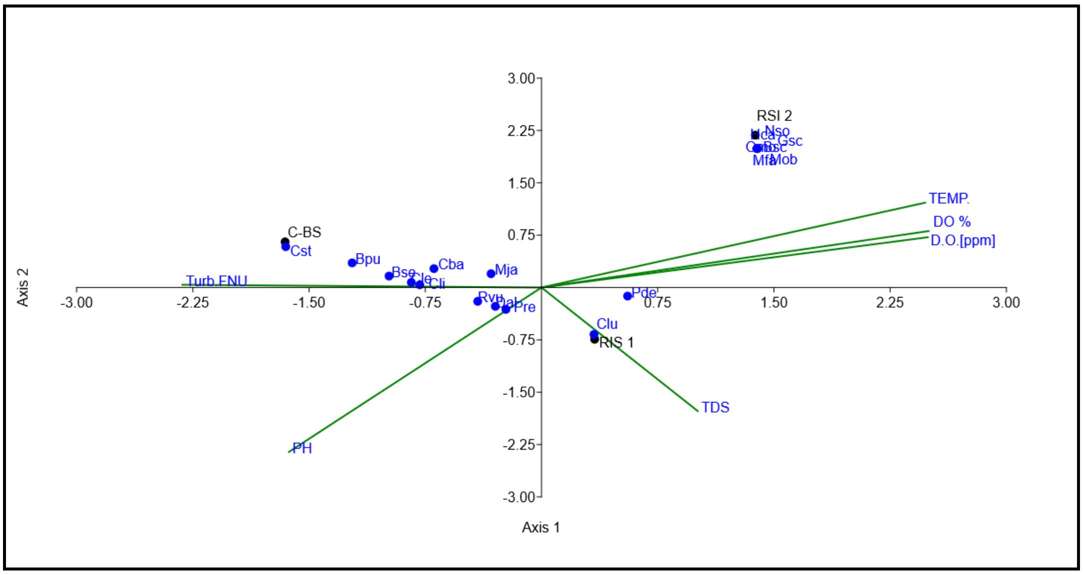

Figure 9. The relationship between fish assemblages and water parameters measured.

Discussion. Despite being surrounded an urban matrix, the streams within the study areas supported low to moderate species richness (19 species across 11 families). This richness is higher than that reported for other urban streams in the region, which recorded fewer species (Jalal et al., 2018; Nurul Fatinizzati et al., 2018). The Cyprinidae, Channidae, and Danionidae families were dominant, consistent with patterns reported for freshwater systems in Peninsular Malaysia (Zakaria-Ismail et al., 2019). The Venn diagram revealed distinct patterns of species sharing and habitat specialization among streams, reflecting underlying environmental heterogeneity. Only three species, *P. deauratus, B. sellifer,* and *R. vulgaris*, were recorded across all streams indicating their role as ecological generalists with broad habitat tolerance. RIS 1 and RIS 2 shared seven species, suggesting comparable environmental conditions or connectivity. However, RIS 1 supported only a single unique species, *Channa lucius*, a forest-associated predator indicative of shaded, minimally disturbed stream conditions (Aqmal-Naser et al., 2024) and also pristine habitat (Aqmal-Naser et al., 2023a). In contrast, RIS 2 exhibited the highest number of unique species (n=7), comprising taxa with diverse ecological traits, including benthic specialists, large-bodied cyprinids, and a non-native species

(*Oreochromis niloticus*), pointing to greater habitat heterogeneity and anthropogenic influence. These patterns demonstrate strong spatial structuring of fish assemblages. The C-BS exhibited no unique species, likely due to its hydrological continuity with RIS 1 as both tributaries flow within the FRIM compound (Azahar et al., 2021). Similar patterns have been observed in the Yangtze River basin, where tributaries and mainstems showed no significant differences in species composition (Shen et al., 2024).

The rank–abundance curves show that *P. deauratus*, *B. sellifer*, and *R. vulgaris* consistently dominated fish assemblages across all three streams, highlighting their ecological generalism and ability to persist under a wide range of environmental conditions. Dominance by small- to medium-bodied cyprinids is a common feature of tropical lowland streams, particularly in systems with moderate flow regimes and heterogeneous habitats that favour species with flexible feeding strategies and broad habitat use (Pusey et al., 2017; Winemiller et al., 2015). Their prominence in RIS 1 and C-BS suggests that reforested riparian conditions, including canopy cover and substrate complexity, continue to support relatively stable and balanced assemblages. The presence of multiple co-dominant species, rather than a single overwhelmingly dominant taxon, indicates moderate disturbance levels rather than severe ecological degradation. Such structures are typical of recovering or semi-natural tropical streams, where habitat complexity promotes niche partitioning among species (Larentis et al., 2022).

C-BS exhibited the highest diversity, suggesting stable assemblage structure and low dominance. Similar patterns of higher fish diversity and evenness in less disturbed stream sections have been observed in more natural or less impacted sites supported richer and more balanced assemblages relative to more disturbed reaches (Yang et al., 2024). Madureira et al. (2024) reported that sites exhibiting higher diversity and low dominance are commonly interpreted as reflecting ongoing ecological recovery in tropical streams. RIS 2, while hosting the highest richness (Margalef=1.75), displayed the lowest evenness (E=0.21) and dominance by a few opportunistic species indicatives of moderate environmental stress.

Comparable patterns where moderately disturbed streams exhibit higher species counts but reduced evenness and dominance by generalist or tolerant taxa have been documented in studies of fish assemblages in urban matrix, highlighting the effect of anthropogenic stressors on community structure (Antoniazzi et al., 2023). RIS 1 showed intermediate diversity (H′=1.18) but the greatest total abundance (1,352 individuals), suggesting ecological balance and resource continuity. A reduced usage RIS1 as recreational within FRIM compound could assist a stable population of fish along the stream. Similar observations, where sites with moderate habitat complexity show high total fish abundance despite intermediate diversity scores, have been reported in tropical stream studies, suggesting that habitat continuity and resource availability can support high biomass even under varying species diversity (Mondal & Bhat, 2020). This pattern aligns with the idea that moderately disturbed streams may retain high richness but lose structural balance, an important nuance when interpreting diversity indices alone.

Clustering analysis and Venn diagrams also revealed clear assemblage differentiation, especially for RIS 2, which supported a distinct fish community. The lack of shared species between RIS 2 and C-BS suggests local-scale environmental filtering rather than regional species pool limitation. Differences in canopy openness, disturbance pressure, and substrate composition likely drove this spatial segregation, reinforcing that small-scale habitat variation matters strongly in urban ecosystems (Aqmal-Naser et al., 2023b). The Bray–Curtis clustering separated the streams into two main assemblage groups, with RIS 1 clustering closely with the C-BS, while RIS 2 formed a distinct group. This pattern indicates that fish assemblage similarity was not strictly aligned with disturbance classification (recreational vs conservation-buffered). Instead, RIS 1 and C-BS likely share similar habitat attributes (e.g., canopy cover, substrate heterogeneity, hydrological stability), which override differences in recreational access. This finding highlights that local habitat conditions or specific niche can be more influential than land-use designation in structuring fish communities in reforested habitat and disturbed landscapes at both spatial and temporal scales (Aqmal-Naser & Ahmad, 2026).

All streams fell within the 95% confidence funnel for average taxonomic distinctness (Δ^+), indicating that none exhibited significant taxonomic over- or under-dispersion. This suggests that, despite differences in species richness and composition, the evolutionary breadth of the assemblages remained intact across the study sites. Such stability implies that recreational use or the urban matrix the streams located have not yet caused selective loss of higher taxonomic lineages, a pattern often observed in heavily degraded streams and rivers (Feio et al., 2023; Toth et al., 2022). The consistency in Δ^+ values suggest that disturbance in the study streams has primarily influenced species identity and dominance, rather than deep taxonomic structure. This aligns with the observation that anthropogenic stressors often lead to species replacement within families or genera, rather than loss of phylogenetic diversity (Xiao et al., 2025; Yang et al., 2024). Maintaining taxonomic breadth is ecologically important, as it preserves functional redundancy and ecosystem resilience, even when assemblage composition shifts. The stable Δ^+ values imply that preventing further habitat simplification and invasive species spread is critical to maintaining taxonomic integrity.

ABC analysis revealed contrasting stability conditions that not fully captured by diversity metrics. RIS 1 exhibited near-overlapping abundance and biomass curves, indicating relatively stable community structure**.** Negative W values in C-BS and RIS 2 suggest dominance by small-bodied species, which may reflect subtle stressors such as sediment accumulation. The higher turbidity in C-BS likely reflects increased natural organic inputs and intact channel processes rather than anthropogenic sedimentation, indicating a functionally healthy but organically enriched system. However, the turbidity did affect fish biomass through prey-predation interactions (Zanghi & Ioannou, 2025) and habitat selection (Miller et al., 2018). Ecologically, most fishes in the small streams of Peninsular Malaysia are dominated by small-sized bodies from the family Cyprinidae and Danionidae (Zain et al., 2021; Aqmal-Naser & Ahmad, 2021). This is also similar to the forest reserve sites attributed to the genera *Barbodes* and *Rasbora* (Fahmi-Ahmad et al., 2025). These taxa, characterized by rapid life cycles, high recruitment, and tolerance to fluctuating environmental conditions, can disproportionately increase numerical abundance without a corresponding rise in biomass, thereby producing an apparent ‘disturbed’ signal even within nominally protected habitats. This apparent contradiction suggests that structural habitat protection does not necessarily equate to functional ecological stability, as natural organic inputs, species composition, and life-history traits may influence biomass patterns independently of disturbance classification. This highlights the value of ABC analysis in detecting early or cryptic ecological stress, even within protected or reforested systems. The non-significant CCA result ($p>0.05$) indicates that measured physicochemical variables explain only a limited portion of assemblage variation. This suggests that fish communities are structured more strongly by habitat complexity and hydrological connectivity than by instantaneous water chemistry (Gebrekiros, 2016; Trovillion et al., 2023).

The occurrence in the study streams of introduced species, which thrive well in modified habitats (Azlan-Hisham et al., 2025), highlights an additional conservation concern, as non-native taxa may alter native fish assemblages through predation (Aqmal-Naser & Ahmad, 2024). Their presence may also modify benthic habitats and food-web structure (Mohammad et al., 2024), potentially reducing native fish biomass and altering community composition even in systems that retain high species richness. In urban forest streams, the establishment of introduced species suggests ongoing connectivity with disturbed catchments and highlights the importance of early detection, pathway management, and long-term monitoring to prevent biotic homogenization and safeguard the ecological integrity of native freshwater assemblages (Pandit et al., 2024). Overall, the current findings indicate that effective management of habitat should prioritize maintaining shaded riparian buffers, minimizing sediment inputs, monitoring invasive species pathways, and protecting headwater and low-access stream segments, as these actions directly influence reach-scale habitat conditions. Conservation outcomes are driven more by local habitat integrity than by zoning or access restrictions alone. Despite recreational use, these streams can remain conservation-relevant when riparian structure, canopy continuity, and hydrological connectivity are retained. The persistence of forest-associated species demonstrates that long-term reforestation and riparian recovery can mitigate anthropogenic impacts on freshwater biodiversity. Collectively, these results indicate that reforested streams can contribute to maintaining freshwater biodiversity surrounded by urban landscapes, emphasizing that such systems can function as ecological analogues of protected habitats when structural habitat integrity is maintained.

Conclusions. This study demonstrates that reforested streams and surrounded by urban matrixes can play an important role in maintaining freshwater biodiversity. Despite varying levels of recreational disturbance, the streams supported moderately diverse fish assemblages with relatively stable taxonomic structure. The findings highlight that local habitat characteristics, including riparian vegetation, canopy cover, and substrate heterogeneity, are more influential in shaping fish assemblages than simple disturbance classification. Although conservation-buffered streams exhibited higher diversity and evenness, patterns of assemblage similarity and ABC analysis indicate that community stability does not always correspond directly with disturbance levels. The presence of introduced species and early signs of ecological stress emphasize the need for continued monitoring and management. Overall, maintaining riparian integrity, habitat complexity, and hydrological connectivity is essential for preserving the ecological function of streams within the urban forest as biodiversity refugia.

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