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RESEARCH ARTICLE

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The significance of the Sikunga Fish Protected Area towards fisheries conservation in the Zambezi River, Namibia

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Abstract

Large cichlids of the genera *Oreochromis*, *Tilapia*, *Serranochromis* and *Sargochromis* are the backbone of the gillnet fishery of the Zambezi River, but *Oreochromis* spp. is currently over-exploited and at risk of local extinction. As a result, the Sikunga Fish Protected Area was established in 2012 to restore a decline in fish stocks of the Zambezi River, but its efficiency has never been assessed. The aim of this study was to compare fish assemblages, abundance and mean sizes between a fish protected area (Sikunga FPA) and nonprotected areas; the Sikunga Buffer Zone and the Lisikili Fished Area on the Zambezi River. Monthly gillnet surveys, using commercial gillnets, were conducted from July to December 2020. Overall catch rate (CPUE) by weight of all species differed significantly among the sampling sites (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = -27.95$, $p = 0.0001$). The highest CPUE was recorded at Sikunga Fish Protected Area (2.85 kg/set ± 0.42 (SE)), followed by the Sikunga Buffer Zone (0.93 kg/set ± 0.16 (SE)) and the Lisikili Fished Area (0.61 kg/set ± 0.14 (SE)). Further analyses showed that the mean sizes of *O. andersonii*, *Coptodon rendalli*, *Serranochromis* and *macrocephalus* were significantly larger at the Sikunga Fish Protected Area than at the Lisikili Fished Area. A particularly interesting observation pertains to a lack of variation in catch rates of *Serranochromis macrocephalus* and mean sizes of *Clarias gariepinus* between the protected and nonprotected areas, signifying that fish protected areas can be species-specific, and not all species in a water body will respond positively to protection. The overall results of this study support the innovation of fish protected areas as a fundamental tool in attaining sustainable management of fishery resources in inland waters of Namibia and other developing countries.

KEYWORDS
co-management, CPUE, Sikunga Buffer Zone, Sikunga FPA, species diversity, Zambezi River

Résumé

Les grands cichlidés des genres *Oreochromis*, *Tilapia*, *Serranochromis* et *Sargochromis* sont l'épine dorsale de la pêche au filet maillant du fleuve Zambèze, mais *Oreochromis* spp. est actuellement surexploitée et menacée d'extinction locale. Ainsi, l'espace protégé des poissons de Sikunga a été créé en 2012 pour rétablir un déclin des stocks

de poissons du fleuve Zambèze, mais son efficacité n'a jamais été évaluée. Le but de cette étude était de comparer les assemblages de poissons, l'abondance et les tailles moyennes entre un espace protégé pour les poissons (Sikunga FPA) et des espaces non protégés; la zone tampon de Sikunga et la zone de pêche de Lisikili sur le fleuve Zambèze. Des enquêtes mensuelles au filet maillant, utilisant des filets maillants commerciaux, ont été menées de juillet à décembre 2020. Le taux de capture global (CPUE) en poids de toutes les espèces différait significativement entre les sites d'échantillonnage (Kruskal-Wallis *H*-test, gamme $\chi^2_{(2)} = -27.95$, $p = 0.0001$). Le CPUE le plus élevé a été enregistré dans l'espace protégé pour les poissons de Sikunga (2.85 kg/set ± SE 0.42), suivi par la zone tampon de Sikunga (0.93 kg/set ± SE 0.16) et la zone de pêche de Lisikili (0.61 kg/set ± SE 0.14). D'autres analyses ont montré que les tailles moyennes de *O. andersonii*, *Coptodon rendalli*, *Serranochromis* et *macrocephalus* étaient considérablement plus grandes dans l'espace protégé pour les poissons de Sikunga que dans la zone de pêche de Lisikili. Une observation particulièrement intéressante concerne l'absence de variation dans les taux de captures de *Serranochromis macrocephalus* et les tailles moyennes de *Clarias gariepinus* entre les espaces protégés et non protégés, ce qui signifie que les espaces protégés pour les poissons peuvent être spécifiques aux espèces-, et que toutes les espèces d'un plan d'eau ne répondront pas positivement à la protection. Les résultats globaux de cette étude soutiennent l'innovation des espaces protégés pour les poissons comme un outil fondamental pour réaliser une gestion durable des ressources halieutiques dans les eaux intérieures de la Namibie et d'autres pays en voie de développement.

1 | INTRODUCTION

Overfishing has adversely affected fish communities worldwide, with large and valuable target species being replaced by smaller, lower-value fishes in the so-called fishing-down process (Welcomme, 1999). As fishing pressure increases, fish populations are subjected to a series of modifications in assemblages, size and abundance because fishers opt to catch the largest or most valuable species in a fish community (Peel, 2012; Saunders et al., 2002; Suski & Cooke, 2007; Welcomme, 1999, 2008). These changes to fish communities driven by high fishing pressure may ultimately alter the structure of food webs and the flux of energy and matter in ecosystems (Andersen & Pedersen, 2010). Protected areas have been proposed as an efficient way to manage fisheries while simultaneously preserving biodiversity in marine ecosystems (Gell & Roberts, 2003). Fish Protected Areas (FPAs) are 'clearly defined aquatic areas developed to protect spawning areas, spawning periods, and nursery sites where juveniles can mature and disperse'(Richardson et al., 2010). Aquatic Protected Areas provide a means of combating fishery-induced evolution by allowing populations to escape the strong size-selective pressure (Baskett et al., 2005). Since some populations, or portions of populations, will still be subject to fishery-induced predation and selection, FPA's may not completely halt this anthropogenic-induced evolution, but may slow the rate of change or maintain genetic variability within a population. Fish Protected Areas can benefit both local biodiversity and local fisheries: target species (adults, juveniles and larvae) are expected to increase in abundance and biomass within the reserve and eventually spill over into nonprotected areas (Koning & McIntyre, 2021). This 'spillover' of fish can be harvested by fishermen to gain a sustainable livelihood as a result of these fish reserves (Koning & McIntyre, 2021). One of the criticisms levelled against FPAs has been the breadth of activities permitted within them (Pittock et al., 2008). Yet, studies of successful establishment of marine protected areas (MPAs) demonstrate that community support for such conservation endeavours is essential to their success (e.g., Jentoft et al., 2012). Primarily, there must be a genuine benefit to local communities derived from the conservation of the target species, which may occur through direct (e.g. increased tourist activity results in increased community-level expenditures), or indirect (e.g. decreased localised industrial pollution inputs increases local crop yields) means. To avoid social conflict resulting from decreased access, these benefits must be both adequately compensated and communicated at the community level (bottom-up support) and must receive legislative and enforcement support from appropriate levels of government (top-down support; Bower et al., 2014).

Although there is little information regarding benefits of FPAs in freshwater systems, some studies have indicated benefits similar to those of MPAs (Kocovsky & Carline, 2001). Unlike in the marine realm, the freshwater conservation community has placed little emphasis on the use of FPAs as a biodiversity protection strategy (Saunders et al., 2002). Some attempts have recently been conducted worldwide, with variable success, to develop FPAs (Keith, 2000; Maitland, 1995; Saunders et al., 2002). Few areas have been created

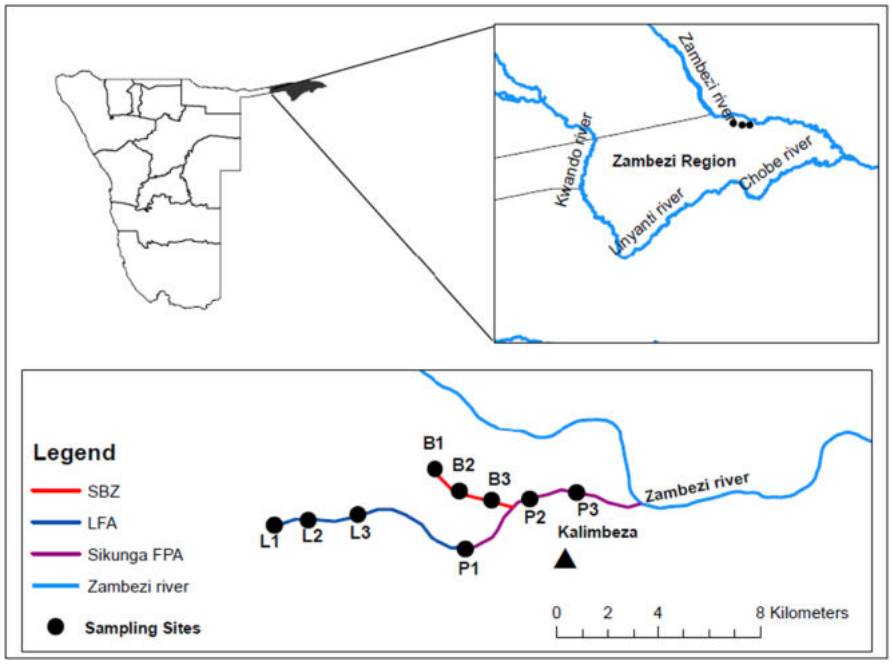

FIGURE 1 Map of the Zambezi Region showing the sampling sites along the Zambezi River, where: P1–P3 denote the Sikunga FPA, B1–B3 denote a tributary of the SBZ and L1–L3 denote the LFA

specifically for freshwater fish, and almost all freshwater protected areas were included 'incidentally' as part of terrestrial reserves (Eybert et al., 1998; Keith, 2000; Self, 2005). However, limited research and earlier studies have shown that freshwater protected areas have been a successful management option for conserving threatened fishes (Cowx, 2002; Lake, 1980; Moyle & Sato, 1991). A nonfishing reserve in a Zimbabwean lake proved successful at increasing both the number and size distribution of several freshwater fish families (Sanyanga et al., 1995), and the establishment of nonfishing refuges has played a large part in the rehabilitation of exploited lake trout populations in lakes (Reid et al., 2001; Schram et al., 1995). In fact, Williams (1991), in his review of preserves and reserves designed explicitly to protect native fishes of the western United States, concludes that most areas evaluated were relatively successful, though success was lower for those larger freshwater systems that were more permeable to invasion by exotics. However, the fact that most freshwater catchments often transverse two or more institutional boundaries, their effective protection may require collaboration of political, social and jurisdictional systems. These shortcomings have resulted in a perceived failure of FPAs as a conservation tool for aquatic systems (Abell et al., 2007).

Large cichlids of the genera *Oreochromis*, *Coptodon*, *Serranochromis* and *Sargochromis* are the major target species of the gillnet fishery on the Zambezi River of Namibia (Hay et al., 2020; Purvis, 2002). However, stocks of the commercially target species such as *Oreochromis andersonii*, *Oreochromis macrochir* and *Coptodon rendalli* are in steep decline and at high risk of local extinction (Tweddle et al., 2015). The most frequently cited causes of decline are as follows: use of destructive and efficient gears, commercialisation and poor law enforcement. Therefore, effective management is urgently required to prevent the overfishing of large cichlids in the Zambezi River where effort is increasing at an alarming rate (Peel, 2012; Tweddle et al., 2015). The solutions proposed by managers for this critical problem rely on reducing the fishing capacity through 'traditional' fisheries measures (e.g. reducing fishing effort, a permit system and regulating fishing equipment). Despite these mechanisms, the general decline of large growing cichlids such as *Oreochromis andersonii*, *Oreochromis macrochir* and *Coptodon rendalli* has been observed (Tweddle et al., 2015), calling for effective management measures. The use of local freshwater protected areas (FPAs) appears to be a relevant way to reconcile these aspects and to respond to both global management constraints. As a result, the Sikunga Fish Protected Area (hereafter Sikunga FPA) was established on the Zambezi River in 2012 in hope that it would maintain and yield high fish species' richness and abundance (Tweddle et al., 2015).

The first study to explore the effectiveness of the Sikunga FPA was conducted by Simasiku et al. (2017); however, the results of this study were inconclusive. Consequently, the performance of the Sikunga FPA and whether it is achieving its intended aims remained uncertain. A follow-up study (this study), based on emulated gillnets of the subsistence fishermen, was initiated to determine whether the Sikunga FPA is meeting its objectives as a management strategy tailored to minimise further depletion of the most commercially target species in the Zambezi River. This study aimed to compare fish assemblages, diversity, abundance and size structure between a fish protected area (Sikunga FPA) and nonprotected areas; the Sikunga Buffer Zone (hereafter Sikunga BZ) and the Lisikili Fished Area (hereafter Lisikili FA) on the Zambezi River. The following questions were addressed: (Abah et al., 2018) Is there a difference in fish composition and diversity among the Sikunga FPA, the Sikunga BZ and the Lisikili FA? (Hay & van der Waal, 2009). Is there a difference in fish abundance (expressed as Catch Per Unit Effort [CPUE]) and mean sizes of the most abundant species among the three sampling sites? (Allan et al., 2005) Do all species respond to protection in the Zambezi River?

2 | MATERIAL AND METHODS

2.1 | Study area

The study was conducted on the Zambezi River, located in the Zambezi Region of Namibia (Figure 1). The region borders Botswana in the south, Angola and Zambia in the north, and Zimbabwe in the east. The Zambezi Region is home to two perennial rivers: the Kwando/Linyanti River to the west and the Zambezi/Chobe River to the east. The Zambezi/Chobe is a highly pulsed and expansive river in terms of water volume during the flooding season. The topography of the Zambezi Region is flat terrain with an altitude ranging between 1100 m in the west and 930 m in the east. Seasonal floodwater transverses from the river catchments and spreads laterally by overflow, creating a single, large floodplain in the eastern Zambezi Region (Lubbers et al., 1990; Mendelsohn et al., 1997). The Zambezi/Chobe River usually reaches its peak flow between March and May, after which the water recedes until the end of September. During the dry months (November–April), the floodplains are dry and covered in terrestrial grasses.

2.2 | Sampling sites

The study was conducted in three sections on the Zambezi River; one section of the 12 km stretch within the Sikunga FPA, the second section of 1.3 km within the Sikunga BZ, and the third section of 8 km within the Lisikili FA (Figure 1). The Sikunga BZ includes the side channels of the Sikunga FPA, while the Lisikili FA is an extension of the Sikunga channel, located at a distance of about 5 km west of the Sikunga FPA. These three sites fall within the same vicinity but differ in their levels of management, in that all fishing activities, except for scientific research and angling, are prohibited at the Sikunga FPA, while selective gears such as regulated static multifilament gillnets and hook and line are allowed at the Sikunga BZ and the Lisikili FA. However, the Sikunga BZ is partially subjected to strict management protocols (i.e. daily patrols) that are devoted for protecting the Sikunga FPA.

2.3 | Gillnet surveys

Monthly gillnet surveys were conducted from July to December 2020. The authors used gillnets similar to those used in the subsistence and commercial fishery on the Zambezi/Chobe floodplain. Four monofilament gillnets of variable mesh sizes (76, 89, 102 and 114 mm) were considered for this study. Individual nets were 100 metres long and about 38 meshes deep. The nets were deployed in the offshore waters at approximately 18.00 h at twilight and retrieved at 06.00 h the next morning. Setting sites were restricted to open waters with comparable depth free of vegetation to minimise the confounding effect of environmental heterogeneity among the sampling sites. Each net was set for two consecutive nights in each site per month, translating to six gillnet nights per event, and 36 gillnet nights for the entire study period. The physicochemical parameters of the water were measured in situ per site as a proxy of water quality. Dissolved oxygen (mg/L), conductivity (μS/cm), pH and temperature (°C) were measured using a multifunction sensor HQ40d portable meter, and the average measurement of each parameter per sampling site was computed. On landing, fish caught per net were removed and sorted into species per mesh size. All fish were measured to the nearest millimetre (mm) total length (TL) or fork length (FL), depending on the species, and wet weight recorded to the nearest gram (g).

2.4 | Data analyses

2.4.1 | Catch composition

An Index of Relative Importance was calculated to determine the most important species captured by each fishing gear by number, weight and frequency of occurrence, and was calculated as:

$$\mathrm{IRI} = (\%\mathrm{N} + \%\mathrm{W}) \times (\%\mathrm{FO}) \quad (1)$$

where %N and %W are the percentage contribution of each species by number and by weight to the total catch and %FO is the percentage frequency of occurrence of each species in the total number of net settings.

2.4.2 | Species diversity

Species diversity was examined as species richness (S = number of species) and composite diversity, which integrates both richness and evenness (Shannon–Wiener H') (Kolding, 1999; Lubbers et al., 1990). These parameters were employed to describe the diversity of the assemblages of fish among the sampling sites. Species diversity based on species abundance data was calculated using the following formulae in Pasgear 2. v 2.2 (Kolding, 1999):

$$\text{Shannon – Wiener Index}\, H' = -\sum P_i * \log_e(P_i) \quad (2)$$

where p^i is the proportion of individuals found in the ith species.

2.4.3 | Catch per unit effort (CPUE)

Relative fish abundance was expressed as CPUE and is expressed in wet weight. The CPUE was calculated as:

$$\mathrm{CPUE} = \mathrm{C}_i / \mathrm{E}_i \quad (3)$$

where C_i is the catch of species i (in weight) and E_i is the effort expended to obtain i. As a result, CPUE was standardised to fish biomass/net.

2.5 | Statistical analysis

Before analysis, all data on physicochemical parameters, CPUE and mean sizes were pooled across the sampling sites and checked for normality and homogeneity of variance using Kolmogorov–Smirnov and Levene's test in SPSS v26. Where data were found to be skewed, a square root transformation was performed to normalise the CPUE and mean sizes data. Where the square root transformation could not satisfy the assumption of normality, logarithmic (X = Ln (X+I)) transformation was applied instead. Subsequently, the one-way ANOVA test was applied to determine differences in spatial physicochemical parameters, CPUE and fish mean sizes. In cases where data failed to meet the normality assumption, the equivalent nonparametric Kruskal–Wallis test was applied. Significant associations at $p < 0.05$ were identified using the Bonferroni correction test. Length structure histograms were developed to depict the size structure of the most dominant species by sampling site.

2.6 | Multivariate analysis on fish assemblages among the sampling sites

Multivariate analysis of variance was carried out to compare fish assemblages among the sampling sites (Anderson et al., 2008; Simasiku, 2019). To visualise multivariate patterns, a cluster analysis (group average) employing the Bray–Curtis similarity index was performed on the standardised abundance values of species using the multivariate techniques in PRIMER *v7* (Clarke & Warwick, 1994). The data were transformed by applying a square root transformation before the cluster analysis to avoid overemphasis of the most abundant species by sampling site (Clarke & Warwick, 1994). Multidimensional scaling (MDS) ordination analysis was performed on the same data as the cluster analysis. Similarity profile routine (SIMPROF) was used to indicate the average contribution of each species to the dissimilarity (discriminating species) among the sampling sites (Clarke & Warwick, 1994). However, only the 10 species contributing most to the similarity and dissimilarity are reported in this study.

3 | RESULTS

3.1 | Physicochemical analyses

Mean values of conductivity were significantly higher at the Sikunga BZ than the Lisikili FA (ANOVA, $df = 2$, $p = 0.001$) but similar to the Sikunga FPA ($p > 0.05$) (Table 1). However, dissolved oxygen and pH were similar in all the sampling sites ($p > 0.05$), as were mean temperature observed at the sampling sites ($p > 0.05$).

3.2 | Catch composition by sampling sites

The catch composition by sampling site in terms of percentage Index of Relative Importance (%IRI) of all species sampled at Sikunga FPA, the Sikunga BZ and the Lisikili FA is summarised in Table 2. A total of 1345 individual fish, representing 24 species and eight families, were caught in the study area between July and December 2020.

3.3 | Sikunga fish protected area

A total of 789 individual fish, representing 22 species and eight families, were recorded at Sikunga FPA (Table 2). The most numerous species was *Schilbe intermedius* (29.8%), while the large predatory catfish, *Clarias gariepinus* contributed the most in weight (16.4%). The five most important (IRI) species, accounting for 79.0%, were *S. intermedius* (31.4%), *Serranochromis angusticeps* (15.9%), *C. gariepinus* (10.1%), *Serranochromis macrocephalus* (8%) and *Oreochromis andersonii* (7%).

3.4 | Sikunga Buffer Zone

A total of 401 individual fish, representing 21 species and eight families, were recorded at the Sikunga BZ (Table 2). The silver catfish, *S. intermedius*, was the most numerous species (34.9%) and the predatory catfish, *C. gariepinus*, accounted for the most in weight (39.6%). The five most important species (IRI), accounting for 91.7%, were *C. gariepinus* (32.8%), *S. intermedius* (23.3%), *S. macrocephalus* (22.9%), *S. angusticeps* (9.2%) and *Clarias ngamensis* (3.5%).

3.5 | Lisikili Fished Area

A total of 155 individual fish, representing 12 species and two families, were recorded at the Lisikili FA (Table 2). *Serranochromis macrocephalus* was the most dominant species by number (21.9%) and weight (21.5%). The five most important species (IRI), accounting for 87.2%, were *S. macrocephalus* (38.9%), *C. rendalli* (27.3%), *S. angusticeps* (7.6%), *O. andersonii* (7.6%) and *C. gariepinus* (2.6%).

TABLE 1 Mean and standard deviation of the water physicochemical parameters grouped by sampling sites along the Zambezi River, sampled between July and December 2020

	Sikunga FPA	Sikunga BZ	Lisikili FA	*p*
Temperature (°C)	24.9 ± 3.26	25.8 ± 1.8	24.9 ± 3.26	0.12
Dissolved oxygen (mg/L)	6.17 ± 1.31	5.9 ± 1.4	6.9 ± 0.5	0.06
pH	7.32 ± 0.63	7.95 ± 0.2	7.57 ± 0.7	0.14
Conductivity (μS/cm)	74.17 ± 25.37[a]	75.16 ± 23.3[a]	70.0 ± 21.6[b]	0.001*

*Denotes significant difference among sites.

TABLE 2 Gillnet catch composition in percentage number (%N), percentage weight (%W) and percentage Index of Relative Importance (%IRI) of all fish species caught at Sikunga FPA, the Sikunga BZ and Lisikili FA, sampled between July and December 2020

	Sikunga FPA			Sikunga Buffer Zone			Lisikili Fished Areas		
Species	**%No**	**%W**	**%IRI**	**%No**	**%W**	**%IRI**	**%No**	**%W**	**%IRI**
Schilbe intermedius	29.8	5.2	31.4	34.9	5.8	23.3	–	–	–
Serranochromis macrocephalus	7.4	10.6	8.0	14.0	17.8	22.9	21.9	21.5	38.9
Clarias gariepinus	4.8	16.4	10.1	15.0	39.6	32.8	9.7	20.6	5.8
Serranochromis angusticeps	10.4	14.2	15.9	8.0	10.0	9.2	9.0	7.9	7.6
Copodon rendalli	8.1	7.6	7.0	4.5	3.0	1.6	21.3	14.3	27.3
Oreochromis andersonii	7.1	10.4	7.0	1.0	1.5	0.2	11.0	6.1	7.6
Serranochromis altus	7.1	7.8	5.2	1.7	1.0	0.3	3.9	2.4	0.8
Hydrocynus vittatus	4.7	4.5	4.4	3.0	2.0	1.2	–	–	–
Clarias ngamensis	1.0	4.6	1.1	3.5	9.4	3.5	4.5	8.7	1.7
Oreochromis macrochir	3.5	3.9	3.0	0.2	–	–	11.6	8.5	9.0
Sargochromis carlottae	4.7	3.7	2.3	3.5	2.6	1.6	0.6	0.5	0.1
Sargochromis codringtonii	1.9	1.5	0.6	4.2	4.3	2.6	1.3	0.9	0.3
Mormyrus lacerda	3.3	5.0	2.9	0.5	0.6	0.1	–	–	–
Synodontis spp.	2.2	0.5	0.5	2.0	0.2	0.3	–	–	–
Hepsetus cuvieri	1.3	2.1	0.6	0.7	0.5	0.1	–	–	–
Sargochromis giardi	0.9	1.8	0.4	0.2	0.6	–	0.6	1.0	0.1
Pharyngochromis acuticeps	1.0	0.1	0.1	1.5	0.2	0.3	–	–	–
Clarias stappersii	–	–	–	0.5	1.0	0.1	4.5	7.6	0.8
Hemichromis elongatus	0.3	–	–	0.5	0.1	–	–	–	–
Tilapia sparrmanii	0.1	–	–	0.2	–	–	–	–	–
Marcusenius altisambesi	0.3	–	–	–	–	–	–	–	–
Serranochromis jallae	0.1	0.1	–	–	–	–	–	–	–
Parauchenoglanis ngamensis	0.1	–	–	–	–	–	–	–	–
Brycinus lateralis	–	–	–	0.2	–	–	–	–	–

TABLE 3 Species richness (S), diversity (H') and evenness (J') indices for gillnet catches at Sikunga FPA, the Sikunga BZ and the Lisikili FA, surveyed between July and December 2020

Indices	Sikunga FPA	Sikunga Buffer Zone	Lisikili Fished Areas	*p*
S	22	21	12	
H'	2.18 ± 0.09 (SE)^{a}	1.77 ± 0.08 (SE)^{ab}	1.34 ± 0.28 (SE)^{b}	0.0001*
J'	0.80 ± 0.52 (SE)	0.7 ± 0.32 (SE)	0.90 ± 1.12 (SE)	0.18

Note: Different letters denote significant differences between sites.

*Denotes significant differences among the sampling sites.

3.6 | Species richness and diversity by sampling sites

Species richness was higher at Sikunga FPA ($S = 22$), followed by the Sikunga BZ ($S = 21$) and Lisikili FA ($S = 12$) (Table 3). However, species evenness was similar among the sampling sites (Kruskal–Wallis H-test, $\chi^2(2) = 3.39$, $p = 0.18$) while species diversity (H') varied significantly among the sampling sites (Kruskal–Wallis H-test, $\chi^2(2) = 5.40$, $p = 0.02$) (Table 3). Sikunga FPA recorded the highest species diversity (2.18 ± 0.09 (SE)), followed by the Sikunga BZ (1.77 ± 0.08 (SE)) and Lisikili FA (1.34 ± 0.28 (SE); Table 3). The Kruskal–Wallis pairwise (Bonferroni correction) test showed a significant difference in species diversity between Sikunga FPA and the Lisikili FA (Kruskal–Wallis H-test, $\chi^2(2) = 9.00$, $p = 0.01$), but no significant difference was observed between Sikunga FPA and the Sikunga BZ (Kruskal–Wallis H-test, $\chi^2(2) = 6.50$, $p = 0.11$) or between the Sikunga BZ and the Lisikili FA (Kruskal–Wallis H-test, $\chi^2(2) = -2.50$, $p = 1.000$) (Table 3). Cichlidae was the most speciose family, with 15 species recorded at Sikunga FPA, 13 species at the Sikunga BZ and nine (Bavins et al., 2000) species at the Lisikili FA.

3.7 | Fish community assemblages by sampling sites

A dendrogram show distinct assemblages of fish caught among the three sampling sites (Sikunga FPA, the Sikunga BZ and the Lisikili FA) on the Zambezi River (Figure 2). Hierarchical cluster analysis revealed two groups with distinct community structures. According to SIMPROF test results, the fish community structure was similar between Cluster 2; Sikunga FPA and the Sikunga BZ, a cluster that is significantly different from Cluster 1; Lisikili FA (Figure 2). The average dissimilarity between Sikunga FPA and the Sikunga BZ is 28.4%, and the dissimilarity was driven by the average abundance of *C. gariepinus*, *S. macrocephalus*, *O. andersonii*, *S. altus* and *C. rendalli*. The average dissimilarity between Sikunga FPA and the Lisikili FA is 52.3% and this was further driven by the average abundance of *S. intermedius*, *C. rendalli*, *O. macrochir*, *O. andersonii* and *S. macrocephalus*. The average dissimilarity between Sikunga BZ and the Lisikili FA is 51.6%, and this was driven by the average abundance of *S. intermedius*, *S. macrocephalus*, *C. rendalli*, *O. macrochir* and *H. vittatus*.

3.8 | Catch per unit effort (CPUE) by sampling sites

Overall CPUE by weight of all large tilapia species differed significantly among the sampling sites (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = -27.95$, $p = 0.000$) (Figure 3). Sikunga FPA recorded the highest CPUE (mean 2.85 kg/set ± 0.42 (SE)), followed by the Sikunga BZ (0.93 kg/set ± 0.16 (SE)) and the Lisikili FA (0.61 kg/set ± 0.14 (SE)). The Kruskal–Wallis pairwise (Bonferroni correction) test showed a significant difference in CPUE of all large growing tilapias combined among the Sikunga FPA, the Sikunga BZ and the Lisikili FA (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = 33.07–44.92$, $p = 0.000$) but no difference between the Sikunga BZ and Lisikili FA (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = -11.849$, $p = 0.480$) (Figure 3).

Further analyses on a species level revealed that the CPUE of *O. andersonii* differed significantly among the sampling sites (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 30.70$, $p < 0.05$) (Figure 4), being highest at Sikunga FPA (0.49 ± 0.16 (SE) kg/net.night^{-1}) followed by the Lisikili FA (0.06 ± 0.02 (SE) kg/net.night^{-1}) and the Sikunga BZ (0.03 ± 0.02 (SE) kg/net.night^{-1}) (Figure 4). The Kruskal–Wallis pairwise (Bonferroni correction) test showed a significant difference in CPUE of *O. andersonii* between Sikunga FPA and the Sikunga BZ and between Sikunga FPA and the Lisikili FA (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = 17.90$, $p = 0.003–0.02$), but no difference in CPUE was observed between the Sikunga BZ and the Lisikili FA (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = 7.000$, $p = 0.16$; Figure 4).

Catch rates of *C. rendalli* also varied significantly among the sampling sites (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 148$, $df = 3$, $p < 0.0001$; Figure 4). The highest catch rate of this species was recorded at Sikunga FPA (0.35 ± 0.10 (SE) kg/net.night^{-1}), followed by the Lisikili FA (0.14 kg/set ± 0.05 (SE)), and less so at the Sikunga BZ (0.07 kg/set ± 0.03 (SE)) (Figure 4). The Kruskal–Wallis pairwise (Bonferroni correction) test showed a significant difference in CPUE of *C. rendalli* between Sikunga FPA and the Sikunga BZ (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 17.31$, $p = 0.018$), but no significant difference was observed between the Sikunga BZ and the Lisikili FA (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = 7.20–10.108$, $p = 0.379$) or between Sikunga FPA and the Lisikili FA (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = 7.204–10.108$, $p = 0.379–0.828$).

Catch rates of *S. macrocephalus* were not significantly different among the sampling sites (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 3.28$, $p = 0.19$), whereas those of *S. angusticeps* varied significantly (Kruskal–Wallis test *H*-test, $\chi^2_{(2)} = 148$, $df = 3$, $p = 0.0001$) (Figure 4). The highest CPUE of *S. angusticeps* was recorded at Sikunga FPA (0.66 kg/set ± 0.14 (SE)), followed by the Sikunga BZ (0.23 kg/set ± 0.06 (SE)) with the Lisikili FA being the lowest (0.08 kg/set ± 0.03 (SE)). The Kruskal–Wallis pairwise (Bonferroni correction) test showed a significant difference in CPUE of this species between Sikunga FPA and the Sikunga BZ (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = -12.46–16.771$, $p = <0.001–0.048$), and between Sikunga FPA and the Lisikili FA (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 29.246$, $p < 0.001$), but no significant difference in CPUE of this species was observed between the Sikunga BZ and the Lisikili FA (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = -12.475$, $p = 0.261$).

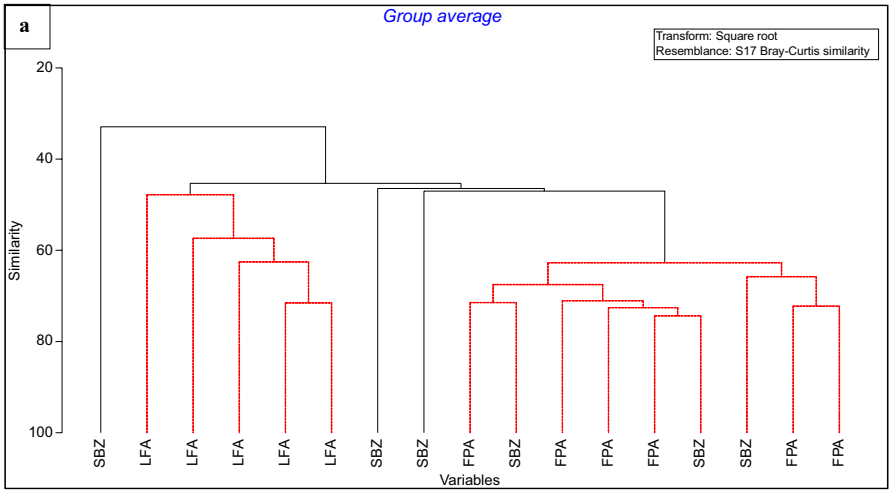

FIGURE 2 Dendrogram for hierarchical clustering analysis (a) based on species abundance by sampling sites; Sikunga FPA, the SBZ and the LFA, surveyed on the Zambezi River between July and December 2020

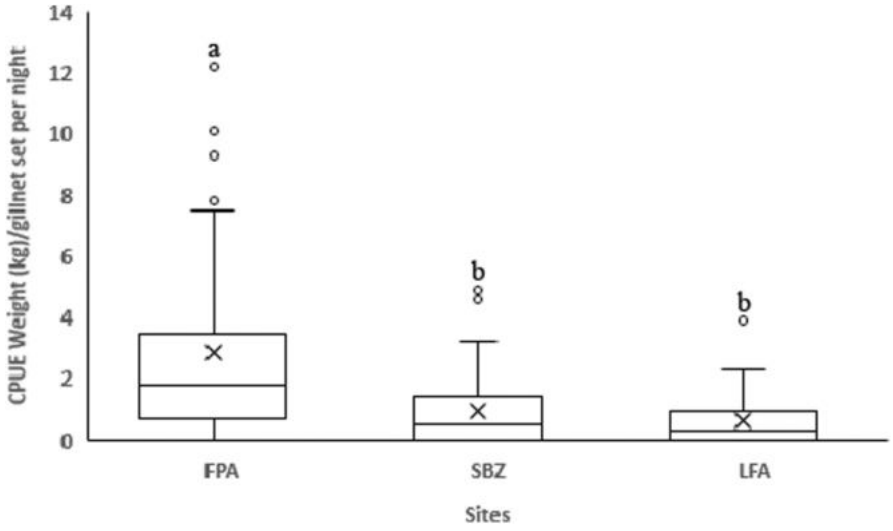

FIGURE 3 Box and whisker plot of the gillnet CPUE by weight (kg) of all species combined in the Sikunga FPA, the SBZ and the LFA, sampled between July and December 2020. Different letters denote significant differences between sites

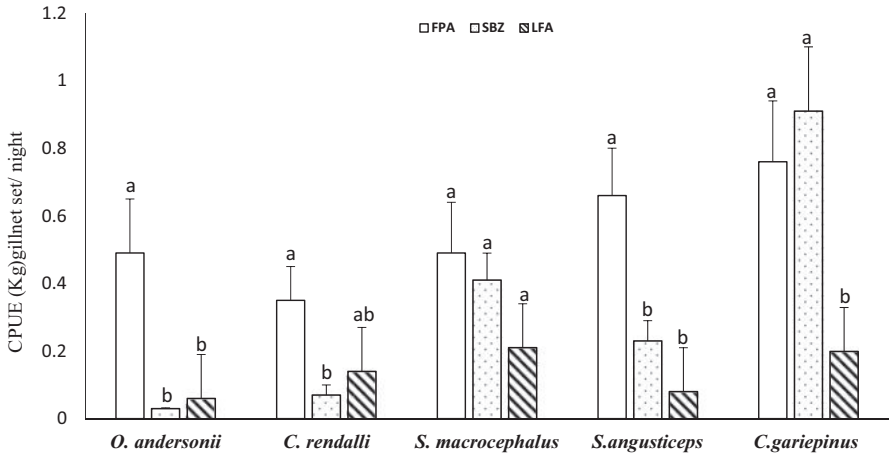

FIGURE 4 CPUE by weight (kg) of *O. andersonii*, *C. rendalli*, *S. macrocephalus*, *S. angusticeps* and *C. gariepinus* of all mesh sizes combined at Sikunga FPA, the SBZ and the LFA, surveyed between July and December 2020. Different letters denote significant differences between sites

Similarly, the catch rates of *C. gariepinus* differed significantly among the sampling sites (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 23$, $df = 3$, $p = 0.0001$) (Figure 4). The highest CPUE of this species was recorded at the Sikunga BZ (0.91 kg/set ± 0.19 (SE)), followed by Sikunga FPA (0.76 kg/set ± 0.20 (SE)) with the Lisikili FA being the lowest (0.21 kg/set ± 0.13 (SE)). The Kruskal–Wallis pairwise (Bonferroni correction) test showed a significant difference in CPUE of *C. gariepinus* between Sikunga FPA and the Lisikili FA (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 20.777$, $p = 0.008$) and between the Sikunga BZ and the Lisikili FA (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = -23.423$, $p = 0.002$) but no difference was observed between Sikunga FPA and the Sikunga BZ (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = -2646$, $p = 1.000$).

3.9 | Catch rates by mesh size and sampling sites

Catch rates of all large tilapia species combined for the 76 mm mesh size (CPUE) differed significantly among the sampling sites (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = -227$, $df = 34$, $p = 0.0001$) (Figure 5). The highest CPUE in 76 mm mesh size was recorded at Sikunga FPA (6.08 ± 1.40 (SE) kg/net.night^{-1}) followed by the Sikunga BZ (2.99 ± 0.78 (SE) kg/net.night^{-1}) and the Lisikili FA (2.70 ± 0.96 kg/net.night^{-1}) (Kruskal–Wallis ANOVA; $p < 0.05$) (Figure 5). The Kruskal–Wallis pairwise (Bonferroni correction) test showed a significant difference in CPUE of the 76 mm mesh size between Sikunga FPA and the Lisikili FA (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 14.250$, $p = 0.05$) but no difference was observed between Sikunga FPA and the Sikunga BZ or between the Lisikili FA and the Sikunga BZ (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = -7.708$–6.542, $p = 1.00$–0.11).

Catch rates of all large tilapia species combined for the 89 mm mesh size (CPUE) also differed significantly among the sampling sites (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = 218$, $p = 0.0001$), with the highest CPUE recorded at Sikunga FPA (5.10 ± 1.06 (SE) kg/net.night^{-1}), followed by the Sikunga BZ (2.98 ± 0.74 (SE) kg/net.night^{-1}) and the lowest at the Lisikili FA (0.66 ± 0.15 kg/net.night^{-1}) (Figure 5). The Kruskal–Wallis pairwise (Bonferroni correction) test showed a significant difference in CPUE of the 89 mm mesh size between Sikunga FPA and the Lisikili FA (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 14.250$, $p = 0.002$) but no significant difference was observed between Sikunga FPA and the Sikunga BZ or between the Lisikili FA and the Sikunga BZ (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = -7.708$–6.542, $p = 0.211$–0.322).

Similarly, the catch rates of all large tilapia species combined for the 102 mm mesh size differed significantly among the sampling sites (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = 194$, $p = 0.0001$), with the highest CPUE recorded at Sikunga FPA (4.4 ± 0.83 kg/net.night^{-1}), followed by the Sikunga BZ (1.66 ± 0.49 (SE) kg/net.night^{-1}), and the Lisikili FA (0.54 ± 0.27 (SE) kg/net.night^{-1}; Figure 5). The Kruskal–Wallis pairwise (Bonferroni correction) test showed a significant difference in CPUE of the 102 mm mesh size between Sikunga FPA and the Lisikili FA, and between Sikunga FPA and the Sikunga BZ (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = 10.000$–16.617, $p = 0.039$) but no significant difference was observed between the Sikunga BZ and the LFA (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = -6.617$, $p = 0.352$).

Catch rates of all large tilapia species combined for the 114 mm mesh size also differed significantly among the sampling sites (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = 208$, $p = 0.001$), with the highest CPUE recorded at Sikunga FPA (2.9 ± 0.64 (SE) kg/net.night^{-1}), followed by the Sikunga BZ (1.52 ± 0.74 (SE) kg/net.night^{-1}), with the lowest at the Lisikili FA (0.001 kg/net.night^{-1}) (Figure 5). The Kruskal–Wallis pairwise (Bonferroni correction) test showed a significant difference in CPUE between Sikunga FPA and the Lisikili FA, and between the Sikunga BZ and the Lisikili FA (Kruskal–Wallis *H*-test, range $\chi^2_{(2)} = -11.485$–16.875, $p = 0.000$–0.016), but no significant difference was observed between Sikunga FPA and the Sikunga BZ (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 5.417$, $p = 0.502$).

3.10 | Size distribution of the dominant species by sampling sites

Overall mean fish sizes of all species combined (large cichlids only) differed significantly among the sampling sites (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 55.3$, $p = 0.000$) (Figure 6). Sikunga FPA recorded the largest mean size (25.2 ± 9.0), followed by the Sikunga BZ (24.7 ± 7.5) and the Lisikili FA (21.6 ± 6.0) (Figure 6). However, no difference in overall mean sizes was observed between the Sikunga FPA and the Sikunga BZ ($p > 0.05$).

Further analyses on the size structure of the five most dominant species by sampling sites is illustrated in Table 4. *Oreochromis andersonii* caught at Sikunga FPA (27.07 ± 0.65 (SE) cm TL) were significantly larger than those caught at the Lisikili FA (19.26 ± 0.49 (SE) cm) (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 31.420$, $p = 0.000$), but similar in length to those caught at the Sikunga BZ (28.15 ± 0.96 (SE) cm TL; Kruskal–Wallis *H*-test, $\chi^2_{(2)} = -4.286$, $p = 1.000$) (Table 4). Individuals of *C. rendalli* caught at Sikunga FPA (21.37 ± 0.39 (SE) cm TL) were significantly larger than those caught at the Lisikili FA (18.92 ± 0.92 (SE) cm TL) (Kruskal–Wallis *H* test, $\chi^2_{(2)} = 19.929$, $p = 0.016$), but were similar in length to those caught at the Sikunga BZ (19.74 ± 0.43 (SE) cm TL) (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 18.767$, $p = 0.104$; Table 4). Similarly, individuals of *S. macrocephalus* caught at Sikunga FPA (25.99 ± 0.46 (SE) cm TL) were significantly larger than those at the Lisikili FA (23.51 ± 0.68 (SE) cm TL; Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 31.601$, $p = 0.002$), but were similar in length to those caught at the Sikunga BZ (25.58 ± 0.48 (SE) cm TL) (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 6.149$, $p = 1.000$; Table 4). However, there were no significant differences in individual sizes of *S. angusticeps* and *C. gariepinus* among the sampling sites (Kruskal–Wallis *H*-test, $\chi^2_{(2)} = 1.089$, $p = 0.578$–1.00) (Table 4).

4 | DISCUSSION

To date, the use of closed areas designed to shield freshwater biota from natural and anthropogenic disturbances has been quite slow relative to the marine environment (Crivelli, 2002; Srinoparatwatana & Hyndes, 2011; Suski & Cooke, 2007). The premise of this study was to determine whether the Sikunga FPA as a management tool is meeting its objectives of minimise the impact of over-exploitation of fish stocks in the Zambezi River. Parallel experiments revealed that both species diversity and richness values were significantly higher at Sikunga FPA than at Sikunga BZ and Lisikili FA. The differences in fish diversity and richness between the Sikunga FPA and the unprotected areas can be linked to differences in their management protocols, whereas all forms of exploitation including fishing are strictly prohibited at Sikunga FPA, while fishing activities in surrounding waters are high. Poor enforcement of existing fisheries regulations by the local and regional authorities may also be attributed to low fish diversity in the unprotected areas, especially at Lisikili FA. For instance, the use of prohibited beach seine nets and monofilament

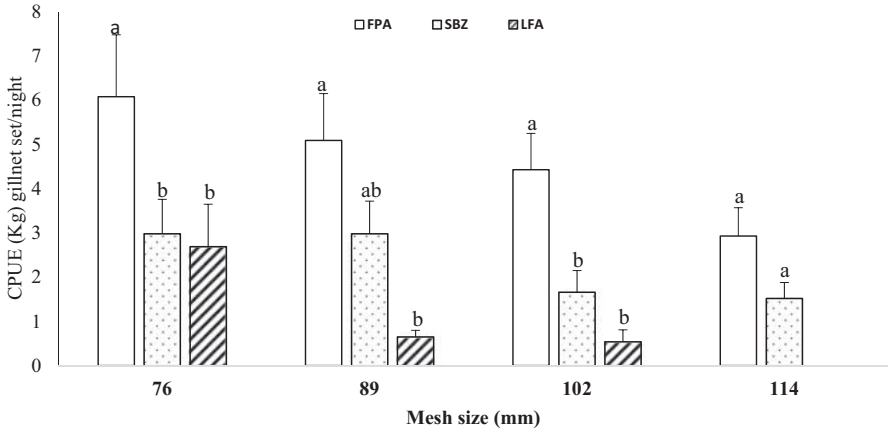

FIGURE 5 Gillnet CPUE by weight of all species combined per mesh size in Sikunga FPA, the SBZ and the LFA, sampled between July and December 2020. Different letters denote significant differences between sites

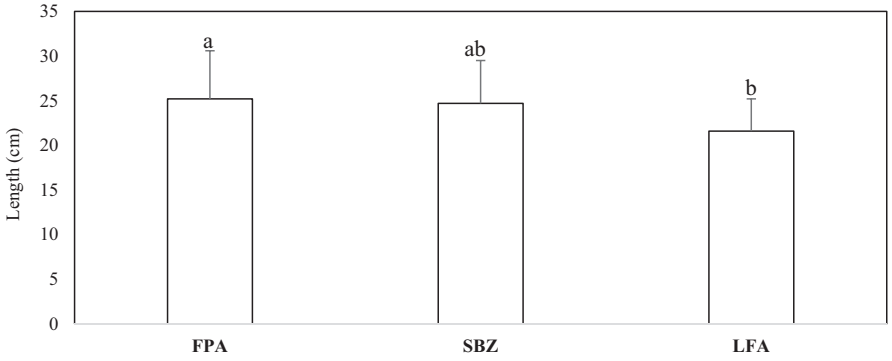

FIGURE 6 Fish mean sizes of all species combined for all mesh sizes combined in the Sikunga FPA, the SBZ and the LFA, sampled between July and December 2020. Different letters denote significant differences between sites

TABLE 4 Fish mean sizes of *O. andersonii*, *C. rendalli*, *S. macrocephalus*, *S. angusticeps* and *C. gariepinus* for all mesh sizes combined in the Sikunga FPA, Sikunga BZ and Lisikili FA, sampled between July and December 2020

	Sikunga FPA	Sikunga BZ	Lisikili FA	*p*
O. andersonii	27.07 ±0.65^{a}	28.15 ±0.96^{a}	19.26 ±0.49^{b}	0.001*
C. rendalli	21.37 ±0.39^{a}	20.74 ±0.43^{a}	18.92 ±0.92^{b}	0.014*
S. macrocephalus	25.99 ±0.46^{a}	25.58 ±0.48^{a}	23.51 ±0.68^{b}	0.003*
S. angusticeps	26.40 ±0.76^{a}	26.10 ±0.92^{a}	24.18 ±0.49^{a}	0.578
C. gariepinus	43.25 ±1.47^{a}	42.43 ±0.79^{a}	40.53 ±0.79^{a}	1.000

Note: Different letters denote significant differences between sites.

*Denotes significant difference among sites.

gillnets was confirmed at Lisikili FA during surveys that could relate to the low species diversity recorded at Lisikili FA. Insignificant differences in the environmental parameters between protected and unprotected areas ruled out their effects on fish species diversity and richness among the sampling sites. Our results on water quality among the sampling sites revealed that the values for the critical water parameters, such as temperature, dissolved oxygen and pH were similar among the sampling sites and were within the safe limits for healthy aquatic systems (Abah et al., 2018; Swingle, 1969).

According to the SIMPROF test results, the fish community structure was similar between the Sikunga FPA and the Sikunga BZ but differed between Sikunga FPA and the Lisikili FA. Similarities in fish communities between the Sikunga FPA and the Sikunga BZ could be explained by their high degree of proximity which might have permitted inward and outward migration of similar species between the two sites. However, the disparity in fish community structure between Sikunga FPA and the Lisikili LFA was driven more by species of economic value (i.e. *S. macrocephalus*, *C. rendalli* and *O. macrochir*). These species were consistently more abundant at Sikunga FPA than at the Lisikili FA, indicating the possible impact of selective fishing for commercially important species at the Lisikili FA. Jul-Larsen et al. (2003) reported that selective fishing can change the structure of fish communities. Selective fishing for commercially important species has negatively affected the floodplain fishery of the Zambezi/Chobe Floodplain Fishery (Tweddle et al., 2015) and failed the cichlid fisheries of Lake Malombe in Malawi and the Kariba Dam in Zambia/Zimbabwe (Tweddle, Makwinja, & Sodzapanja, 1995). Thus, selective fishing for large growing cichlids of economic importance might have altered the fish community structure at Lisikili FA compared with the Sikunga FPA and Sikunga BZ respectively.

Fish densities and maximum attainable length values of the most abundant species were often higher at Sikunga FPA than at the Sikunga BZ and the Lisikili FA. Overall catch rates at Sikunga FPA were twice as high as those of the Sikunga BZ, and five times higher than those at the Lisikili FA. Over nine (9) years protection from any extractive activity at Sikunga FPA is likely to have promoted high fish densities and enhanced growth rates at this site. Therefore, we conclude that the conditions provided by the protected areas in the Sikunga FPA, such as lower human population density and some management rules (e.g. a ban on commercial fishing) may act synergistically to reduce the levels of fishing pressure and increase fishing productivity (catch per unit effort) for local fishers. Most of the species that responded positively to protection (i.e. *O. andersonii*, *C. rendalli* and *S. angusticeps*) are species of economic value. These three species typically have low natural mortality, late maturity, relatively long lifespan, slow to medium growth rates and large maximum size (Skelton, 2001), and have recently been under fishing pressure from the local fishermen (Hay et al., 2020; Simasiku, 2019; Tweddle et al., 2015). Our positive results on the effect of FPAs on the Zambezi River are in support with other studies within and beyond the study area. For instance, FPAs designed to protect nesting black bass (*Micropterus* spp.) from angling during the brood guarding stage in Lake Erie, New York, United States has proved to increase anglers' CPUE (Sztramko, 1985) and increase population-level reproductive success (Suski et al., 2002). In Hawaii, the duration of protection in sanctuaries had a significant effect on mean fish length, abundance and fish maturity (Sackett et al., 2014). Similarly, the establishment of FPAs played a significant role in increasing the diversity, biomass and mean sizes of commercially exploited fish species in Ngao River, Thailand (Koning & McIntyre, 2021) and the rehabilitation of exploited lake trout (*Salvelinus namaycush*) populations in both Lake

Huron and Lake Superior (Reid et al., 2001). A no-fishing reserve in Lake Kariba, Zimbabwe, increased both the number and size distribution of several freshwater fish families (Sanyanga et al., 1995). Positive findings include those of (Baird & Flaherty, 2005) who observed that villagers in the Mekong River, Thailand, reported increased fish abundance after protection zones were established. Cucherousset et al. (2007) found that eels were larger and more abundant in protected portions of a French wetland than in fished areas. Similarly, Sanyanga et al. (1995) reported that the mean body size of commercial species was larger in protected than in fished areas of Lake Kariba, Zimbabwe.

A particularly interesting observation pertains to a lack of variation in catch rates of *S. macrocephalus* and mean sizes of *C. gariepinus* between the protected and nonprotected areas, signifying that fish protected areas can be species-specific, and not all species in a water body will positively respond to protection. The large home range migratory behaviour of *C. gariepinus* could have compromised for positive results being found for the protected areas (Kadye & Booth, 2013; Skelton, 2001). This implies that species that exhibit a large home range may not be conserved efficiently in small reserves such as the Sikunga FPAs. Palumbi (2004) also reported that the effectiveness of protected areas for fish protection depends on fish movement and the size of the protected areas. Sedentary animals tend to be better protected than those that cross protected area boundaries. Maitland and Lyle (1992) found that Great Britain's National Nature Reserves fortuitously included populations of most native fish species despite not having been designed for this purpose; however, many of the species most in need of protection lacked adequate protected area coverage. Similarly, Impson et al. (2002) analysed national parks and nature reserves of South Africa's Cape Floral Kingdom and concluded that, although this set of protected areas contained populations of most indigenous fish species, actual protection was seriously impaired because the species' ranges extended largely outside the areas. Equally, the success of FPAs failed to protect highly mobile freshwater species, such as Lake Trout (Reid et al., 2001). Thus, the medium to long-distance longitudinal movements between habitats of many fish species make them particularly challenging taxa to conserve through place-based strategies (Fausch et al., 2002; Schlosser & Angermeier, 1995).

Another possible explanation to substantiate for the lack of variation in catch rates of *C. gariepinus* between the protected and nonprotected areas could be that *C. gariepinus* can flourish in any habitat (Kadye & Booth, 2013) and might have occupied the vacant predatory niche left by other commercially cropped species such as *S. angusticeps* in the nonprotected areas, resulting in an even distribution between the fish protected area and nonprotected areas. Similarly, in the Kariba system of Zimbabwe, *Synodontis zambezensis* showed a tendency to expand rapidly to occupy the habitats left vacant by other commercially cropped cichlid species (Sanyanga et al., 1995). However, the only potential explanation for a lack of variation in catch rates of *S. macrocephalus* between the protected and nonprotected areas is that this species is lightly fished as to be little influenced by the effective local protection provided at Sikunga FPA. According to Hay et al. (2020), *S. macrocephalus* accounts <4% by number and weight to the total catch of the commercial gillnet fishery of the Zambezi/Chobe Rivers. Thus, the frequencies of this species were very low in counts such that the likelihood of detection of differences was small given the low statistical power available.

Further analyses on the catch rates of the dominant species by mesh sizes showed that the catch rates for the larger mesh sizes (≥3.5 inch) were exceptionally lower at the Lisikili FA compared with the Sikunga FPA and the Sikunga BZ, suggesting the impact of high fishing pressure at the Lisikili FA. It has been postulated that, with cumulative fishing pressure, fish populations go through a series of changes in abundance and size because multi-species fisheries initially target the largest or most valuable species in a fish community (Allan et al., 2005; Tweddle, Turner, & Seisay, 1995; Welcomme, 2008). As larger individuals of species are removed, there is a decline in both fish densities and average sizes (Welcomme, 1999). Fishermen adapt to the decrease in average fish size by reducing their mesh sizes and targeting smaller-sized specimens of the larger growing species (Jul-Larsen et al., 2003; Karenge & Kolding, 1995; Welcomme, 1999).

The success and contribution of fish protected areas to fisheries management in the overall perspective of sustainability as well as to the conservation of biodiversity cannot neglect the socio-cultural and socioeconomic contexts in the target area. The current study reveals the potential of the Sikunga Buffer Zone through enhanced catch rates compared with other fished areas such as the Lisikili FA. This is mainly because the Sikunga BZ is partially subjected to community-based management systems that are tailored to safeguard the Sikunga FPA. Outsiders who wish to fish in the Sikunga BZ must express their interest and seek approval from the headman (Purvis, 2002). As a result, the Sikunga BZ provides an extra layer of protection through sustainability of human activities compared with the Lisikili FA. Such innovation has allowed for accumulation of more fish resources in Sikunga BZ that can support livelihoods for communities that have accepted and embraced the establishment of a Fish Protected Area in their vicinity. Despite much enthusiasm about the establishment of such fish protected areas in the region, this initiative has not been piloted intensively. This calls for an urgent need to establish more reserves along the entire river course of the Zambezi River as a step towards sustainable fisheries. The design of a specific FPA must accommodate the target fish species or community. Species that exhibit a large home range (i.e. *C. gariepinus*) may benefit from a network of more reserves in the study area. Networks of reserves can have collective benefits that exceed expectations for individual protected areas, and such emergent effects have been enshrined in design principles for maximising net benefits from marine reserve networks (Kramer & Chapman, 2004; Mas, 2005). However, this should be carefully considered because it might redirect excessive fishing pressure to unprotected areas and cause more management challenges. One approach to this concern can be addressed by allowing the local communities to fish buffer zones around the conserved areas.

Positive findings of this study underscore the fact that empowering communities to manage local resources can achieve

conservation and ecosystem service outcomes more effectively than top-down, centralised management. This concept is defined as co-management, and such management strategies focus on the recognition of active involvement of users in fisheries management if the regime is to be both effective and legitimate. The importance of involving riparian communities in a full capacity (i.e. collecting fisheries catch data, disseminating knowledge about the fisheries), to understand the temporal dimension and practical socioeconomic dynamics involved, is key for community-based interventions to ensure sustainable utilisation. Other studies have shown that local organisations in co-management have been effective in maintaining or even increasing fishing yields and fish abundance in the Brazilian Amazon (Almeida et al., 2009; Castello et al., 2013; Lopes et al., 2011; Silvano et al., 2014), as well as in other tropical rivers (Gupta et al., 2016) and marine ecosystems (Campbell et al., 2012).

Finally, it should be reemphasised that there is a disparity in results on the effectiveness of the Sikunga FPA between this study and that of Simasiku et al., 2017. This difference could be attributed to the sampling equipment employed by the two studies and age of the FPA. The current study used commercial gillnets similar to those fished by the local fishers in the region (see methods) while the former study used the multifilament experimental gillnets with variable mesh sizes (Peel et al. 2015). It was argued that, while experimental gillnets are good on assessing and reflecting the catch composition of an aquatic ecosystem, this gear cannot adequately reflect the catches of the gillnet fishery (Peel et al. 2015). Secondly, the former study (Simasiku et al., 2017) was conducted in 2016, when the Sikunga FPA was just 4 years from its inception in 2012. It is reported that the age of a protected zone is important for achieving long-term conservation goals. Older effective reserves show better results than younger reserves, with densities of fish increasing by approximately 5% per year in protected areas compared with unprotected areas (Simasiku et al., 2017). In view of the above, Simasiku et al. (2017) might have underestimated the effectiveness of the Sikunga FPA towards fisheries conservation in the Zambezi River, suggesting that future research efforts should account for sampling artefacts and the age of protected areas.

5 | CONCLUSIONS

Smaller fish reserves such as the Sikunga FPA has achieved its potential to enhance depleted fish stocks of economic value (i.e. *O. andersonii*, *C. rendalli* and *S. angusticeps*) in the Zambezi River. The model arising from this study is that the recovery of the commercially exploited fish species in Sikunga FPA will enable them to live longer, grow bigger and multiply in numbers over time. This can benefit local fisheries through fish protected area buffer zones, where target species are also expected to increase in abundance, biomass and subsequently harvested by fishers who can gain a sustainable livelihood as a result of FPAs. The study emphasises on the pivotal need, for government and nongovernmental institutions to involve the local community in decision-making and empower them to take full responsibility of safeguarding their fishery resources through the concept of co-management. There is a clear need for science to inform and improve management in a region where limited information is available on the effectiveness of fish protected areas (FPAs) as a management tool. Management approaches that foster awareness and engage with communities surrounding the FPAs are recommended for successful conservation of the fish resources. More importantly, studies that include monitoring prior to reserve establishment and report abundances of all species censused are apt to yield the greatest information about why some species respond to protection more strongly than others.

AUTHOR CONTRIBUTIONS

Dr. Evans Simasiku is the main author and initiator of the project and participated in data collection, data entry, data cleaning and writing up of the paper, with contributions to all sections of the paper. Dr. Clinton Hay supervised the initiation and facilitation of the project. He also conducted most multivariate statistical analyses of the project.

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CONFLICT OF INTEREST

We declare a close relationship between one of the editors for this respective journal; Mr Denis Tweddle and his review for this article might bias the content and findings of this article. For the sake of fairness and the outmost interest of this article and the target journal, it will be appreciated if other independent reviewers should be considered.

DATA AVAILABILITY STATEMENT

Derived data supporting the findings of this study are available from the corresponding author upon reasonable request. However, such data should be archived and only be shared at the discretion of the corresponding author.

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