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Marine Biodiversity of Coral Reef Fishes in Pieh Marine Recreational Park After Bleaching and Acanthaster Outbreaks

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Abstract

Pieh MRP encountered non-anthropogenic phenomena, precisely massive coral bleaching in 2016, 2017, and massive predators from *Acanthaster planci* outbreaks in 2018. This study aimed to understand the pattern of coral reef diversity in the core zone and utilization in the MRP area and compare it to non-MRPS locations that accept the same non-anthropogenic pressure conditions. Coral fish sampling using a UVC is categorized into three zones: the core zone, the utilization zone, and outside the MRP area. 8 Families of coral reef fishes were counted based on categories of level function in ecologies and economy. Taxonomic distinctiveness estimates were calculated mathematically for each sample, including species richness and taxonomic diversity were compared among zonation area. Pearson's Coefficient Correlation Matrix was used to measure the correlation relationship between zonation areas. There are 91 species of fish and 3002 individuals found. The richest family in the MRP Core Zone and MRP Utility Zone was Acanthuridae with 20 species and non-MRP has a lower species richness and abundance of fish communities. The dominant species in Pieh MRP was *Ctenochaetus striatus with* average abundant per site (21.3 ± 7.62, $n = 3$). Acanthuridae represents 55.98% of the total biomass in MRP-Core Zone, 63.13% in MRP-Utility Zone, and 41.55% in Non-MRP Area. This study showed the number of species and populations from corallivores fishes have decreased but has been an increase in herbivorous and carnivore diversity. The diversity indices (H') and ENS also shows no differ significantly between zonation.

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1. Introduction

Awareness of the threat of coral reef ecosystems has increased globally (Gardner *et al.*, 2003). Continuous pressure on coral reefs ecosystems caused by anthropogenic and non-anthropogenic factors leads to a decline in coral cover globally and reduce ecological conditions in the coastal ecosystem (Harvey *et al.*, 2018; Heery *et al.*, 2018; Hoegh-Guldberg *et al.*, 2017). The decline of the coral reefs ecosystem is a major concern for ecologists. The pressure from non-anthropogenic and anthropogenic factors including global warming with related coral bleaching (Jones *et al.*, 2004), increasing human populations and activities in coastal areas (Trenouth and Campbell, 2013), overfishing (McClanahan and Jadot, 2017), and destructive fishing activities (McManus and Polsenberg, 2004) can cause loss of flora and fauna species that depend on this ecosystem. The ecological role of various species of flora and fauna is very crucial for the functioning of coral reef ecosystems. The healthy functioning of coral reef ecosystem services is highly beneficial to human society (Thibaut and Connolly, 2013).

The usefulness and high ecological function of coral reef ecosystems increase awareness about the necessity to manage and reduce pressure on coral reef ecosystems, including by decreasing anthropogenic stressor caused by increasing population and human activities (Bruno and Valdivia, 2016; Trenouth and Campbell, 2013) with Marine Protection Area management. The formation of Marine Protects Area (MPA) and Marine Recreational Park (MRP) is very appropriate in succeeding in the controlled pressure on coral reef ecosystems. As in Pieh Island and the surrounding sea located in West Sumatra that used as the location of MRPs. Formation Pieh as MRPs is formed by legislation in the Minister of Forestry Decree Number 070 / Kpts-II / 2000, dated March 18, 2000, as Nature Marine Tourism Park (NMTP) (Abrar *et al.*, 2014), In its development, the implementation of Pieh Island as NMTP management submitted to the Ministry of Maritime Affairs and Fisheries under the Decree of the Ministry of Maritime Affairs and Fisheries Number Kep. 70 / Men / 2009 and became the Marine Recreational Park (MRP) of Pieh Island and the Surrounding Sea. Currently, Pieh MRP is managed by the Directorate General of Treasury Affairs Pekanbaru with field technical implementation work units stationed in Padang City, West Sumatra Province (Abrar *et al.*, 2014). The conservation area program in MRPs in Pieh has implemented specific zoning areas and divided into two main zones namely core zone (no-take zone) and the utilization zone. The core zone in Pieh MRP is part of the protected area of small islands, which is intended for the protection of habitats and populations, the resources and their use is limited to research, while the utilization zone in Pieh MRP is primarily utilized for the benefit of nature tourism and other environmental conditions/services. This division of zones provides an appropriate level of protection for a variety of representative species and habitats (Hammerton, 2017; Himes, 2007) and provides opportunities for the government and the community to manage their marine areas.

In the past few decades, global efforts have been made to develop marine protected areas as both MPAs and MRPs. This program is not only arranged in Indonesia, but also in global effort to preserve increasingly threatened marine ecosystems to protect genetic diversity, species ecosystems extensively (Abelson *et al.* 2016; Hammerton, 2017). Although there are several regional and global successes in establishing MPAs and MRPs, the fact is that coral reefs around the world is experiencing a severe decline (Attamimi, 2019; Bellwood *et al.*, 2004; Paddack *et al.*, 2009). The weak management and supervision have resulted in the objectives to manage coral reef ecosystems not functioning correctly. Some problems are excessive utilization and harvesting (Jackson *et al.*, 2001; Pandolfi *et al.*, 2003), pollution (McCulloch *et al.*, 2003), disease (Harvell *et al.*, 2002), and non-anthropogenic factors such as climate change (Hughes *et al.*, 2003; Wilkinson, 2008). At present, the remaining coral populations are increasingly affected by increasingly prevalent coral diseases and climate change that triggered coral bleaching (Gardner *et al.*, 2003; Putra *et al.*, 2019) and marine predator outbreaks (*A. planci*) (Kayal *et al.*, 2012; Leray *et al.*, 2012; Mellin *et al.*, 2016). This degradation of coral reef cover health impacts the existence of coral reefs in the long term and will indirectly affect several related biotas that depend on coral reefs, especially reef fish. A decrease in the structural complexity of reef habitats is often associated with changes in fish communities and influence on the role of fish in coral reefs (Woodhead *et al.,* 2019). Reef fish also have an essential role in the recovery *of* coral reefs and the presence of certain species can accelerate the recovery of coral reefs (Cheal *et al.*, 2008)

In its development, the Pieh MRP encountered non-anthropogenic phenomena, in particular massive coral bleaching in 2016, 2017, and massive predators from crown-of-thorns starfish (*A. planci*) outbreaks in 2018. The pressure caused almost the majority of Pieh MRP areas to change in coral reef ecosystems. The emergence of Rubble and Dead Coral Algae (DCA) after disturbance of non-anthropogenic pressure in the MRP in Pieh increases deterioration of coral reef ecosystems. The *A. planci* outbreaks in Pieh MRP in 2018 contributed to the decrease of health of the coral reef cover and reduced coral reefs recovery. The Directorate General of Treasury, the Ministry of Maritime Affairs

and Fisheries of Indonesia with several NGOs have tried to maintain and control the location by removing nearly 800 individuals of *A. planci* in the coral reef to maintain and succeed in the MPA management program. The increasing of non-anthropogenic stressor as massive coral bleaching and the appearance of marine predators (*A. planci*) are the main concerns in evaluating the health of coral reefs globally because these cause a shift from living corals to algal dominance (McManus and Polsenberg, 2004). The dominance of algal growth causes disruption on hard coral, where macro and fleshy algae covers live coral tissue and cause disturbance to coral growth (Chabanet *et al.*, 2016). The response of coral reef fish to coral ecosystems after the impact of these pressures can be an essential factor influencing the success of this protected area (De Freitas *et al.*, 2013). It is necessary to understand the pattern of coral reef diversity and to recognize the influence on reef fish diversity in order to support coral reef recovery from the magnitude of non-anthropogenic pressure and to assist in making policies and managing MRPs sustainability. It is necessary to understand the influence and role of the core zones and utilization zones for changing the diversity of reef fish after the bleaching and Acanthaster outbreak in Pieh MRP. Understanding the patterns of reef fish diversity can help support coral reef recovery from the magnitude of non-anthropogenic pressure and can be used to make policies and management for sustainable MRPs. This study provided the biodiversity of reef fish after the phenomenon of coral bleaching and *A. planci* outbreaks in the Pieh MRP, based on different zonation: Core Zone (No-take zone), Utility Zone (Recreational Zone) and we compare with Non-MPA (Fishing Zone) areas. The phenomenon of coral bleaching and the *A. planci* outbreaks impact on coral conditions and lead to the dominance of algal growth in the Pieh areas. This study aims to examine the condition of coral ecosystems by evaluating coral reef fish composition after the appearance of bleaching phenomena and outbreaks of *A. planci* to found a possible recovery of the coral ecosystem at MRPs Pieh location.

2. Materials and Methods

2.1 Study Sites

The study was conducted in March 2019 in Marine Recreational Park (MRP) in Pieh and the surrounding sea, consisting of 5 small uninhabited islands (Pieh Island, Air Island, Bando Island, Pandan Island, and Toran Island) (Figure 1). Marine Recreational Park (MRP) Pieh with 39,900 ha covers a strip along western coastline, western of Padang, Indonesia. The coral reef ecosystem spread out, extending north-south along the shoreline of West Sumatra, and the main benthic features were *fringing reef* with *Pocilloporiid,* and *Acroporiid* (Abrar *et al.*, 2014). Management and utilization of (MRP) Pieh area focused on environmentally friendly marine tourism by taking into account the aspects of shared ownership, management, and responsibility, to get harmonized with its use as marine tourism. These areas were also a fishing location (fishing ground) for local fishermen, as evidenced by the number of Fish Aggregating Devices (FADs) in the sustainable utilization zone. Almost all islands become dive sites with clear waters, and beautiful coral reefs as the main attraction. Conservation directed towards efforts to protect the diversity of marine biota in this area, especially rare and protected biota, several marine mammals such as dolphins and whales, and near-threatened fish species including sharks and Napoleon (Abrar *et al.*, 2014). The sampling design of the research location was categorized into three group zonation areas: (1) core zone area (no-take zone) in MRP, (2) utilization zone area (recreational zone) in MRP, and (3) outside MRP area (fishing zone). A total of thirteen sampling sites (10 sites within the MRP area, divided by 2 categorized zonation that was five sites in core zone area in MRP and five sites in utilization zone area in MRP), other three sites outside MRP area selected to compare coral reef fishes diversity, composition, and biomass. The geographical positioning system (GPS) locations of the sites was recorded (Table 1)

2.2 Data Collection

The coral reef fishes were initially surveyed using Underwater Visual Censuses (UVC) (Floeter *et al.*, 2004). Underwater Visual Census method (UVC) for reef fishes was done along 70 m of transect line straight and follows the contour of the shore and laid out parallel to the shore (Bouchon-Navaro, 1981; Giyanto *et al.*, 2014). The basic unit of data collection recording fish encountered within 70 meters and 2.5 meters of observation on either side to record fish species, estimated length, abundance, and several reef characteristics of interest (e.g., live coral cover, algal cover, depth, visibility, etc.) (Zenone *et al.*, 2017). This observation generated an observation area of 350 m^2 (Putra *et al.*, 2018). All observations of UVC made while scuba diving at depths of 4–10 m in the coral reef ecosystem. Constant speed was maintained as far as possible to prevent the recounting of fish (Buxton and Smale, 1989). There were 8 families of coral reef fishes that were counted based on its ecologies and economical function. The first category was a coral feeder or corralivore (Chaetodontidae), the second category in herbivore fishes group (Acanthuridae, Scaridae, and Siganidae), the third category is carnivore fishes group (Haemulidae, Lethrinidae, Lutjanidae and Serranidae). The corralivore fish (Chaetodontidae) are indicator fishes that are strongly associated with coral reef health and used to
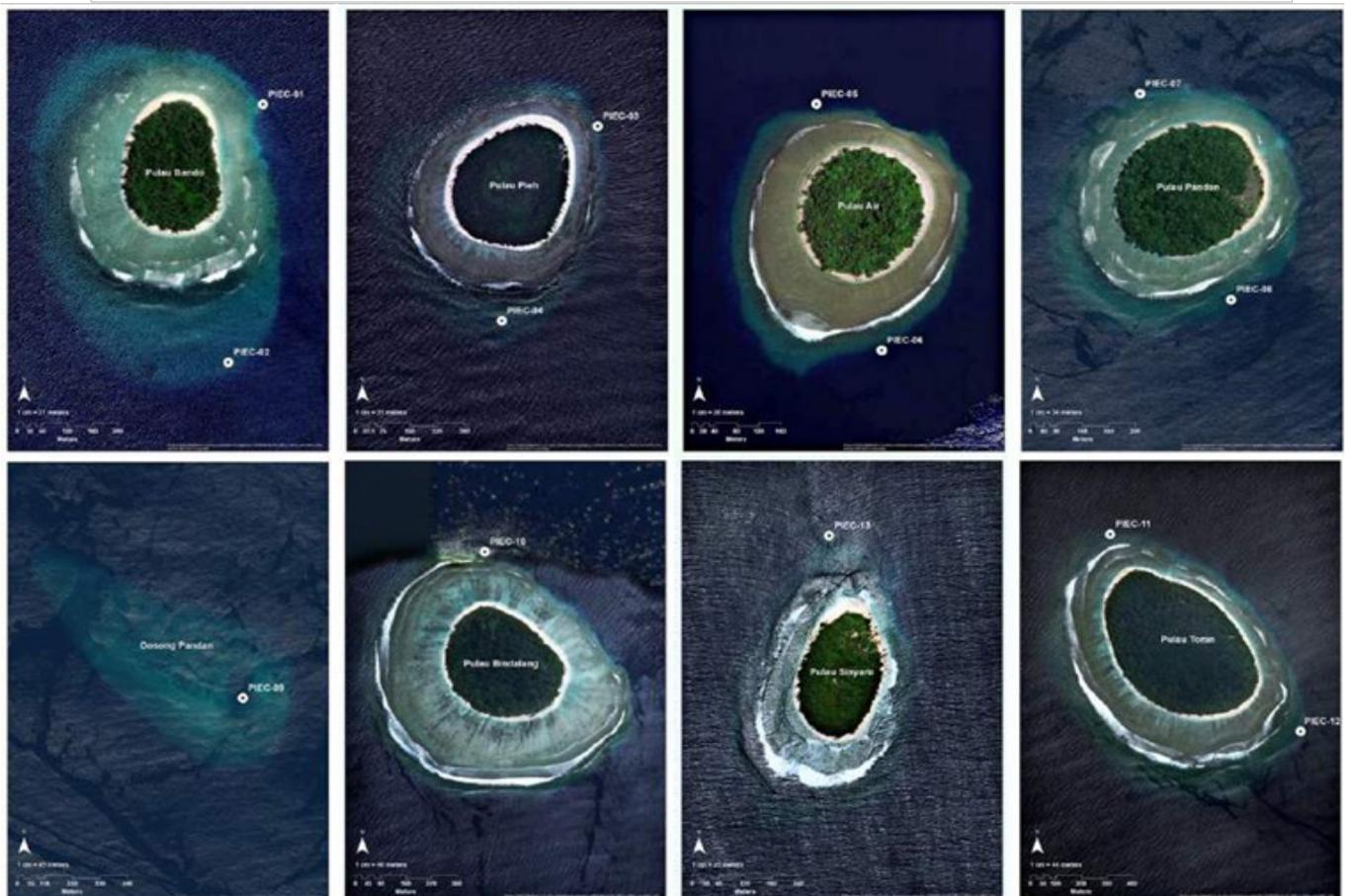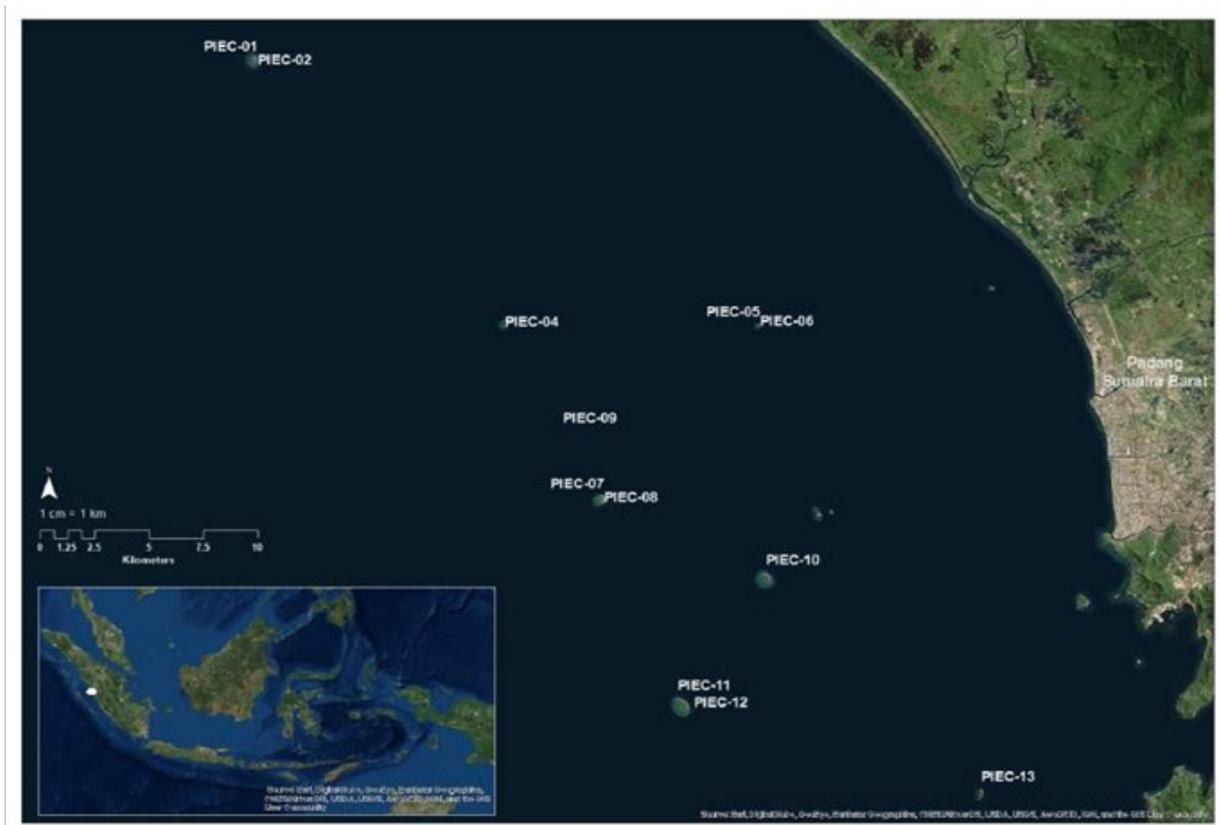
Figure 1. Map of study sites: MRP Area (Air Island, Bando Island, Pandan Island, Pieh Island, and Toran Island) and the Non-MRP Area (Bindalang Island, Sinyaru Island, and Gosong Pandan Reef) Padang City, West Sumatra

determine the coral reef health. The majority of herbivore and carnivore fish in these family group are economical target fish were fish caught by fishermen because of their high economic, commercial value and usually sold on the market for food consumption (Corrales *et al.*, 2015; Madduppa *et al.*, 2014).

2.3 Data Analysis

The number of total individual in each species, estimates of reef fish total length recorded, and bias measurement of estimation total length for each individual reef fishes should not exceed 5 cm (Buxton and Smale, 1989). The estimated length of fish data was converted to fish biomass using length-weight relationships for fish species using the following equation W = a x L^b, where W is weight (g), L is total length (cm), and parameters a and b are constants for the allometric growth equation (Luiz *et al.*, 2015). These coefficients come from data published in FishBase (fishbase.org) when coefficient values not found for the species; this study used coefficients defined for a species with morphological characteristics similar to the observed species. The total biomass for each station corresponds to the total weight of all fish per unit area (kg/m^2). The identification of reef fish is based primarily on the identification of reef fish by (Allen *et al.*, 2003). Most values and attributes were obtained from FishBase (Froese and Pauly, 2017). Reef fish data collected were analyzed for ranking diversity based on family per zonation (MRP-CZ, MRP-UZ, and Non-MRP) and for each species of fish will be analyzed based on rank species per zonation.

In the particular case of zonation area of MRP, several taxonomic distinctiveness estimates were calculated mathematically for each sample, including species richness (the number of fish taxa per area) and taxonomic diversity by the Shannon– Wiener diversity index (*H'*), Alpha fisher Index (*AF*), ENS (Effective Number of Species) and Simpson Dominance Index (*S*) were compared among zonation area. To measure how strong the correlation of relationships between zonation area in each site using Pearson's coefficient correlation matrix (CCM). This study compared the changing coral reef composition (species richness, abundance and biomass) before (2014) and after (2019) bleaching phenomena and *A. planci* outbreaks.

3. Results and Discussion

3.1 Result

A total of 91 fish species and 3002 individuals are observed from the eight-coral reef fish families (Chaetodontidae, Acanthuridae, Scaridae, Siganidae, Haemulidae, Lethrinidae, Lutjanidae, and Serranidae) in 13 sites location. The most diverse family in core zone Area MRP (MRP-CZ) is Acanthuridae with 20 species, followed by Chaetodontidae with 18 species, Scaridae with 13 species, Serranidae with eight species, Lutjanidae with six species, Siganidae with three species, Haemulidae with three species, and Lethrinidae with three species (Table 2). Fish communities in core zone area MRP are dominated by Acanthuridae, which represented 60.73% of the total number of individuals coral

Table 1. Location coordinates of sites and zonation in Marine Recreational Park Pieh

No	Location	Sites.id	Zonation	Depth (meters)	Longitude	Latitude
1	Bando Island	PIEC.01	MRP-CZ	7	99° 59' 52.738'' E	0° 45' 53.712'' S
2	Pandan Island	PIEC.02	MRP-UZ	9	99° 59' 57.012'' E	0° 45' 41.508'' S
3	Pieh Island	PIEC.03	MRP-UZ	7	100° 06' 9.360'' E	0° 52' 28.128'' S
4	Pieh Island	PIEC.04	MRP-CZ	5	100° 06' 0.360'' E	0° 52' 38.712'' S
5	Air island	PIEC.05	MRP-UZ	5	100° 12' 25.56'' E	0° 52' 28.092'' S
6	Air Island	PIEC.06	MRP-CZ	6	100° 12' 19.80'' E	0° 52' 37.632'' S
7	Pandan Island	PIEC.07	MRP-UZ	10	100° 08' 34.08'' E	0° 56' 55.788'' S
8	Pandan Island	PIEC.08	MRP-CZ	5	100° 08' 27.96'' E	0° 57' 09.468'' S
9	Gosong Pandan Island	PIEC.09	Non-MRP	8	100° 07' 26.76'' E	0° 55' 6.8988'' S
10	Bindalang Island	PIEC.10	Non-MRP	5	100° 12' 28.44'' E	0° 58' 46.499'' S
11	Toran Island	PIEC.11	MRP-CZ	6	100° 10' 16.68'' E	1° 02' 0.8880'' S
12	Toran Island	PIEC.12	MRP-UZ	7	100° 10' 37.56'' E	1° 02' 15.360'' S
13	Sinyaru Island	PIEC.13	Non-MRP	4	100° 17' 48.84'' E	1° 04' 21.900'' S

Note: MRP CZ (Marine Recreational Park – Core Zone); MRP-UZ (Marine Recreational Park–Utility Zone); Non-MRP (non-Marine Recreational Park).

reef fishes with *Ctenochaetus striatus* (48.4 ± 10.97, n = 5) and *Acanthurus lineatus* (27.4± 9.48, n=5) being the two most average abundant species per site (Table 2). The most diverse family of coral reef fishes in utilization zone area MRP (MRP-UZ) almost similar condition in MRP-CZ, where Acanthuridae is richest family with 17 species, followed by Chaetodontidae with 15 species, Serranidae with 11 species, Scaridae with ten species, Siganidae with five species, Lutjanidae with five species, Lethrinidae with three species, and Haemulidae with two species (Figure 2). Fish communities in MRP-UZ are dominated by Acanthuridae, which represent 61.13% of the total number of individuals coral reef fishes with *C. striatus* (31.8 ± 5.76, n = 5) and *A. lineatus* (27.0 ± 7.11, n=5) being the two most average abundant per site (Table 2).

Outside Area MRP (Non-MRP) has a lower species richness and fish community abundance, where Acanthuridae is richest family with 17 species, followed by Chaetodontidae with 16 species, Scaridae with nine species, Serranidae with six species, Siganidae with five species, Lutjanidae with 5 species, Haemulidae with 1 species, and Lethrinidae with one species (Figure 2). Fish communities in non-MRP were dominated by Acanthuridae, which represented 47.74% of the total number of individuals coral reef fishes with *C. striatus* (21.3 ± 7.62, n = 3) being the most average abundant per site and followed by *Acanthurus tristis* (14.3 ± 8.69, n=3), *Forcipiger flavissimus* (10.3 ± 3.53, n=3), *Siganus virgatus* (10.3 ± 2.96, n=3), *A. lineatus* (10.0 ± 3.46, n=3), *Zebrasoma scopas* (9.3 ± 4.81, n=3), *Chlorurus capistratoides* (8.3 ± 4.48, n=3), *Cephalopolis argus* (8.3 ± 4.48, n=3), *Heniochus pleurotaenia* (7.7 ± 1.76, n=3), and *Naso hexacanthus* (7.7 ± 5.78, n=3) (Table 2). Through the Kruskal-Wallis test to species richness, mean abundance and mean biomass of composition reef fishes in three region area (core zone, utility zone, and non-MRP), the significant difference was showed by species richness (Kruskal-Wallis chi-squared = 12.558, df = 2, p-value = 0.001876) (Figure 2). Pairwise Wilcoxon test for the species diversity of reef fish shows a significant difference between MRP-CZ with Non-MRP (p-value = 0.0220) and between MRP-UZ with Non-MRP (p-value =0.0015). The mean abundance and biomass of reef fish shows no differences between 3 locations with (Kruskal-Wallis chi-squared = 2.66, df = 2, p-value = 0.2645) and (Kruskal-Wallis chi-squared = 1.0339, df = 2, p-value = 0.5963), respectively (Figure 2).

Table 2. List of taxa, number of individuals ((± SE individual/350 m2), in Marine Recreational Park (Core Zone and Utilization Zone) and Non-Marine Recreational Park in Pieh after non-anthropogenic pressure condition (2019)

Species	Common Names	Trophic Status	CZ-MRP n=5	UZ-MRP n=5	(N-MRP) n=3
Chaetodontidae					
Chaetodon citrinellus (Cuvier, 1831)	Speckled butterflyfish	Corallivores	2.2 ± 1.28	2.6 ± 1.08	3.3 ± 3.33
Chaetodon collare (Bloch, 1787)	Redtail butterflyfish	Corallivores	0.2 ± 0.20	1.6 ± 0.98	3.3 ± 2.03
Chaetodon falcula (Bloch, 1795)	Blackwedged butterflyfish	Corallivores	1.6 ± 0.75	1.0 ± 0.63	0.3 ± 0.33
Chaetodon kleinii (Bloch, 1790)	Sunburst butterflyfish	Corallivores	2.4 ± 0.93	-	1.0 ± 0.58
Chaetodon lunula (Lacepède, 1802)	Raccoon butterflyfish	Corallivores	0.4 ± 0.40	0.2 ± 0.20	0.7 ± 0.67
Chaetodon meyeri (Bloch & Schneider, 1801)	Scrawled butterflyfish	Corallivores	0.6 ± 0.24	0.6 ± 0.60	1.0 ± 1.00
Chaetodon ornatissimus (Cuvier, 1831)	Ornate butterflyfish	Corallivores	1.0 ± 0.63	0.8 ± 0.49	0.3 ± 0.33
Chaetodon oxycephalus (Bleeker, 1853)	Spot-nape butterflyfish	Corallivores	0.6 ± 0.40	-	-
Chaetodon rafflesi (Bennett, 1830)	Latticed butterflyfish	Corallivores	1.6 ± 0.93	1.6 ± 0.75	2.0 ± 1.15
Chaetodon triangulum (Cuvier, 1831)	Triangle butterflyfish	Corallivores	1.4 ± 0.87	1.0 ± 0.55	0.3 ± 0.33
Chaetodon trifascialis (Quoy & Gaimard, 1825)	Chevron butterflyfish	Corallivores	0.2 ± 0.20	0.4 ± 0.40	0.3 ± 0.33
Chaetodon trifasciatus (Park, 1797)	Melon butterflyfish	Corallivores	4.0 ± 1.41	5.8 ± 1.28	4.3 ± 1.45
Chaetodon vagabundus (Linnaeus, 1758)	Vagabond butterflyfish	Corallivores	4.0 ± 1.30	4.8 ± 1.71	5.7 ± 3.18
Forcipiger flavissimus (Jordan & McGregor, 1898)	Longnose butterfly fish	Corallivores	5.2 ± 2.22	5.6 ± 1.08	10.3 ± 3.53
Forcipiger longirostris (Broussonet, 1782)	Longnose butterflyfish	Corallivores	-	0.4 ± 0.40	-
Hemitaurichthys zoster (Bennett, 1831)	Brown-and-white butterflyfish	Corallivores	0.4 ± 0.24	-	-
Heniochus acuminatus (Linnaeus, 1758)	Pennant coralfish	Corallivores	0.2 ± 0.20	-	0.7 ± 0.67

Heniochus pleurotaenia (Ahl, 1923)	Phantom bannerfish	Corallivores	7.0 ± 1.92	7.6 ± 2.54	7.7 ± 1.76
Heniochus singularius (Smith & Radcliffe, 1911)	Singular bannerfish	Corallivores	4.8 ± 0.92	4.2 ± 0.86	2.0 ± 0.58
Acanthuridae					
Acanthurus nubilus (Fowler & Bean, 1929)	Bluelined surgeon	Herbivores	5.2 ± 3.07	0.6 ± 0.60	-
Acanthurus grammoptilus (Richardson, 1843)	Finelined surgeonfish	Herbivores	-	3.2 ± 1.62	-
Acanthurus leucocheilus (Herre, 1927)	Palelipped surgeonfish	Herbivores	0.2 ± 0.20	-	0.7 ± 0.67
Acanthurus leucosternon (Bennett, 1833)	Powderblue surgeonfish	Herbivores	10.4 ± 9.67	0.8 ± 0.58	3.0 ± 3.00
Acanthurus lineatus (Linnaeus, 1758)	Lined surgeonfish	Herbivores	27.4 ± 9.48	27 ± 7.11	10.0 ± 3.46
Acanthurus maculiceps (Ahl, 1923)	White-freckled surgeonfish	Herbivores	1.4 ± 1.17	3.4 ± 1.21	4.0 ± 4.00
Acanthurus mata (Cuvier, 1829)	Elongate surgeonfish	Herbivores	0.2 ± 0.20	-	-
Acanthurus nigricans (Linnaeus, 1758)	Whitecheek surgeonfish	Herbivores	0.6 ± 0.40	1.0 ± 0.77	7.0 ± 5.57
Acanthurus thompsoni (Fowler, 1923)	Thompson's surgeonfish	Herbivores	3.0 ± 2.07	10.6 ± 5.77	0.7 ± 0.67
Acanthurus triostegus (Linnaeus, 1758)	Convict surgeonfish	Herbivores	1.0 ± 1.00	-	-
Acanthurus tristis (Randall, 1993)	Indian Ocean mimic surgeonfish	Herbivores	5.0 ± 1.38	13.0 ± 3.49	14.3 ± 8.69
Acanthurus tennentii (Günther, 1861)	Doubleband surgeonfish	Herbivores		-	0.7 ± 0.67
Ctenochaetus binotatus (Randall, 1955)	Twospot surgeonfish	Herbivores	0.4 ± 0.40	14.2 ± 11.05	-
Ctenochaetus striatus (Quoy & Gaimard, 1825)	Striated surgeonfish	Herbivores	48.4 ± 10.97	31.8 ± 5.76	21.3 ± 7.62
Ctenochaetus truncatus (Randall & Clements, 2001)	Indian gold-ring bristle-tooth	Herbivores	5.2 ± 3.15	3.8 ± 1.56	3.3 ± 2.03
Naso brachycentron (Valenciennes, 1835)	Humpback unicornfish	Herbivores	2.6 ± 1.66	1.4 ± 1.40	0.3 ± 0.33
Naso hexacanthus (Bleeker, 1855)	Sleek unicornfish	Herbivores	15.6 ± 11.63	25.4 ± 19.18	7.7 ± 5.78
Naso elegans (Rüppell, 1829)	Elegant unicornfish	Herbivores	1.0 ± 1.00	1.2 ± 0.80	1.3 ± 1.33
Naso lituratus (Forster, 1801)	Orangespine unicornfish	Herbivores	0.2 ± 0.20	0.6 ± 0.40	0.3 ± 0.33
Naso thynnoides (Cuvier, 1829)	Oneknife unicornfish	Herbivores	7.0 ± 4.63	-	6.7 ± 3.76
Naso unicornis (Forsskål, 1775)	Bluespine unicornfish	Herbivores	0.4 ± 0.40	-	-
Naso vlamingii (Valenciennes, 1835)	Bignose unicornfish	Herbivores	-	0.2 ± 0.20	-
Zebrasoma scopas (Cuvier, 1829)	Twotone tang	Herbivores	6.2 ± 4.52	11.2 ± 4.52	9.3 ± 4.81
Zebrasoma veliferum (Bloch, 1795)	Sailfin tang	Herbivores	-	-	0.7 ± 0.67
Scaridae					
Cetoscarus bicolor (Rüppell, 1829)	Bicolour parrotfish	Herbivores	0.6 ± 0.60	-	-
Chlorurus bleekeri (de Beaufort, 1940)	Bleeker's parrotfish	Herbivores	0.8 ± 0.37	1.2 ± 1.20	2.7 ± 1.45
Chlorurus capistratoides (Bleeker, 1847)	Indian parrotfish	Herbivores	1.8 ± 0.66	4.0 ± 2.02	8.3 ± 4.48
Chlorurus sordidus (Forsskål, 1775)	Daisy parrotfish	Herbivores	0.6 ± 0.24	0.2 ± 0.20	-
Chlorurus troschelii (Bleeker, 1853)	Troschel's parrotfish	Herbivores	0.2 ± 0.20	-	-
Scarus dimidiatus (Bleeker, 1859)	Yellowbarred parrotfish	Herbivores	0.8 ± 0.80	1.0 ± 1.00	2.7 ± 1.33
Scarus ghobban (Forsskål, 1775)	Blue-barred parrotfish	Herbivores	1.2 ± 1.20	0.6 ± 0.40	0.7 ± 0.67
Scarus niger (Forsskål, 1775)	Dusky parrotfish	Herbivores	2.8 ± 1.02	4.6 ± 1.81	4.0 ± 2.08
Scarus oviceps (Valenciennes, 1840	Dark capped parrotfish	Herbivores	0.2 ± 0.20	-	-
Scarus prasiognathos (Valenciennes, 1840)	Singapore parrotfish	Herbivores	0.2 ± 0.20	-	0.3 ± 0.33
Scarus quoyi (Valenciennes, 1840	Quoy's parrotfish	Herbivores	-	0.6 ± 0.60	-
Scarus rubroviolaceus (Bleeker, 1847)	Ember parrotfish	Herbivores	4.8 ± 2.37	1.8 ± 0.66	5.0 ± 4.04
Scarus tricolor (Bleeker, 1847	Tricolour parrotfish	Herbivores	1 ± 0.55	2.6 ± 0.68	2.0 ±
Scarus viridifucatus (Smith, 1956)	Roundhead parrotfish	Herbivores	0.2 ± 0.20	0.2 ± 0.20	0.3 ± 0.33
Siganidae					

Siganus corallinus (Valenciennes, 1835)	Blue-spotted spinefoot	Herbivores	0.2 ± 0.20	0.8 ± 0.49	0.3 ± 0.33
Siganus guttatus (Bloch, 1787)	Orange-spotted spinefoot	Herbivores	0.8 ± 0.80	6.8 ± 5.23	0.3 ± 0.33
Siganus magnificus (Burgess, 1977)	Magnificent rabbitfish	Herbivores	-	0.8 ± 0.49	0.7 ± 0.67
Siganus punctatus (Schneider & Forster, 1801)	Goldspotted spinefoot	Herbivores	-	0.8 ± 0.49	-
Siganus vermiculatus (Valenciennes, 1835)	Vermiculated spinefoot	Herbivores	-	-	0.3 ± 0.33
Siganus virgatus (Valenciennes, 1835)	Barhead spinefoot	Herbivores	1 ± 0.63	0.6 ± 0.60	10.3 ±
Haemulidae					
Diagramma melanacrum (Johnson & Randall, 2001	Blackfin slatey	Carnivores	0.6 ± 0.60	-	-
Plectorhinchus chaetodonoides (Lacepède, 1801)	Harlequin sweetlips	Carnivores	-	0.2 ± 0.20	-
Plectorhinchus gibbosus (Lacepède, 1802)	Harry hotlips	Carnivores	3.2 ± 3.20	-	-
Plectorhinchus vittatus (Linnaeus, 1758)	Indian Ocean oriental sweetlips	Carnivores	2.8 ± 1.83	1.6 ± 0.93	3.3 ± 3.33
Lethrinidae					
Lethrinus ornatus (Valenciennes, 1830	Ornate emperor	Carnivores	1.2 ± 0.80	2.6 ± 0.81	-
Monotaxis grandoculis (Forsskål, 1775)	Humpnose big-eye bream	Carnivores	1.4 ± 0.75	3.8 ± 1.56	2.3 ± 1.20
Monotaxis heterodaon (Bleeker, 1854)	Redfin emperor	Carnivores	2 ± 2.00	1.0 ± 0.77	-
Lutjanidae			-		
Lutjanus argentimaculatus (Forsskål, 1775)	Mangrove red snapper	Carnivores		0.4 ± 0.40	-
Lutjanus biguttatus (Valenciennes, 1830)	Two-spot banded snapper	Carnivores	0.6 ± 0.60	-	-
Lutjanus decussatus (Cuvier, 1828)	Checkered snapper	Carnivores	3.4 ± 1.03	4.4 ± 1.12	3.3 ± 0.33
Lutjanus fulviflamma (Forsskål, 1775)	Dory snapper	Carnivores	-	1.2 ± 1.20	3.3 ± 0.33
Lutjanus fulvus (Forster, 1801)	Blacktail snapper	Carnivores	1.2 ± 1.20	0.2 ± 0.20	2.0 ± 2.00
Lutjanus monostigma (Cuvier, 1828)	One-spot snapper	Carnivores	1.8 ± 1.36	-	-
Lutjanus quinquelineatus (Bloch, 1790)	Five-lined snapper	Carnivores	0.2 ± 0.20	-	-
Macolor macularis (Fowler, 1931)	Midnight snapper	Carnivores	-	-	1.3 ± 1.33
Macolor niger (Forsskål, 1775)	Black and white snapper	Carnivores	7.0 ± 4.16	8.0 ± 2.59	6.0 ± 3.79
Serranidae					
Aethaloperca rogaa (Forsskål, 1775)	Redmouth grouper	Carnivores	3.4 ± 0.81	1.2 ± 0.58	0.6 ± 0.67
Cephalopholis argus (Schneider, 1801	Peacock hind	Carnivores	15.2 ± 0.80	7.8 ± 2.15	8.3 ± 4.48
Cephalopholis cyanostigma (Valenciennes, 1828)	Bluespotted hind	Carnivores	-	0.4 ± 0.24	-
Cephalopholis miniata (Forsskål, 1775)	Coral hind	Carnivores	0.2 ± 0.20	0.2 ± 0.20	-
Cephalopholis polyspila (Randall & Sata-poomin, 2000)	polyspila hind	Carnivores	-	0.4 ± 0.40	1.7 ± 0.88
Cephalopholis spiloparaea (Valenciennes, 1828)	Strawberry hind	Carnivores	1.2 ± 0.80	0.4 ± 0.24	1.3 ± 1.33
Cephalopholis urodeta (Forster, 1801)	Darkfin hind	Carnivores	0.2 ± 0.20	0.6 ± 0.40	2.0 ± 2.00
Epinephelus coeruleopunctatus (Bloch, 1790)	Whitespotted grouper	Carnivores	-	0.4 ± 0.40	-
Epinephelus fasciatus (Forsskål, 1775)	Blacktip grouper	Carnivores	0.2 ± 0.20	-	-
Epinephelus merra (Bloch, 1793)	Honeycomb grouper	Carnivores	0.2 ± 0.20	0.4 ± 0.24	-
Epinephelus quoyanus (Valenciennes, 1830)	Longfin grouper	Carnivores	-	0.2 ± 0.20	-
Variola louti (Forsskål, 1775)	Yellow-edged lyretail	Carnivores	0.4 ± 0.40	1.0 ± 1.00	0.3 ± 0.33

Note: Mean abundance and SE of coral reef fishes for corallivores (Chaetodontidae), herbivores (Acanthuridae, Scaridae, and Siganidae), and carnivore (Haemulidae, Lethrinidae, Lutjanidae, and Serranidae) in Marine Recreational Park: Core Zone (No-take Zone), Utilization Zone (Recreational Zone) and Non-Marine Recreational Park (Fishing Zone) in Pieh per area 350 m^2
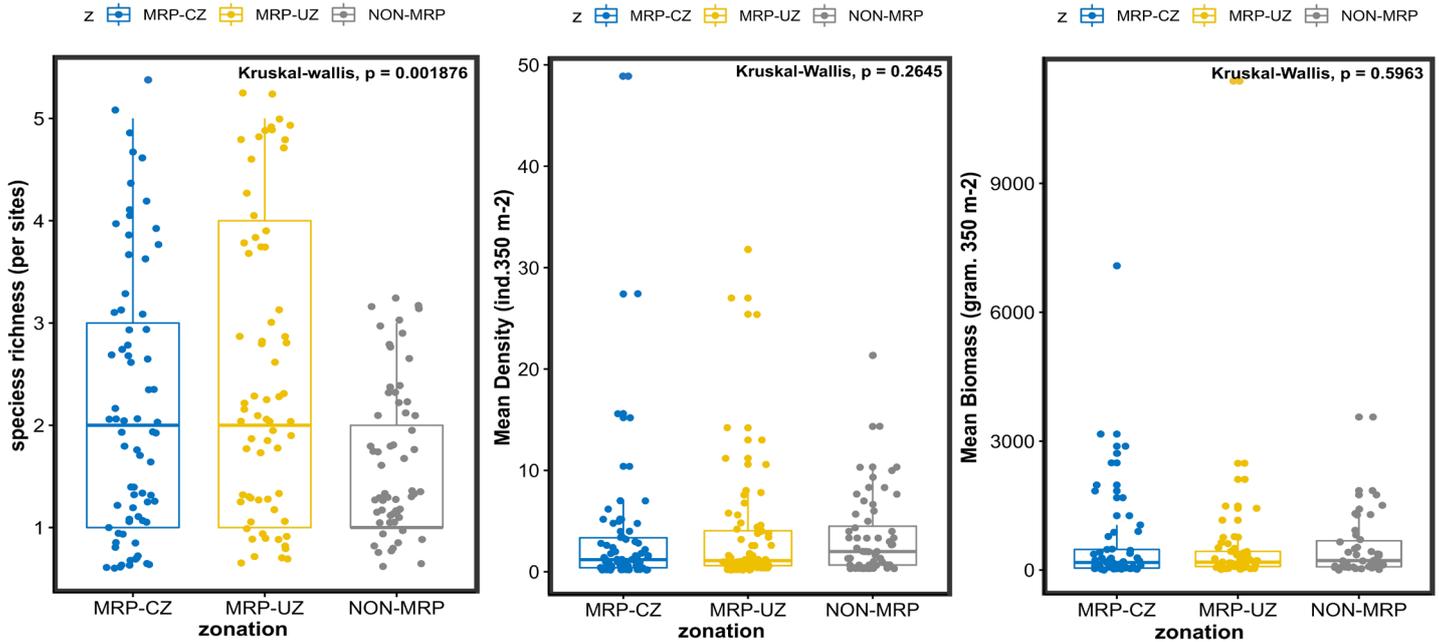
Figure 2. Reef fishes composition (species richness, density, and biomass) in 3 zonation are Pieh MRP (core zone / no-take zone, utility zone, and non - MRP area)
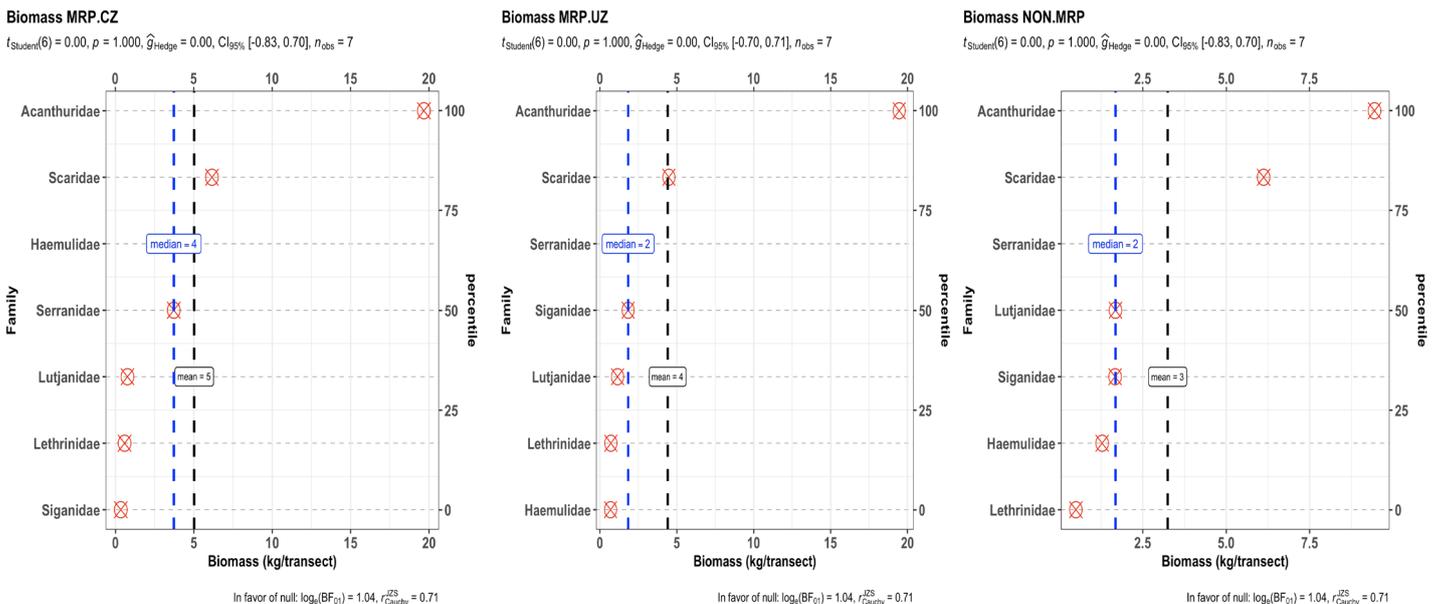

Figure 3. Mean Biomass Rank in 7 Family Coral Reef Fishes (Acanthuridae, Scaridae, Siganidae, Haemulidae, Letrhinidae, Lutjanidae, Serranidae) in Marine Recreational Park for Core Zone (No-take Zone), Utilization Zone (Recreational Zone) and Non-Marine Recreational Zone Park (Fishing Zone) in Pieh (kg/transect)

Based on overall average biomass in different location MRP Area, Acanthuridae represents 55.98% of the total biomass in MRP-CZ, 63.13% in MRP-UZ, and 41.55% in Non-MRP Area (Figure 3). The mean biomass from eight family coral reef fishes in MRP-CZ is higher than MRP-UZ and Non-MRP with 5.02 kg/transect area (MRP-CZ, $p = 0.035$, r = 0.83, CI95% = 0.65, 1.00, n = 7), 4.40 kg/transect area (MRP-UZ, $p = 0.035$, r = 0.83, CI95% = 0.59, 1.10, n = 7) and 3.25 kg/transect area (non-MRP, $p = 0.022$, r = 0.89, CI95% =

0.88, .90, n = 7), respectively (Figure 3). Biomass ranking for each MRP location shows that herbivorous fish groups (Acanthuridae, Scaridae) have higher biomass-compared to the carnivorous fish group (Haemulidae, Lethrinidae, Lutjanidae and Serranidae) where at the MRP-CZ location, Acanthuridae has biomass of (19.67 ± 6.63) kg/transect area, followed by Scaridae (6.15 ± 2.48), Haemulidae (3.91 ± 3.07), Serranidae (3.71 ± 0.44), Lutjanidae (0.76 ± 0.35), Lethrinidae (0.58 ± 0.30),
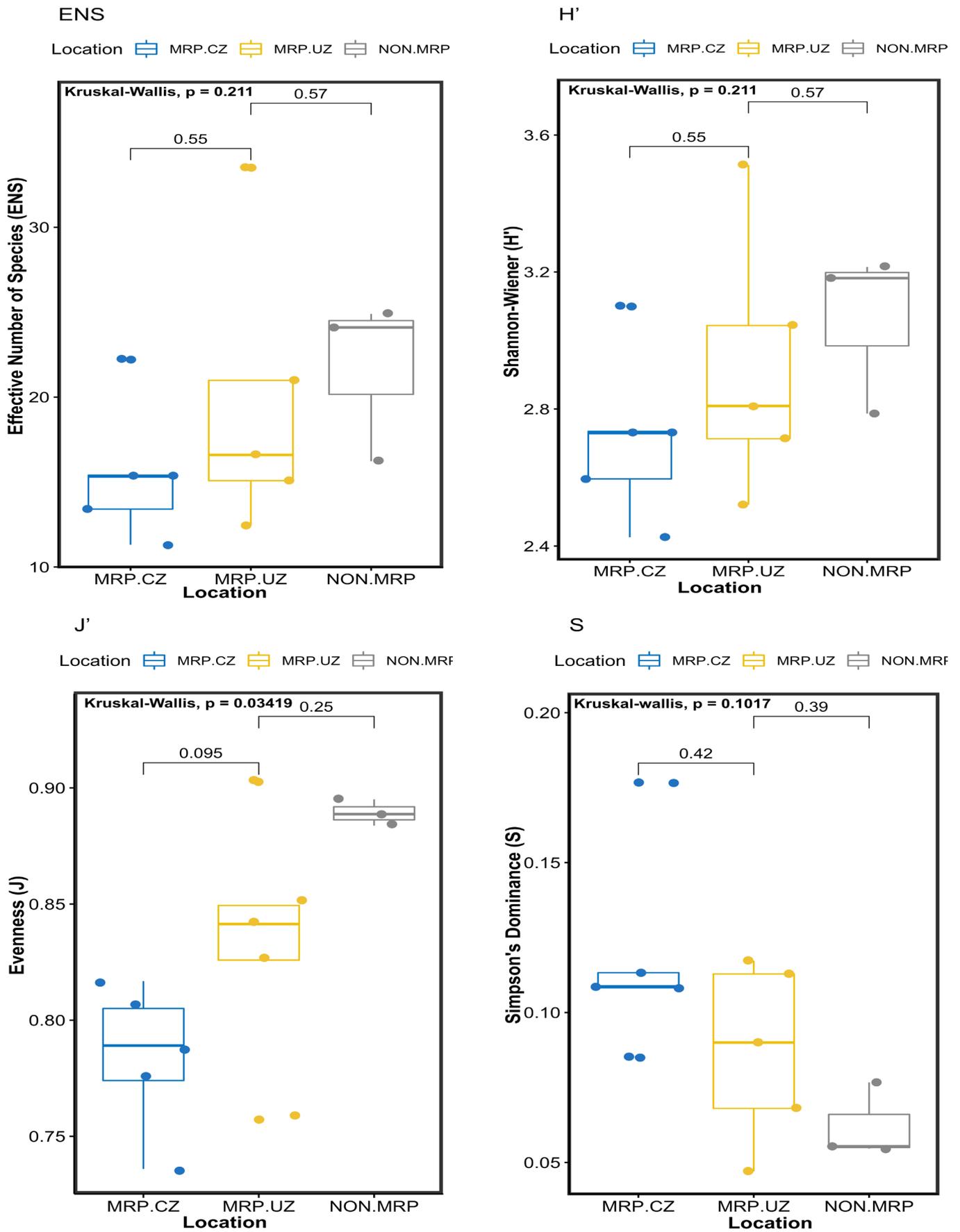
Figure 4. Marine Biodiversity Indices (ENS = Effective Number Species, H' = Shannon-Weaver Index, S= Simpson Dominance, Evenness Equitability (J) between three locations of MRP: Core zone (No-take Zone), Utility Zone (Recreational Zone) and Non-MRP Area (Fishing Zone) in Pieh MRP
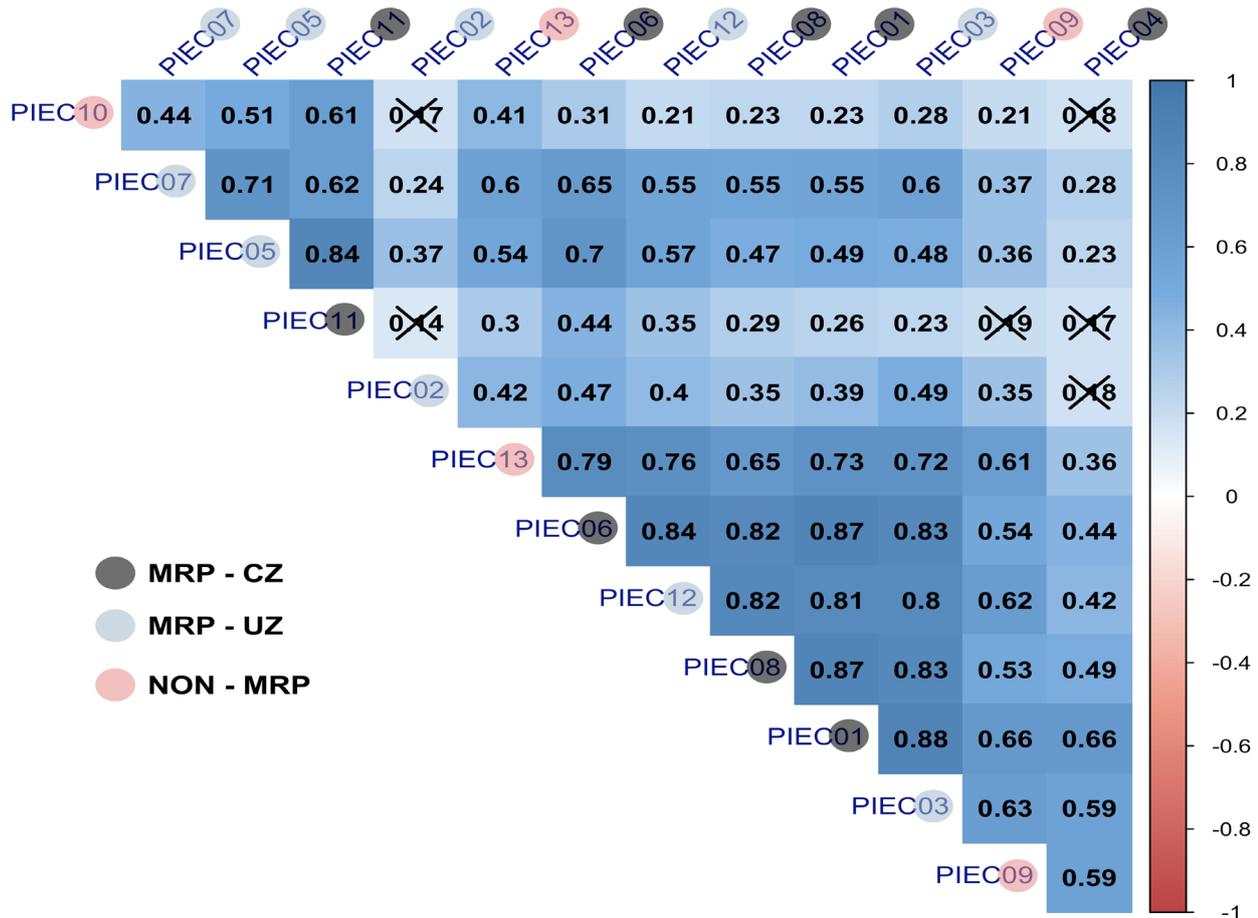
Figure 5. Pearson correlation coefficient matrix with significance level = 0.05, comparing paired sites per location covariates. Negative correlations are shaded white; positive correlations are shaded grey. The strength of the correlation is indicated by dark grey color saturation. The color of dark grey, red, blue and red in circle shape is also indicated the different locations (MRP-CZ, MRP-UZ, and Non-MRP) where the definition of each covariate (y-axis) and its coded counterpart (upper x-axis) are defined per comparison.
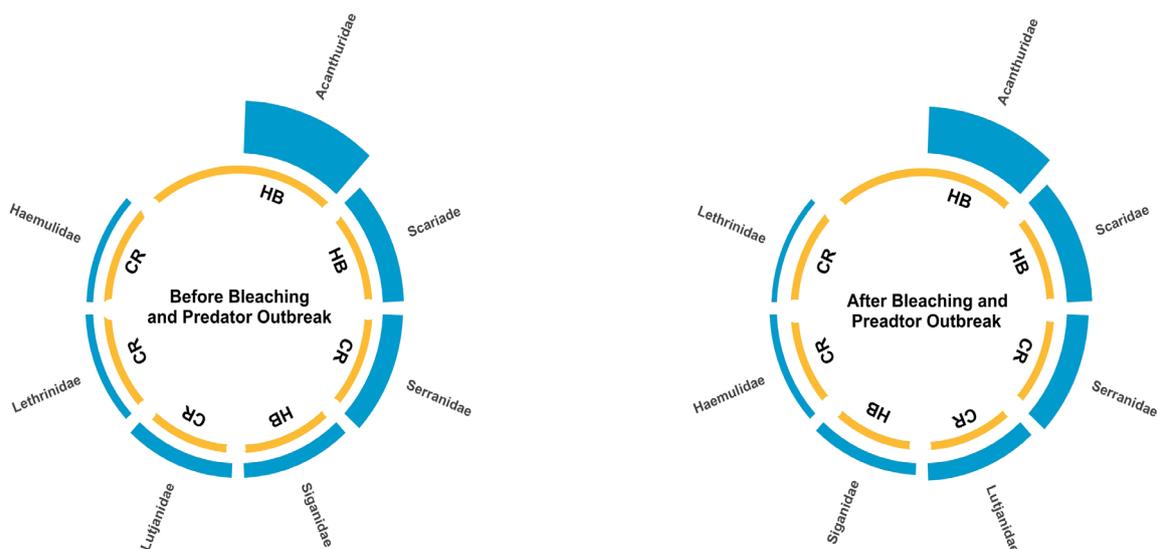

Figure 6. Reef fishes composition before - after bleaching and Acanthaster planci outbreak in Pieh MRP. HB (herbivore), CR (Carnivore)

and Siganidae (0.33 ± 0.27). Biomass conditions in the MRP-UZ have an almost similar situation as MRP-CZ, where Acanthuridae is dominant biomass with 19.45 ± 10.32 kg/transect area, followed by Scaridae (4.48 ± 1.00), Serranidae (2.59 ± 0.61), Siganidae (1.83 ± 1.17), Lutjanidae (1.14 ± 0.43), Lethrinidae (0.71 ± 0.20), and Haemulidae (0.69 ± 0.48). Locations in Non-MRP have different conditions with the other two locations with a significantly lower average biomass where Acanthuridae has mean biomass 9.44 ± 1.33 kg/transect area, followed by Scaridae (6.12 ± 2.94), Serranidae (2.03 ± 1.06), Lutjanidae (1.68 ± 1.28), Siganidae (1.67 ± 0.70), Haemulidae (1.28 ± 1.28), and Lethrinidae (0.50 ± 0.26) (Figure 3).

The indices of diversity from at MRP-CZ / No-take zone area shows an interesting result, the average Shannon index number (H), the Effective Number of Species (ENS), Evenness (J) of coral reef fish on MRP-CZ is lower than location MRP-UZ and Non-MRP. The value of the mean diversity index for the Shannon-Weaver index to MRP-CZ was H' (2.72), ENS (15.53), and J (0.78) (Figure 4). Diversity indices for MRP-UZ are little higher than MRP-CZ, where the value of mean diversity index for Shannon-Weaver index to MRP-UZ was H' (2.92), ENS (19.72) and J (0.84), as for the Non-MRP location has the highest diversity indices, where the value of mean diversity index for Shannon-Weaver index to Non-MRP H' (3.06), ENS (21.74) and J (0.89) (Figure 4). Based on measurement by the Simpson's index for each location (MRP-CZ, MRP, UZ, and Non-MRP), each location was dominated by a single family of Coral Reef Fishes, apparent from its high value. The dominance of Simpson's index shows, MRP-CZ area has a higher value than MRP-UZ and Non-MRP, where the value of Simpson for MRP-CZ, MRP-UZ, and Non-MRP were 0.12, 0.09, and 0.06, respectively (Figure 4). The Kruskal-Wallis test for Shannon- Wiener diversity indices (H') (Kruskal-Wallis chi-squared = 3.11, df =2, p = 0.211) shows no significant difference in diversity in each location. Diversity analysis with Effective Number of Species (ENS) also shows no significant difference between location (Kruskal-Wallis chi-squared = 3.11, df =2, p = 0.211). The equitability analysis with Evenness (J) shows non-MRP location to have significantly higher value than MRP-UZ and MRP-CZ (Kruskal-Wallis chi-squared = 6.75, df =2, p = 0.034). The dominance analysis with Simpson's (S) showed no significant difference between each location (Kruskal-Wallis chi-squared = 4.57, df =2, p =0.10) (Figure 4).

Result from Pearson correlation analysis co-efficient matrix shows there are some strong positive correlations between each sites location in Marine rec-reational Park Pieh (MRP). Several locations in no-take zone area shows strong positive correlation, including Pandan Island (PIEC.08) and Pieh Islands (PIEC.01) (r = 0.87), Air Island (PIEC.06) and Pieh Island (PIEC.01) (r = 0.87), and Pandan Island (PIEC.08) and Air Island (PIEC.06) (r = 0.82) (Figure 5). Another strong positive correlation shows by no-take zone area and utility zone, including Bando Island (PIEC.01) and Pieh Island (PIEC.03) (r = 0.88), Air Island (PIEC.06) and Toran Island (PIEC.12) (r = 0.84) and Pandan Island (PIEC.08) and Toran Island (PIEC.12) (0.82). In addition, several sites have a weak positive and moderate positive correlation with other sites as non-MRP area (PIEC.10 and PIEC.09) and utility zone, PIEC.07, PIEC.05, PIEC.02, PIEC.09, and no-take zone area (PIEC.04) (Figure 5).

3.2 Discussion

The coral bleaching event in 2016 was a global catastrophe on the reefs. Almost in the all tropical or subtropical coral ecosystem reported this phenomenon, including, in Indonesia region (Bachtiar and Hadi, 2019; Ampou et al., 2017; Putra et al., 2019; Wouthuyzen et al., 2018), the Great Barrier Reef (GBR) (Harvey et al., 2018; McMahon et al., 2019; Tebbett et al., 2019; Wismer et al., 2019; Wolanski et al., 2017), Maldives (Nizam et al., 2016), Brazil (Teixeira et al., 2019), Mexico (Johnston et al., 2019), Indian Ocean (Gudka et al., 2018; Head et al., 2019; Ranith and Kripa, 2019; Thinesh et al., 2019), Seychelles (Robinson et al., 2019), Guam (Raymundo et al., 2019), Japan (Nishiguchi et al., 2018), and several other locations that are widespread in the Indo Pacific Ocean. The bleaching phenomena in 2016 in Pieh MRP were largely caused by the increase of Sea Surface Temperature (SST) anomaly due to short duration climate pattern El Niño-Southern Oscillation (ENSO) (Booth and Beretta, 2002). The bleaching event due to El Niño had higher magnitude in 2016 compared to previous events in 2010 (Wouthuyzen et al., 2018) and El Niño's effects are worse due to the long-term global warming (Baird et al., 2009; Hughes et al., 2003) and climate changes (Baker et al., 2008; Hughes et al., 2018). After the bleaching event, the Pieh MRP location hit by A. planci outbreak, and there is no evidence why an outbreak of coral predators occurred in the Pieh MRP location after the bleaching event. There was a strong suggestion that the outbreaks of A. planci in Pieh MRP were due to warmer sea surface temperatures and higher nutrients (Wouthuyzen et. al., 2020). Previous study documented from Haywood et al. (2019), the pattern of A. planci outbreaks have occurred after bleaching in Pilbara offshore bioregion. However, the reason on why this pattern can occur in Pieh MRP has not yet discovered. It can only be ensured that suitable habitat conditions and food resources on the reef can support an outbreak of A. planci. The high population of A. planci
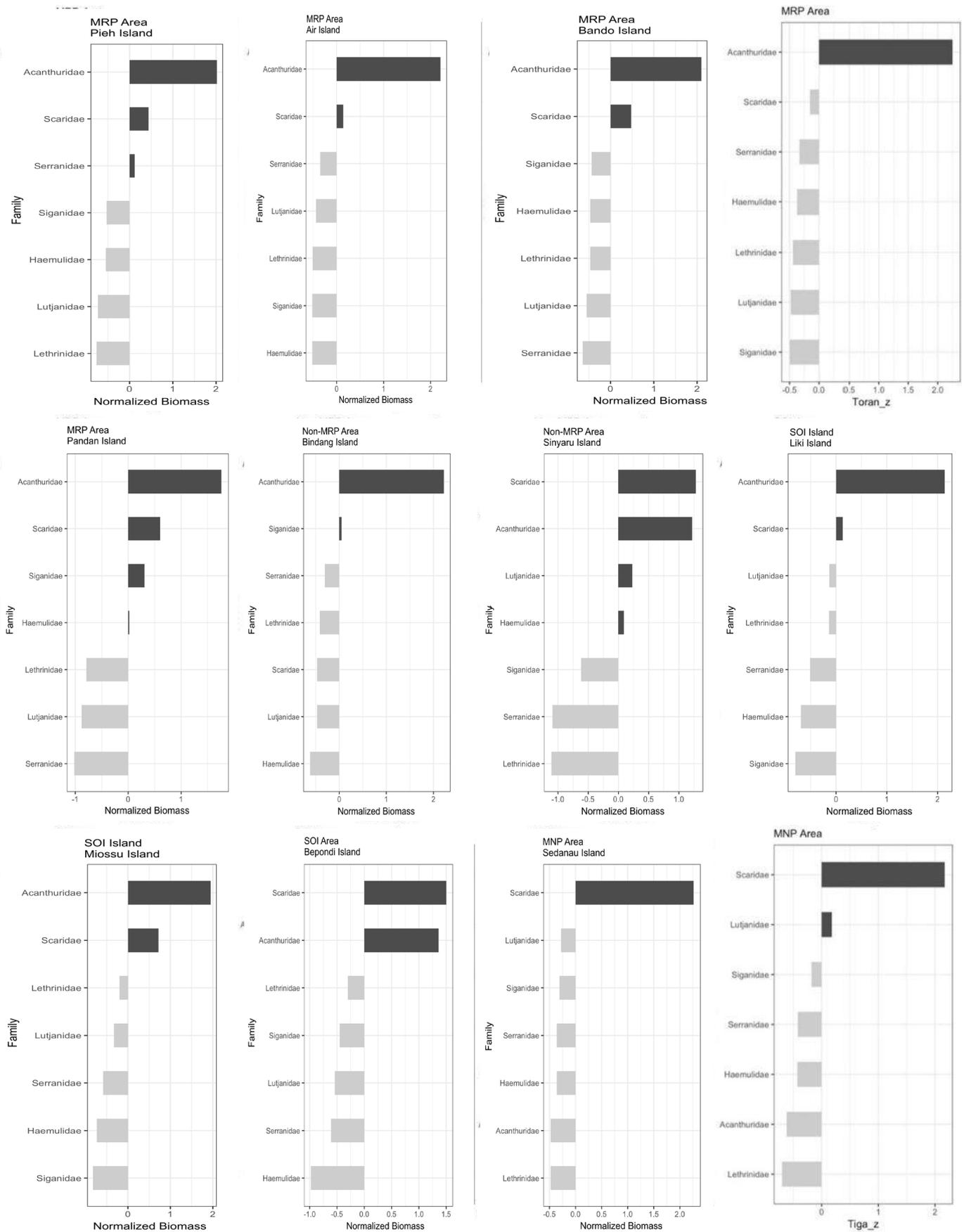
Figure 7. Diverging Bars of Biomass Coral Reef Fishes from Herbivore group (Acanthuridae, Scaridae, Siganidae) and Carnivore group (Haemulidae, Lethrinidae, Lutjanidae, Serranidae) in some oceanic Island in Indonesia; Pieh Area (Pieh island, Air island, Bando island, Toran island, Pandan island, Bindalang island, Sinyaru island); Papua area (Liki island, Miossu island, Bepondi island); Natuna area (Sedanau island, Tiga island). MRP (Marine Recreational Park); SOI (Smalls Outer Island); MNP (Marine National Park). Dark Grey is above the average of Biomass; Light Grey is below the average of biomass.

is the response to seasonal variation in temperature and storm-generated by wave (Haywood *et al.*, 2019), and increase in the concentration of *chlorophyll-a* as food availability for *A. planci* larvae due to upwelling process that transport rich-nutrient to coral ecosystem (Baker *et al.*, 2008), but Pratchett *et al.* (2017) explained the *A. planci* outbreak phenomenon still cannot be explained and remains mostly unresolved. In current study, it is aimed to highlight no significant change in the fish composition after the bleaching and *A. planci* outbreak in Pieh MRP (Figure 6). Before and after non-anthropogenic disturbance events, the composition of fish in Pieh MRP is still dominated by herbivores fishes (Acanthuridae and Scaridae), followed by Serranidae as carnivore fish. (Figure 6).

The similar fish composition before and after disturbance in Pieh MRP shows the coral reef fish ill endurance on bleaching event and *A. planci* outbreak, but the coral reef ecosystem requires full recovery condition. Current study result suggested the need for integrated management after post-disaster in Pieh MRP due to the possibility of accelerated recovery in the coral reef ecosystem after post-disaster for the next future event. The possibility of accelerated recovery condition after coral bleaching and *Acanthaster outbreak* in Pieh MRP enhanced by geographical condition and physical environment that reduce the temperature of the sea (Morgan *et al.*, 2017; West and Salm, 2003). Pieh MRP is a group of islands facing the Indian Ocean and included in the category of open ocean exposure (OOE) islands. The OOE islands usually present a coral reef system that is strongly influenced by strong waves and currents (Teixeira *et al.*, 2019), which in turn supports the occurrence and abundance of planktivorous (Floeter *et al.*, 2004; Pinheiro *et al.*, 2011) and reduce bleach event (Baker *et al.*, 2008). Most of the small Island condition Pieh MRP had a deeper reef with a reef slope near 90º (Figure 1) (Abrar *et al.*, 2014). The topography of reef slope potential had the strength of upwelling conditions and some locations in western Indonesia at the Indian Ocean region has generally had the strength of upwelling condition as Mentawai Island(Abram *et al.*, 2003). The reef slope condition in open ocean exposure islands can change dramatically to reduce the high temperature due to the impact of cold-water upwelling conditions and the previous study by Wouthuyzen *et al.* (2018) explained that strong cold-water upwelling could assist in the recovery process from coral bleaching event in the Indian Ocean. Another oceanographic phenomenon that allows Pieh MRP accelerated recovery in bleaching coral was strong current and waves, where the rapid mass of water flow can protect coral from bleaching event by removing harmful oxygen radical (Grimsditch and Salm, 2006). The accelerated recovery of coral reefs in Pieh MRP was also influenced by the dominant composition of herbivorous fish. Herbivore fish has an essential

role after the disturbance by reducing algal (McManus and Polsenberg, 2004). An interesting result of this study, all small outer islands that are facing the ocean (OOE) have a similar fish composition pattern as herbivorous fish dominate in coral reef ecosystems. Previous studies show the outer island and OOE can provide a large presence roving herbivores fish (Friedlander and DeMartini, 2002; Pinheiro *et al.*, 2015; Sandin *et al.*, 2008b). The large presence ofherbivorous fish also plays a role in maintaining the sudden growth of macroalgae after bleaching. Based on current study, it has found several important herbivore fish species as macroalgal remover in Pieh MRP which are *Naso unicornis* and *Chlorurus sordidus*, both were recorded feeding on macroalgal in Seychelles Islands (Chong-Seng *et al.*, 2014); and *Zebrasoma veliferum* has been recorded feeding on macroalgal in Great Barrier Reef (Hoey and Bellwood, 2011). The presence of these species is expected to help the recovery process in Pieh, especially from macroalgae.

After bleaching disturbance and *A. planci* outbreaks, herbivore fishes species were dominated (60.73 %) in Pieh MRP, including the three most abundant species were *Ctenochaetus striatus*, *A. lineatus* and *N. hexacanthus* (Table 2 and Figure 3). The dominance of these herbivores fishes group species contributes to the higher biomass in all area in Pieh MRP: core zone (no-take zone), utility zone (recreational zone), and non-MRP (fishing zone) (Figure 3). The presence of herbivorous fish after a non-anthropogenic disturbance provides important information on the resilience of coral reef ecosystems in Pieh MRP. Herbivorous fish have an important role in coral reef resilience after climate change phenomena (Hughes *et al.*, 2007) including coral bleaching and *A. planci* outbreaks due to their ability to control blooms of turf algal and fleshy seaweeds and support coral recruitment and growth (Mumby *et al.*, 2006; 2007). The mass-bleaching coral events may result in massive algal overgrowth (Diaz-Pulido and McCook 2002) and algae become dominant in competitive interaction when the coral colonies suffer from bleaching (Swierts and Vermeij, 2016). The condition after disturbance is an essential indicator of understanding the resilience of the coral ecosystem and the presence of herbivorous fish can strengthen the resilience. The result of current study showed that after catastrophic disturbance herbivore fish composition (species richness, abundance, and biomass) was higher within the no-take area (core zone) than in utility zone and non-MRP area (Table 2). This finding was consistent with previous studies that suggested biomass of herbivorous fishes is higher within MPAs area than in non-protected areas (Graham *et al.*, 2007; Wilson *et al.*, 2012) and the fishing practice was a major cause to present the low composition of herbivorous fish in the non-MPA region. If several herbivorous fishes species

from Scaridae, Siganidae, and Acanthuridae family continues to be fishing objects after catastrophic disturbance (coral bleaching and *A. planci* outbreak), it will have a significant impact on the recovery process on coral reefs in Non-MRP Pieh area.

The dominant presence of herbivorous fish, *C. striatus*, *A. lineatus,* and *N. hexacanthus* after non-anthropogenic disturbance in the Pieh MRP region provide valuable information for the recovery process of coral reef ecosystems condition after post-disaster (Table 2 and Figure 3). This family group of fish has an essential part of its diet consisting of detritus and calcified ingredients (Scaridae and Acanthuridae); the abundance and distribution of these fish are strongly influenced by warmer locations (Floeter *et al.,* 2004). This confirms the process of increasing water temperatures in 2016 and 2017 in Pieh MRP that resulted in coral bleaching which in turn increased the abundance of this group of fish. The dominance of this group is also strengthened by the coverage of benthic of the coral reef area, which is dominated by Dead Coral Algae and rubble which are primary components diet for these fish groups to grow. The increase in the number of herbivorous fish groups in Pieh MRP indicated a dramatic increase in the representation of dead corals and rubble. The case of mass coral bleaching in 2016 at Pieh MRP resulted in a significant decrease in live coral cover and an increase in dead corals. The research from Putra *et al.* (2018) explained that there was a positive correlation of biomass of herbivorous fish with dead coral with algae (DCA), especially for herbivorous fish in the Scaridae family, and Sandin *et al.* (2008a) found there was a significant positive correlation between herbivorous biomass and macroalgae cover, but Newman *et al.* (2006) and Sandin, *et al.* (2008b) noticed there was a negative correlation between herbivorous fish and fleshy algae. The Pieh MRP is dominated by Acanthuridae especially from *C. striatus*. On the coral reef ecosystem, *C. striatus* plays a critical role in several key ecosystem processes (Tebbett *et al.,* 2018) and has a contribution to remove the epilithic algal matrix (EAM) or algal turf (Tebbett *et al.,* 2017). Additionally, this study found several other important herbivorous fish feeding EAM beside *C. striatus*, including *Scarus ghobban, S. dimidiatus, S. niger, S. oviceps, S. rubroviolaceus, Chlorurus bleekeri,* and all these fishes have been identified consumer EAM (Bellwood *et al.,* 1990; Vergés *et al.,* 2012). Based on the previous study, the colonization of EAM or algal turf in coral was a consequence of bleaching events and responsible for coral mortality (Diaz-Pulido and McCook, 2002). The interaction of the turf algal and coral causes tissue damage and decrease in pigmentation in coral (Wild *et al.,* 2014) due to turf algae acting as poison to scleractinian coral and able to kill coral tissue (Jompa and McCook, 2003; Titlyanov *et al.,* 2007). Brown *et al.* (2018) suggested that low

turf algal coverage indicates a healthy coral reef. The presence of *C. striatus* can prevent massive coral deaths due to tissue damage by algae turf after the catastrophic coral bleaching. This fish consumes more intensively on sparse/short algae turf and significantly removes more algae turf per hour than other herbivore fish from the Acanthuridae family (Marshell and Mumby, 2012).

In current study, it has found several groups of herbivorous fish from *A. tristis*, *C. striatus*, *C. binotatus,* and *S. niger* species with less than 5 cm in size which indicates an increase in number and high regeneration process of herbivore fish groups in Pieh MRP. The increased herbivorous grazing in MPAs results in substantial reductions in algae and encourage coral recovery (Mellin *et al.,* 2016; Mumby *et al.,* 2006), but in reality, the herbivorous fish populations are strongly influenced by region. Current study's results show that after bleaching and *A. planci* outbreak, the mean abundance of *C. striatus* was two times higher in the core zone / no-take area (48.4 ± 10.97) than non-MRP / fishing zone area (21.3 ± 7.62) (Table 2). Although *C. striatus* is one of the most abundant surgeonfish in the Indo-Pacific coral reef, but their existence and population could be threatened because large environmental disturbances (Lin *et al.* 2021) or several fishing practice (Jones *et al.,* 2004). However, this is a fundamental problem with the resilience process. If *C. striatus* and several herbivorous fish populations become lower in number and provide low grazing pressure to algae, herbivores may not be able to resist increasing algal population and macroalgae bloom (Hughes *et al.,* 2007; Pratchett *et al.,* 2011). Unfortunately, it brings a rapid phase in changing from coral-dominated to algal-dominated (Pratchett *et al.,* 2011). The increasing number of dead coral algae (DCA) serves as herbivore fish food source and causes a dramatic increase in the grazing group of herbivore fish (Jones *et al.,* 2004; Putra *et al.,* 2018). In addition, herbivore fish populations can still play an essential role in performing the function of regeneration in the shifting phase of the reef, which is dominated by algae. Grazing herbivorous fish helps maintain the health of coral reefs dominated by algae (Bellwood *et al.,* 2004; Thibaut and Connolly, 2013). The only significant correlation noted between the benthic and fish assemblages was a positive relationship between herbivorous fish biomass and macro-algal cover (Putra *et al.,* 2018; Sandin, *et al.,* 2008b). It is most detrimental if phase shift from coral-dominated to algal- dominated are not able to regenerate optimally and apprehended the reefs without coral, it will no longer support diverse reef fish but will be dominated numerically by a small group of reef fish species that prefer algae or rubble (Jones *et al.,* 2004). A basic understanding of the phase shift from coral-dominated to the algal-dominated mechanism that informs coral reef degradation will affect coral reef fish communities differently but limited time scales. The

long-term effects of reducing coral cover in coral reef fish communities can arise through mechanisms that don't have a direct impact on demographic levels such as mortality but change regularly and cause a significant reduction in the physiological conditions of the population (Feary *et al.*, 2009).

Other findings from current study showed, after the disturbance, the number of species and populations from corallivore fishes (Chaetodontidae) has decreased. Still, there has been an increase in carnivorous fish populations (Table 3). Several species of corallivore fishes (Chaetodontidae) were not found at the time of the study, including *Chaetodon bennetti* (obligate corralivore), *C. xanthocephalus* (Facultative corraLivore)*,* and *Heniochus diphreutes* (generalist corralivore). This study's result showed, more than two years, the population of corallivores fishes has reduced from (38 ± 3.59) in 2014 to (31 ± 4.13) in 2019 (Table 3). The reduction of corallivores fish composition caused by nutritional deficit and decrease in aggressive behavior among corallivore fish due to after bleaching event (Keith *et al.*, 2018). The aggression behavior provides information changing the population of corallivores fishes due to competing individuals in obtaining enough food resources (Tricas, 1989; Yahya *et al.*, 2011) and influence in reducing butterflyfish abundance (Pratchett *et al.*, 2006) in the long term (> 4 weeks) lethal or sub-lethal effect of feeding form (Cole *et al.*, 2009). In short term, corallivores are found to impose further stress and increase the feeding rate on bleach coral (Cole *et al.*, 2009) and may contribute to increase mortality of bleached corals. If it lasts longer, food sources are limited and may contribute in reducing in the abundance of corallivore fish. On the other hand, the population of carnivore fish increased more than 20 percent and biomass increased more than 30 percent (Table 3). Explosion of herbivorous fish populations after bleaching and *A. planci* outbreaks provide a substantial food source for carnivorous fish. Current study shows, after the bleaching and *A. planci* outbreak, the population of herbivorous fishes had increased (6.77 ±0.78).

The increased population of herbivorous provides food source for carnivorous fishes which increased carnivorous fish population and biomass (Table 3). *C. striatus* contributed as the highest food source for several carnivorous fish, especially for *C. argus*. In current study, *C. argus* is the dominant species for the carnivorous group. After the disturbance, the population of *C. argus* increased the abundance of *C. striatus* (Table 2). A previous study showed 16.9% (Hawaii) composition of the fish portion of the diet of *C. argus* was Acanthuridae family with %IRI = 20.9 (Dierking *et al.*, 2009). The diet of *C. argus* is dependent on considering the prey availability and influence on prey composition (Meyer and Dierking, 2011). The high dominance of *C. striatus* (Quoy & Gaimard, 1825) (Acanthuridae) and another herbivorous group after disturbance in Pieh MRP impact on an increase in population *C. argus*.

Biomass and the diversity of reef fish have an essential role in maintaining the structure and resilient process of coral reefs (Chong-Seng *et al.*, 2014; McClanahan *et al.*, 2011; Thibaut and Connolly, 2013). The results of the study showed MRP area has higher biomass than the non-MRP area (Figure 3). The non-MRP area is a location that is permitted for fishing activities by local fishermen. Consequently, this area has an impact on reducing fish biomass, especially economically targeted fish from carnivorous groups (Haemulidae, Lethrinidae, Lutjanidae, Serranidae) (Table 2). Location with the highest fish biomass is mostly within the boundaries of marine protected areas, suggesting that human exploitation is a major factor in reducing fish biomass (Sandin *et al.*, 2008a). The core zone / no-take zone and utilization zone in the MRP area have almost similar mean biomass value of reef fish. This explains the marine recreational activities in utilization zone Pieh MRP have no impact on reducing fish biomass and this also confirms that the management of the MRP area in Pieh by the government has been correctly implemented. The overall findings indicate that the highest reef fish abundance and biomass were in the MRP zone management control area. The results of this study also show that

Table 3. The changing coral reef composition before (2014) and after (2019) bleaching phenomena and *Acanchaster planci* outbreaks

Condition	Corralivore		Herbivore		Carnivore	
	Species Richness (SR)	Mean abundance ± se	Mean SR ± se	Mean biomass ± se	Mean SR ± se	Mean Biomass ± se
Before Pressure (2014)	21	38 ± 3.59	4.10 ± 0.70	6.66 ± 1.47	5.70 ± 0.87	4.34 ± 0.88
After Pressure (2019)	17	31 ± 4.13	6.77 ± 0.78	6.40 ± 1.23	7.69 ± 0.85	7.01 ± 1.50

Note: The Corallivore counted for Chaetodontidae family; Herbivore (Scaridae and Siganidae) family; Carnivore Fish (Haemulidae, Lethrinidae, Lutjanidae, Serranidae) family. Mean abundance (individual /350m2) and mean biomass (kg/350 m2).

management of control in the MRP area must be part of a future coral reef management strategy. Not only in Pieh MRP but in several other locations that have a high potential of marine tourism can hold integrated controls between conservation and recreation to create sustainable marine tourism (Madduppa *et al.*, 2014; Rudi *et al.*, 2014). The process of restoring coral reef ecosystems caused by post-bleaching phenomena and massive predators of *A. planci* on the Pieh MRP area could be executed by implementing excellent management, including reducing recreational activities in utility zone in the MRP area (Hughes *et al.*, 2003). When the appearance of benthic fleshy algae has decreased dramatically, the presence of large predators from car-nivorous fish family (Haemulidae, Lethrinidae, Lut-janidae, Serranidae), and recovery in the population of corallivorous fishes in the Pieh MRP area, this indicator shows that in Pieh MRP area has contributed recovery from post-bleaching and massive predator of *A. planci* outbreaks and provides a good indicator for coral reef health (Chabanet *et al.*, 2016).

The coral reef recovery process after non-anthropogenic disturbance must be assisted with integrated management of Pieh MRP, especially those related to tourism activities in the MRP area. Tourism contributes to several benefits. Proper tourism management contributes to increasing the economy of the community. Some of the benefits of tourism development include increasing employment opportunities, increasing economic income, increasing population living standards, and promoting culture (Wu, 2014). On the other hand, if tourism development not appropriately managed, it can make a significantly negative contribution, especially to the environment. Comparative research on recreational zones involved in diving and non-diving activities on coral reefs proves that diving with SCUBA has a significant impact on the area of coral reefs visited, especially on hard corals (Tratalos and Austin, 2001) where 15% of diving activities give damage on coral reefs, with diving fins being the leading cause (95%) of all damage (Rouphael and Inglis, 1997). In addition to implementing restrictions on activities in the utilization zone, providing qualified control in recreational activities is particularly critical at Pieh MRP. SCUBA diving activities at high recreational levels affect coral communities through direct contact by divers from SCUBA equipment on sensitive benthic organisms (Hammerton, 2017; Luna *et al.*, 2009). Snorkeling and SCUBA activities in the MRP area significantly change benthic topographic features and cause physical damage to coral polyp organisms which are the primary organisms as health indicator of coral reef ecosystems and over the past two decades, there is increasing evidence that many of the biological and aesthetic were damaged by recreational diving (Hammerton, 2017).

4. Conclusion

The Biomass and the diversity of reef fish have an essential role in maintaining the structure and process of coral reefs. The herbivorous fish is the most dominant group in the locations of Pieh MRP and Non-MRP. After disturbance in Pieh MRP, the coral reef fishes still shows endurance after the bleaching and *A. planci* outbreaks. However, the composition of corallivorous has decreased while herbivorous and fish populations (diversity, abundance and biomass) has increased. The herbivore fishes could shift algal reef assemblages to states that are beneficial for corals and improve corals' ability to thrive, thereby increasing corals' ability to recover from destructive events such as bleaching and *A. Planci* outbreaks. It is a necessity to restore the condition of coral reefs in the MRP area of Pieh from the phenomenon of mass coral bleaching and attack by *A. planci*. Management strategy is important to protect herbivorous fish and other fish by constricting supervision at Pieh MRP. In addition, proactive management of the Pieh MRP aims to ensure sufficient stock of herbivores before the re-occurrence of future bleaching events. With sufficient stocks of herbivory in Pieh, the condition of coral reefs in Pieh MRP will recover, although it will take several years. This study suggests that herbivore management is part of a broader strategy to manage and reduce threats to coral reefs in Pieh.

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Authors' Contributions

All authors have contributed to the final manuscript. The contributions of each author are as follows, Risandi, Rikoh, and Abrar; collected the data, compiled the manuscripts, and designed the visualization from R software. Risandi; made a graphic visualization of the data analysis. Ni Wayan Puranmasari, and Jayedul; helped to compile information related to the research. All authors discussed the results and contributed to the final manuscript.

Conflict of Interest

The authors declare that they have no competing interests.

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