The Pattern of Herbivorous Fish Assemblages in The Western and Eastern Outermost Island Indonesia

Risandi Dwirama Putra1, Rikoh Manogar Siringoringo2, Muhammad Abrar2, Ni Wayan Purnamasari2, Agung Dhamar Syakti1

1Marine Science Department, Faculty of Marine Science and Fisheries, Maritime Raja Ali Haji of University, Jl. Raya Senggarang, Campus UMRAH Senggarang, Tanjungpinang 29111, Indonesia
2Research Centre of Oceanography, Indonesia Institute of Science, Pasir Putih I,Road. East Ancol, North Jakarta 14430, Indonesia

*Corresponding author: risandi@umrah.ac.id

Received 15 May 2020; Accepted 2 August 2020; Available online 31 December 2020

ABSTRACT

The herbivorous fishes have been considered as a critical functional group and have capability maintaining coral reef resilience and avoiding coral-algal phase-shifts. The present condition shown, almost in tropical reef location, alga has dominated coral, even in the small outer island. The requirement to conduct comprehensive basic research in studying the patterns and composition of herbivorous fish, especially on the small outer islands. Twelve coral reef sites in eastern Indonesia (Liki Islands) and western Indonesia (Natuna Island) used as a research location for comparing the structure patterns of herbivorous fish communities (diversity, density, and body size) using the Underwater Visual Census (UVC) method. There was different pattern of herbivorous fishes families in Liki Island and Natuna Islands, where Acanthuridae is dominant in eastern Indonesia (Liki Islands), including Ctenochaetus striatus (41,00 ± 11,72 se) individuals/350m2, A. maculiceps (23,33 ± 13,61 se) individuals/350m2, Naso hexacanthus (18,67 ± 6,34 se) individuals/350m2 while Scaridae is dominant in western Indonesia (Natuna island), including Scarus rivulatus (31,67 ± 10,61 se) individuals/350m2, Chlorurus sordidus (30,00 ± 8,52 se) individuals/350m2 and Scarus quoyi (19,00 ± 9,73 se) individuals/350m2. Based on herbivore fishes composition Liki Island has a higher density and biomass compared to Natuna Island.

Keywords: herbivore, fish, coral, small outer island, Indonesia

The herbivorous fishes have been considered as a critical functional group and have capability maintaining coral reef resilience and avoiding coral-algal phase-shifts. The present condition shown, almost in tropical reef location, alga has dominated coral, even in the small outer island. The requirement to conduct comprehensive basic research in studying the patterns and composition of herbivorous fish, especially on the small outer islands. Twelve coral reef sites in eastern Indonesia (Liki Islands) and western Indonesia (Natuna Island) used as a research location for comparing the structure patterns of herbivorous fish communities (diversity, density, and body size) using the Underwater Visual Census (UVC) method. There was different pattern of herbivorous fishes families in Liki Island and Natuna Islands, where Acanthuridae is dominant in eastern Indonesia (Liki Islands), including Ctenochaetus striatus (41,00 ± 11,72 se) individuals/350m2, A. maculiceps (23,33 ± 13,61 se) individuals/350m2, Naso hexacanthus (18,67 ± 6,34 se) individuals/350m2 while Scaridae is dominant in western Indonesia (Natuna island), including Scarus rivulatus (31,67 ± 10,61 se) individuals/350m2, Chlorurus sordidus (30,00 ± 8,52 se) individuals/350m2 and Scarus quoyi (19,00 ± 9,73 se) individuals/350m2. Based on herbivore fishes composition Liki Island has a higher density and biomass compared to Natuna Island.

Keywords: herbivore, fish, coral, small outer island, Indonesia

Received 15 May 2020; Accepted 2 August 2020; Available online 31 December 2020

ABSTRACT

The herbivorous fishes have been considered as a critical functional group and have capability maintaining coral reef resilience and avoiding coral-algal phase-shifts. The present condition shown, almost in tropical reef location, alga has dominated coral, even in the small outer island. The requirement to conduct comprehensive basic research in studying the patterns and composition of herbivorous fish, especially on the small outer islands. Twelve coral reef sites in eastern Indonesia (Liki Islands) and western Indonesia (Natuna Island) used as a research location for comparing the structure patterns of herbivorous fish communities (diversity, density, and body size) using the Underwater Visual Census (UVC) method. There was different pattern of herbivorous fishes families in Liki Island and Natuna Islands, where Acanthuridae is dominant in eastern Indonesia (Liki Islands), including Ctenochaetus striatus (41,00 ± 11,72 se) individuals/350m2, A. maculiceps (23,33 ± 13,61 se) individuals/350m2, Naso hexacanthus (18,67 ± 6,34 se) individuals/350m2 while Scaridae is dominant in western Indonesia (Natuna island), including Scarus rivulatus (31,67 ± 10,61 se) individuals/350m2, Chlorurus sordidus (30,00 ± 8,52 se) individuals/350m2 and Scarus quoyi (19,00 ± 9,73 se) individuals/350m2. Based on herbivore fishes composition Liki Island has a higher density and biomass compared to Natuna Island.

Keywords: herbivore, fish, coral, small outer island, Indonesia

Research Article

journal homepage: http://ojs.omniakuatika.net

http://dx.doi.org/10.20884/1.oa.2020.16.2.805
1. Introduction

Herbivorous fishes have a significant role in reducing algal standing stock in the coral ecosystem (McManus & Polsenberg, 2004), and the population of herbivorous fishes consequence of influence in algal cover after the loss of corals (Rongo & van Woensik, 2013; Sheppard, Ateweberhan, Bowen, et al., 2012; Wilson, Graham, Pratchett, et al., 2006). There is an interesting hypothesis in interaction from coral, algae, and herbivorous fishes. After the loss of coral cover, the densities of herbivores increased significantly (Ruppert, Travers, Smith, et al., 2013). The increase of herbivorous fish can be related to higher food availability (Adam, Schmitt, Holbrook, et al., 2011; Fong, Frias, Goody, et al., 2018) and might be expected the abundance of algae trigger herbivorous fishes to grow following an extensive coral loss (Jayewardene, 2009). The hypothesis confirms the role and importance of herbivorous reef fishes in maintaining coral reef resilience and avoiding coral-algal phase-shifts (Bellwood, Hughes, Folke, et al., 2004; Hughes, Baird, Bellwood, et al., 2003; Hughes, Rodrigues, Bellwood, et al., 2007). The shifting coral to algal indicates reef in a condition not healthy due to the intensive of anthropological pressure and natural disturbance. Without the capacity to support herbivores, these reefs are likely to be vulnerable to algal overgrowth and potential long-term phase shifts (Rogers, Blanchard, Newman, et al., 2018).

Understanding algal succession and the influence of herbivorous fishes is central to the development of knowledge about how coral reef ecosystems may respond to and recover from increasing anthropogenic pressures (Burkepile & Hay, 2010; Ceccarelli, Jones & McCook, 2011). The herbivorous fishes have been considered as a critical functional group and have capability maintaining coral reef resilience and avoiding coral-algal phase-shifts (Bellwood et al., 2004; Dromard, Bouchon-Navaro, Harmelin-Vivien, et al., 2015; Fong et al., 2018; Hughes et al., 2003, 2007). The strong functional group of herbivorous fish may maintain the coral reef and remove algal states (Fong et al., 2018), with consuming primary benthic algae and regulate competition between algae and reef-building corals (Edwards, Friedlander, Green, et al., 2013; Wilson, Bellwood, Choat, et al., 2003). The present condition shown, almost in tropical reef location, alga has dominated coral, even in the small outer island and several oceanic islands in pacific.

There are only a few small studies on the small outer islands that discuss the ecological function of coral reefs. Some obstructions become significant problems in researching the small outer islands, including high costs, difficult access, limited transportation and accommodation, limited research equipment, and several islands that are uninhabited and far from the mainland. Therefore, knowledge to understand the ecological conditions of ecosystems in the small outer islands still needs to be explored further. Decades have shown that most ecosystems are subject to massive pressure and degradation caused by anthropological factors (Brown, Bender-Champ, Kubicek, et al., 2018; Ferrigno, Bianchi, Lasagna, et al., 2016; Quimbayo, Dias, Kulbicki, et al., 2019) and natural disturbance (Guest,

| Sites .id | Latitude | Longitude | Location |
|-----------|----------|-----------|----------|
| LIKC.01   | 1° 37' 34.255 S | 138° 43' 43.411 E | Liki Island |
| LIKC.02   | 1° 38' 43.451 S | 138° 46' 47.402 E | Liki Island |
| LIKC.03   | 1° 37' 42.305 S | 138° 44' 42.862 E | Liki Island |
| LIKC.04   | 1° 36' 52.902 S | 138° 42' 49.052 E | Liki Island |
| LIKC.05   | 1° 35' 32.028 S | 138° 43' 56.024 E | Liki Island |
| LIKC.06   | 1° 37' 12.299 S | 138° 44' 54.210 E | Liki Island |
| NTNC.01   | 4° 02' 28.716 N | 108° 20' 12.119 E | Natuna Island |
| NTNC.02   | 4° 08' 20.004 N | 108° 06' 55.439 E | Natuna Island |
| NTNC.03   | 4° 12' 33.336 N | 108° 10' 53.075 E | Natuna Island |
| NTNC.04   | 3° 41' 15.360 N | 108° 04' 23.627 E | Natuna Island |
| NTNC.05   | 3° 34' 43.644 N | 108° 04' 45.804 E | Natuna Island |
| NTNC.06   | 3° 59' 30.156 N | 108° 22' 32.160 E | Natuna Island |
The research studies from the impact on acute threat and chronic stress from coral reef ecosystem which happened in the last few decades in the small outer islands were still limited, including the rising sea surface temperatures that cause coral bleaching (Pratchett, Bridge, Brodie, et al., 2019; Wild, Hoegh-Guldberg, Naumann, et al., 2011), nutrification (Leão, Kikuchi & Oliveira, 2019), destructive fishing (Edinger, Jompa, Limmon, et al., 1998; Kunzmann & Samsuardi, 2017), and over-exploitation (Martin, Montaz, Jordan, et al., 2016; Rogers et al., 2018) in the small outer island. All of this disturbance are the major problems being faced by coral reef ecosystems in the small outer island with causes phase-shifts in coral dominated to algal dominated, and the level of anthropogenic activities was very different in the western and eastern regions of Indonesia. In the long term, the pressure will have a significant impact on coral mortality (Jompa & McCook, 2002). The herbivore fishes had an essential role in maintaining the changing phase-shift coral-algal dominated, and the pattern of herbivore fishes composition play an important role in the process of coral reef recovery in the small outer island. They understand the shifting phase from coral to algae dominated, and the pattern of herbivore fishes composition help to prevent massive coral mortality and the needs required for comprehensive basic research to study the pattern and composition of herbivorous fish, especially in small outer islands.

2. Materials and Methods

2.1. Study site

Measurements of the herbivorous fish community were conducted in two locations of Small Outer Island in the Eastern Indonesia Region (Liki Island) and Western Indonesia Region (Natuna Island) (Figure 1). Both islands have six sites points for sampling and enumerated herbivorous fish (Table 1) and distributed evenly across the Island to represent the overall condition of both islands. Natuna and Liki have inhabited islands with a higher population intensity and human activities (including fishing, tourism, and boat anchoring) occurring on Natuna.

Liki Island is the outermost island of Indonesia, located north of the island of Papua, and faced the Pacific Ocean. Liki Island is also
the easternmost island of the Republic of Indonesia and bordered by the country of Papua New Guinea. Most of Liki local community activities are fishermen and selling marine resources, including tuna, red snapper, and mackerel, and sold directly to the market in Sarmi Regency. The local community in Liki island has local wisdom in protecting the local marine ecosystem and resources called "Sasi" which is temporarily closed for all fishing activities in a specific area to aim to condition and protecting marine resources in Liki Island. The main goal of Sasi is to give time for marine biota to grow and raise. The condition of marine ecosystems on Liki Island consisted of coral reefs with good condition and had Live coral cover reaching 60%. The growth form mostly branching, *Porites* spp., *Pocillopora* spp., and *Acropora* spp. were the dominant species and followed by *Acropora* tabulate, *Pavona* spp., *Lobophyllia* spp., and also some sponges, and the coral on Liki Island can be found up to a depth of 20 meters.

Natuna is one of the outer islands in Indonesia and faces the Natuna Sea. Natuna Regency, Riau Islands Province, is one area that has a high marine resource potential, with live coral cover reaching 24.53% and included in the medium coral cover category. The growth form of coral mostly massive, encrusting, and branching. *Porites lutea*, *Porites cylindrical*, *Porites rus* very common species. The coral still found to a depth of 10 m. Most of the local people in Natuna are fishermen, and their lives depend on marine resources. The northern of the Natuna Islands had overexploitation fishing activities. Based on information from the local government, most areas in Natuna,
experienced destructive fishing activities, including bombing and potassium poison.

2.2. Survey Methods

To quantify herbivorous fishes community structure (diversity, density, and length of body size) was recorded using SCUBA diving equipment with Underwater Visual Census (UVC). The UVC method is the most common method for non-destructively surveying community dynamics (Davis & Smith, 2017; Figueroa-Pico, Carpio & Tortosa, 2020), superior to collection techniques for the marine organism, especially in sensitive areas (Pinault,

Table 2. List of Herbivore Fishes Species and density (mean ± standard deviation) individuals/350m² in Liki Island and Natuna Island

| Herbivore Fish Species          | IUCN | Liki (n_sites = 6) | Natuna (n_sites = 6) |
|---------------------------------|------|-------------------|---------------------|
| **Family: Acanthuridae**        |      |                   |                     |
| Acanthurus albifrons           | LC   | 7.50 ± 7.11       | 0.00 ± 0.00         |
| Acanthurus auranticus          | LC   | 8.17 ± 1.08       | 0.83 ± 0.83         |
| Acanthurus blochii             | LC   | 5.00 ± 1.71       | 0.00 ± 0.00         |
| Acanthurus fowleri             | LC   | 0.67 ± 0.67       | 0.00 ± 0.00         |
| Acanthurus grammoptilus        | LC   | 1.50 ± 1.50       | 1.00 ± 0.68         |
| Acanthurus leucocheilus        | LC   | 4.33 ± 4.33       | 0.00 ± 0.00         |
| Acanthurus lineatus            | LC   | 5.17 ± 0.98       | 5.00 ± 2.76         |
| Acanthurus maculiceps          | LC   | 23.33 ± 13.61     | 0.00 ± 0.00         |
| Acanthurus mata                | LC   | 6.33 ± 2.44       | 0.00 ± 0.00         |
| Acanthurus nigricans           | LC   | 9.17 ± 1.11       | 0.00 ± 0.00         |
| Acanthurus nigriacauda         | LC   | 2.50 ± 1.12       | 0.00 ± 0.00         |
| Acanthurus nigrofuscus         | LC   | 1.33 ± 1.13       | 0.00 ± 0.00         |
| Acanthurus ruber               | LC   | 5.50 ± 5.50       | 0.00 ± 0.00         |
| Acanthurus olivaceus           | LC   | 5.83 ± 3.54       | 0.00 ± 0.00         |
| Acanthurus pyroferus           | LC   | 12.00 ± 2.59      | 0.00 ± 0.00         |
| Acanthurus thompsoni           | LC   | 12.67 ± 3.96      | 0.00 ± 0.00         |
| Acanthurus triostegus          | LC   | 0.00 ± 0.00       | 1.33 ± 0.88         |
| Acanthurus xanthopterus        | LC   | 15.67 ± 11.55     | 0.00 ± 0.00         |
| Ctenochaetus binotatus         | LC   | 13.67 ± 1.09      | 0.00 ± 0.00         |
| C. cyanochilus                 | LC   | 7.33 ± 1.36       | 0.00 ± 0.00         |
| C. striatus                    | LC   | 41.00 ± 1.72      | 1.17 ± 1.17         |
| C. tominiensis                 | LC   | 5.50 ± 2.70       | 0.00 ± 0.00         |
| Naso brachysilus               | LC   | 3.33 ± 2.17       | 0.33 ± 0.33         |
| N. brevirostris                | LC   | 2.00 ± 1.37       | 0.00 ± 0.00         |
| N. hexacanthus                 | LC   | 18.67 ± 6.34      | 0.00 ± 0.00         |
| N. lituratus                   | LC   | 3.50 ± 0.96       | 2.00 ± 0.93         |
| N. lopesi                      | LC   | 7.17 ± 4.09       | 0.00 ± 0.00         |
| N. thynnoides                  | LC   | 1.33 ± 1.33       | 0.00 ± 0.00         |
| N. viamigui                    | LC   | 10.17 ± 4.61      | 0.00 ± 0.00         |
| Paracanthurus hepatus          | LC   | 4.00 ± 1.91       | 0.00 ± 0.00         |
| Zebrasoma scopas               | LC   | 17.33 ± 1.56      | 1.00 ± 1.00         |
| Zebrafish spilus               | LC   | 0.83 ± 0.54       | 1.83 ± 1.33         |
| **Family: Scridae**            |      |                   |                     |
| Bolbometopon muricatum         | LC   | 0.50 ± 0.34       | 1.67 ± 1.67         |
| Calotomus carolinus            | LC   | 0.17 ± 0.17       | 1.00 ± 1.00         |
| Cetoscarus ocellatus           | LC   | 1.50 ± 0.67       | 0.33 ± 0.33         |
| Chlorurus bleekeri             | LC   | 5.50 ± 0.56       | 7.17 ± 1.22         |
| C. bowersi                     | NT   | 0.17 ± 0.17       | 3.33 ± 0.76         |
| Chlorurus capistratioides      | LC   | 0.17 ± 0.17       | 2.67 ± 1.45         |
| Chlorurus frontalis            | LC   | 0.50 ± 0.50       | 0.50 ± 0.00         |
| C. japonensis                  | LC   | 2.67 ± 1.36       | 0.00 ± 0.00         |
| C. micrornis                   | LC   | 0.83 ± 0.31       | 2.50 ± 0.96         |
| C. sordidus                    | LC   | 14.17 ± 3.48      | 30.00 ± 8.52        |
| C. splilurus                   | LC   | 0.00 ± 0.00       | 2.00 ± 1.44         |
| Hipposcarus longiceps          | LC   | 0.83 ± 0.83       | 0.50 ± 0.50         |
| Leptoscarus vaigiensis         | LC   | 0.00 ± 0.00       | 0.00 ± 0.00         |
Bissery, Gassiole, et al., 2014), and the method was sufficient to long term monitoring program (Kulbicki, Cornuet, Vigliola, et al., 2010). The UVC method is the most common technique used in fishes census (Ricart, Sanmartí, Pérez, et al., 2018). Most of the fish researcher was applied this method to records reef fishes, invertebrates and benthic substrate (Dwirama Putra, Suryanti, Kurniawan, et al., 2018; Figueroa-Pico et al., 2020; Jones, Davidson, Gardner, et al., 2015; Pereira, Moraes, dos Santos, et al., 2014; Robinson, Baum & Giacomini, 2016; Rongo & van Woeskik, 2013). The study uses UVC methods to record each observed herbivore fish species along modification line transects based on (Lewis & Wainwright, 1985; Rotjan & Lewis, 2006; Wen, Chen, Hsieh, et al., 2013) with 70 m long at 5 - 10 m depth and the observer within 5 m on either side of the transect was identified herbivore fishes.

The UVC performed in diurnal times due to the increase in temperature during diurnal due to heating causes an increase in dissolved oxygen from photosynthesis results on the reef, where these environmental variables in diurnal conditions correlate with eating patterns and abundance of herbivorous fish; thus it can maximally to observe herbivorous fish in the reef area (Polunin & Klumpp, 1989). Three families of major herbivore fish identified by underwater visual census methods (Acanthuridae, Scaridae and Siganidae) (Chabanet, Bigot, Nicet, et al., 2015) with 70 m long at 5 - 38 m, 2018; Robinson, Baum & Giacomini, 2016; Lewis & Wainwright, 1985; Williams, 1991).

### 2.3. Data Analysis

Three main herbivorous fishes families (Acanthuridae, Scaridae, and Siganidae) were identified by Underwater Visual Census (UVC) methods were registered according to the lowest species of the taxon. The study focuses on herbivorous fish composition with diversity (number of species), abundance (number of individuals) and biomass per transect, was calculated using the length-weight relationship by (Chabanet et al., 2016; Putra, Afatta, Wilson, et al., 2015) with equation $W = aLT^b$. Where $W$ is the weight (g), $LT$ the fork length of herbivore
fish (cm), and \(a\) and \(b\) are coefficients specific to each species from FishBase data (Froese and Pauly, 2003). The total biomass for each station corresponds to the total weight of all fish per unit area (g/m²). The research concentrates on the main herbivorous fishes by analyzing mean density with standard error (se) herbivore fishes (Acanthuridae, Scaridae, and Siganidae) in Liki Island and Natuna Island. The nominally of size-frequency distribution analyses using “ggstatsplot” packages (Patil, 2018) implemented in the R Statistical Software Analysis. Subsequently, a non-parametric Kruskal-Wallis was performed also using “ggstatsplot” packages (Patil, 2018) to compare density from three herbivorous families (Acanthuridae, Scaridae, and Siganidae). For a more comprehensive analysis, the mean density of herbivore fishes was compared in every herbivore family in each location with a pairwise Wilcoxon test with a significant difference, p-value < 0.05 (Booth & Beretta, 2002). The relationship between density and biomass in each herbivorous fishes families was analyzed using a biplot density-biomass distribution regression linear analysis with dominant species, where the axis variable (\(x\)) is herbivore fishes density and ordinate (\(y\)) is the herbivore fishes biomass.

3. Results and Discussion

3.1. Size frequency distribution

The standard length-frequency distributions of these individuals were summarized in Figure 2. The total herbivorous fishes counted was 1808 individuals in Liki Island (Eastern Indonesia) from 3 reef fishes families, including Acanthuridae (\(n = 1571\)), Scaridae (\(n = 148\)), and Siganidae (\(n = 89\)).

Table 3. Table showing parameter values for Turbidity in Liki Island and Natuna Island

| Turbidity (NTU) in six sites location sampling |
|----------------------------------------------|
| Liki Island (eastern)                        |
| 0.25                                         |
| 0.16                                         |
| 0.29                                         |
| 0.22                                         |
| 0.24                                         |
| 0.15                                         |
| Natuna Island (western)                      |
| 0.9                                          |
| 0.44                                         |
| 0.32                                         |
| 0.31                                         |
| 0.54                                         |
| 0.32                                         |
Acanthuridae fishes in Liki Islands ranged to 60.0 cm, with a mean standard length of 19.63 cm \((t = -38.02 ; g_{\text{hedge}} = -0.96, CI_{95\%} [-1.02, -0.90])\), Scaridae ranged to 80 cm, with a mean standard length of 21.41 cm \((t = -9.70 ; g_{\text{hedge}} = -0.79, CI_{95\%} [-0.98, -0.61])\), and Siganidae ranged from 20 cm to 40.0 cm \((t = -0.18 ; g_{\text{hedge}} = -0.02, CI_{95\%} [-0.23, -0.19])\). There were some differences in the length–frequency distributions for Natuna Islands (Western Indonesia) herbivorous fishes composition counted with total of 1043 individuals of herbivorous fishes from 3 reef fishes families, including Acanthuridae \((n = 87)\), Scaridae \((n = 846)\), and Siganidae \((n = 110)\). Acanthuridae fishes in Natuna Islands ranged to 40.0 cm, with a mean standard length of 18.23 cm \((t = -13.49 ; g_{\text{hedge}} = -1.43, CI_{95\%} [-1.74, -1.14])\), Scaridae ranged to 40 cm, with a mean standard length of 14.71 cm \((t = -54.95 ; g_{\text{hedge}} = -1.89, CI_{95\%} [-2.00, -1.78])\), and Siganidae ranged to 35.0 cm \((t = -31.26 ; g_{\text{hedge}} = -2.96, CI_{95\%} [-3.41, -2.54])\). The highest number of individuals fishes per family was obtained from the Acanthuridae family with the total number of individuals fishes of both locations was \((n = 1658)\) individuals, followed by Scaridae family \((n = 994)\) individuals, and Siganidae family \((n = 199)\) individuals. There was a clear trend in herbivorous fishes population in small outer islands for Acanthuridae families more abundant in Liki island, while the Scaridae family was more dominant in Natuna Islands. From both location, the Siganidae families had the lowest number of individual and smallest mean body size.

Our result showed the size-frequency distributions of nominally in each herbivorous fishes family was dominant in the eastern region (Liki Islands) than the western region (Natuna Island). Even the population of Acanthuridae families in Liki more sufficient ten times than the Natuna (Figure 2). The size distribution pattern also is shown herbivorous fishes in Liki bigger than Natuna, and it indicates the coral ecosystem in Natuna Island had more pressure than the Liki Island. In addition, the presence of the Scaridae family in the high population in Natuna has a positive impact on the coral ecosystem for the recovery process (Figure 2). Coral reefs rely on a range of ecological functions to maintain their resistance to disturbance. Many factors can disrupt the composition of coral reefs, and herbivorous fish, including several significant factors, are sedimentation, coral bleaching, nutrification, and destructive fishing. Apart from the factors that construct the composition of herbivorous fish, the level of disturbance suffered by a certain region in terms of herbivorous grazing activity can also affect the algal succession pathway in coral ecosystems (Fox & Bellwood, 2007). The continuous

**Figure 4.** Comparison mean density of herbivorous fishes families in each family of herbivorous fishes with Wilcoxon test.
disruption will lose the resilience of coral reef ecosystems and cause obstruction of coral regrowth and recolonization; consequently, the condition of coral reefs has shifted from a state of coral domination to algal dominance (Nystrom, Folke & Moberg, 2000). The result

Figure 5. The relationship between density and biomass in each herbivorous fishes families (Acanthuridae, Scaridae, and Siganidae) with dominant appearance species in Eastern SOI Indonesia (Liki) and Western SOI Indonesia (Natuna)
from rapid process changes in coral-algal interactions also depends on specific coral and algal taxa, and several principle factors such as herbivorous fish composition, habitat complexity, water quality, and type of disturbance (Jompa & McCook, 2002). Several coral reef ecologists believe, depending on the level of herbivores and other disturbances to algae, nutrification reefs will be dominated by reef algae that will eventually grow too fast and kill corals (McClanahan, 2002). However, research from (Graham, Wilson, Jennings, et al., 2007) shows that when the disturbance is extensive and occurs on a large spatial scale, it can lead to an increase in the number of large herbivores with a faster growth rate. Eventually, herbivorous fish are the primary ecological process on coral reefs that support complex food webs and significantly contribute to the resilience of this system, particularly their had ability to reorganize and maintain ecosystem functions after disturbance (Vergès, Bennett & Bellwood, 2012).

Another factor causing the herbivorous fish composition is due to the influence of the bleaching phenomenon that occurred in Natuna during 2016 thermal heat stress. Based on studies from (Putra, Suhana, Kurniawna, et al., 2019), the majority reef location in Riau Island in 2016 had an impact on bleaching coral, including Natuna Island during the extreme events of El-Nino Southern-Oscillation (ENSO). Several studies have shown bleaching event result of gradual increases in stress due to loss of herbivory and increases in nutrient levels and cause the coral-algal phase-shift phenomenon (McManus & Polsenberg, 2004). Bleaching events decrease not only live coral cover but also provide large areas for macroalgal colonization to prevent corals from recovery and restoring if herbivores fishes are not present in sufficient numbers to suppress macroalgal colonization and growth (Burkepile & Hay, 2018). Finally, the numbers of herbivorous fishes positively associated with the increasing algal cover that replaced corals in consequence of bleaching events (Ruppert et al., 2013; Sheppard, Spalding, Bradshaw, et al., 2002; Wismer, Tebbett, Streit, et al., 2019).

3.2. The density of Herbivorous Fishes families

A total of 77 species (3520 individuals) of herbivore fishes was recorded in Liki Island (2361 individuals) and Natuna Island (1159 individuals). Ctenochaetus striatus was the highest density of herbivore fish in the Liki Island with mean density (41.00 ± 11.72 se) individuals/350m² where Scarus rivulatus was the highest density of herbivore fish in Natuna Island (31.67 ± 10.61 se) individuals/350m² (Table 2). Based on herbivore fishes categories, the most dominant herbivore fishes for Acanthuridae family was Ctenochaetus striatus, while another dominant species in Liki Island for Scaridae and Siganidae were Scarus niger (19.33 ± 3.02 se) individuals/350m² and Siganus canaliculatus (5.00 ± 3.20 se). The different condition herbivorous fishes density based families group shown in Natuna Island, the dominant Acanthuridae species in Natuna Island was A. lineatus (5.00 ± 2.76 se), where for Scaridae and Siganidae family were Scarus rivulatus (31.67 ± 10.61 se) and Siganus vulpinus (3.17 ± 1.17 se).

The highest individuals of herbivore fishes was recorded in Liki Island from Acanthuridae family, with mean density per sites location (n = 6) was (μ = 268.17 individuals/350 m², CI95% [114.80, 421.53]), followed by Scaridae family (μ = 115.83 individuals/350 m², CI95% [13.72, 245.38]), and Siganidae family (μ = 99.00 individuals/350 m², CI95% [-18.53, 216.53]). There was no significant differences were recorded between the herbivorous fishes families in Liki Islands (Kruskal-Wallis = 5.12, p-value = 0.777, CI95% [0.09, 0.78], nobs = 18) (Figure 3a). There was different pattern of herbivorous fishes families in Natuna Islands. The highest individuals of herbivorous fishes was recorded in Natuna Islands from Scaridae family, with mean density per sites location (n=6) was (μ = 141.33 individuals/350 m², CI95% [82.06, 200.60]), followed by Siganidae family (μ = 18.67 individuals/350 m², CI95% [4.91, 32.42]), and Acanthuridae (μ = 14.5 individuals/350 m², CI95% [-0.23, 29.23]). The significant differences were recorded only Scaridae family in Natuna Island (Kruskal-Wallis = 11.87, p-value = 0.003, CI95% [0.32, 0.85], nobs = 18) (Figure 3b).

Based on several water quality parameters data from our research, including temperature, salinity, and pH did not provide a significant difference for Liki and Natuna Island. Our research shows that the water turbidity between Liki and Natuna provides a high interval difference in each research sites location (Table 3). The differences in water turbidity intervals indicate a high sedimentation process and affect herbivorous fish populations on Natuna Island, especially in the Acanthuridae family. The low Acanthuridae population in the Natuna Islands due to its location has higher sedimentation than Liki Island (Figure 3; Table
Individuals/350 m². The mean density of the Scaridae family in Liki Island was lower than Natuna Island. The mean density of the Acanthuridae family in Liki Island was higher than Natuna Islands (μLiki Islands = 14.5 individuals/350 m²) in the comparison location in Liki island (μ = 99.00 individuals/350 m²) and Natuna Islands (μ = 18.67 individuals/350 m²) in the non-significant difference of number individual herbivorous fishes with Wilcoxon rank test (p-value = 0.26). The relatively uncommon pattern of Siganidae families was shown in the comparison location in Liki island (μ = 99.00 individuals/350 m²) and Natuna Islands (μ = 141.33 individuals/350 m²) shows the non-significant difference in the number of individual herbivorous fishes with Wilcoxon rank test (p-value = 1.00). The Acanthuridae family has shown 42.99% density composition of herbivorous fishes, followed by Scaridae (39.11%), and Siganidae (17.90%).

The increasing population and the presence of migrants to Natuna have a negative influence on coral reef ecosystems. Intensive development in coastal areas and the increasing destructive behavior of local communities have caused the coral reef ecosystem in Natuna to be depressed. The increased land activities and destructive fishing behavior will reduce coral reef resilience with exacerbate the increase in algal abundance (Burkepile, Allgeier, Shantz, et al., 2013; Burkepile & Hay, 2008) and elevate nutrients concentrations via sewage (Littler, Littler & Brooks, 2006). All of these factors indicate that nutrient enrichment can influence coral reef community development and reduced herbivorous fishes (Harvey, Nash, Blanchard, et al., 2018; Roth, Stuhldreier, Sánchez-Noguera, et al., 2015). Previous studies from (Jayewardene, 2009) highlight the main factor of promoting the proliferation of algae on coral reefs, including increased nutrient availability due to eutrophication, and reduced grazing resulting from fishery depletion of herbivorous fishes, so the increasing nutrient concentration could be decreased with reduced destructive fishing activities (Burkepile et al., 2013). It is strongly advocated that the maintenance or restoration of herbivore populations within reef areas should be part of management, in addition to controlling nutrient pollution (Dubinsky, 2013).

Natuna has a very high fishing activity and includes overexploitation both from the local community and foreign vessels that come to Natuna. In addition, several destructive fishing activities found in Natuna, information from the local government in Natuna, and some of the local fishers still use bombing and poisons in fishing practices. Different conditions from Natuna, Liki apply the Sasi system in fishing
practice, where the system serves to protect the diversity and population of marine organisms in Liki. Our results indicate that locations with are high fishing grounds are more likely to have lower herbivorous fish compositions (Figure 4). Previous studies shown high fishing impacts strongly decrease herbivory performance on a coral reef (Kupec & Reuter, 2016) and reduce on the abundance, biomass and community composition of these fishes (Edwards et al., 2013). The studies from (McClanahan, 2014) also shown the fishing impact influence the larger diversity of herbivores fishes, especially in over-fishing practice in the coral region. Overfishing occurs on virtually every reef, including in small outer islands and impact to larger economic fishes group (grouped, snapper, sweetlips), even herbivorous fish are much reduced (Burke, Reytar, Spalding, et al., 2011). The historical shown overfishing practice not only reduces important economic fishes, but several piscivores, corallivorous, endangered species, and herbivorous fishes decrease significantly (Hopley, 2011). In recent decades roving herbivorous fishes have been identified as critical elements of coral reef communities, and overfishing of these fishes is considered a significant factor contributing to global reef degradation (Vergés et al., 2012).

3.3. The relationship between density and biomass

Both of research location (Liki and Natuna) showed the relationship biomass, and density was positively linear correlated in each family herbivorous fish (Acanthuridae, Scaridae, and Siganidae). The growth in density and biomass of herbivorous fish shows a positive balanced proportion, which indicates herbivore fishes can still grow in the coral reef ecosystem of both islands. If the correlation between growth in density and biomass herbivorous fishes shown the constant linear trend or towards a negatively linear relationship, this indicates extremely high pressure on the herbivorous fish group impartially. Although the relationship of density and biomass of herbivorous fish shows a positive linear relationship, the comprehensive analyzed show the density and biomass distribution of herbivorous fish for each family (Acanthuridae, Scaridae, and Siganidae) on Natuna Island is lower than in Liki Island which biomass can reach two times bigger for the Scaridae fish family group and six times greater for the Siganidae group in Liki Island, and this indicates that there is pressure on the coral reef ecosystem on Natuna Island.

From the result, the correlation from distribution grand mean biomass-density of the Acanthuridae family in the Liki Island was ($\mu_b = 4662.96$ gram/350 m$^2$ ; $\mu_d = 13.41$ individuals/350 m$^3$) for each site’s location with several dominant of species including Naso hexacanthus, N. brachycenetrn, N. lopezi, and Acanthurus xanthopterus. The correlation distribution from grand mean biomass-density of Scaridae family in the Liki Island for the six sites location was lower than Acanthuridae family with ($\mu_b = 2263.19$ gram/350 m$^2$ ; $\mu_d = 6.67$ individuals/350 m$^3$) for each sites location and Bolbometopon muricatum was dominant species from Scaridae family in Liki Island. The lowest correlation distribution from grand mean biomass-density in Liki Island from Siganidae family with ($\mu_b = 2263.19$ gram/350 m$^2$ ; $\mu_d = 6.67$ individuals/350 m$^3$) for each sites location.

Natuna Islands had different patterns of distribution grand mean biomass-density compare with Liki Island. Scaridae was the highest grand mean biomass-density in the Liki Island ($\mu_b = 1191.14$ gram/350 m$^2$ ; $\mu_d = 9.98$ individuals/350 m$^3$) for each site’s location with several dominant species, including B. muricatum, Chlorurus microrhinos, C. bleekeri, C. sordidus, Scarus quoyi, S. niger, and S. rubroviolaceus. The correlation from distribution grand mean biomass-density of Acanthuridae family in the Natuna Islands was lower compared to Scaridae with ($\mu_b = 945.50$ gram/350 m$^2$ ; $\mu_d = 5.44$ individuals/350 m$^3$) and only dominant species was N. brachycentron. From 3 herbivorous family fishes, the Siganidae has shown the lowest grand mean biomass-density in the Natuna Islands ($\mu_b = 207.93$ gram/350 m$^2$ ; $\mu_d = 5.60$ individuals/350 m$^3$) with several dominant species, including Siganus punctatus, S. virgatus, and S. vulpinus.

Although from our studies, we found the pressure to herbivorous fishes family in Natuna island, the lack of monitoring and controlling in the long-term period in the small outer island will increase the overfishing and destructive fishing activities, especially in reducing herbivore fishes. The impact of reducing herbivory through overfishing on a reef in the long-term period with high productivity contribute an increase in macro-algae density, and the long term provides persistence to algal dominance (Hughes, Graham, Jackson, et al., 2010; Littler et al., 2006; Marshall & Mumby, 2012; McClanahan, 2002; McManus & Polsenberg, 2004; Teixeira-Neves, Neves &
Araújo, 2016). Another previous research from (Burkepile & Hay, 2018) shown the removed herbivorous fish in the coral ecosystem from overfishing cause the macroalgal to replace the coral and biogenic structure of the reef will degrade. Other evidence that can indicate overexploitation of coral reef ecosystems is the lack of presence of large excavator, including Bolbometopon muricatum and Cetoscarus ocellatus from Scaridae family, due to a large-bodied excavator are usually among the first to disappear on an overfished reef (Grimsditch, Tamelander, Mwaura, et al., 2009). From our result, we found several sites presence of very low large excavators in the Natuna Islands, including B. m. and C. ocellatus from the Scaridae family. The low presence of large excavator herbivorous fishes in Natuna Island was a dominant influence by the benthic community (Dwirama Putra et al., 2018) (Figure 5; Table 4).

Our study shows exciting results in herbivorous fishes assemblages, where the eastern region (Liki Island) is dominated by Acanthuridae while in the western region (Natuna Island) is dominated by Scaridae. Both of fishes families is a part of Roving herbivorous and identified as the key functional group performing the significant ecological role concerning the dynamics of benthic communities (Dromard et al., 2015; Hoey & Bellwood, 2008; Mumby, Dahlgren, Harborne, et al., 2006). Several studies also have shown the main herbivorous fishes on coral reefs are generally surgeonfishes (Acanthuridae), parrotfishes (Scaridae) with rabbitfishes (Siganidae), and several territorial herbivorous fishes from damselfishes (Pomacentridae) also responsible for considerable herbivory in specific locations (Burkepile & Hay, 2018; Dubinsky, 2013; Jompa & McCook, 2002). In several reef sites location, both fishes have a dominant population with the most abundant of herbivorous fishes (Grimsditch et al., 2009; Pereira et al., 2014).

As a part of roving herbivorous fishes, Scaridae and Acanthuridae have special functions in the coral reef ecosystem, the surgeonfishes (Acanthuridae) has dominant play a role in keeping EAM (Marshall & Mumby, 2012), while Parrotfishes (Scaridae) are essential as grazers and bioeroders on coral reefs (Bellwood, Hoey & Choat, 2003; Comerros-Raynal, Choat, Polidoro, et al., 2012; Rice, Ezzat & Burkepile, 2019). Based on previous studies from (Fox & Bellwood, 2007; Marshall & Mumby, 2012) shown the surgeonfishes represented 74% of the herbivorous fish biomass in feed Epilithic algal matrix (EAM) and responsibility removed 73% of daily EAM productivity in the shallow zone. The differences and the existence of populations of these two fish can explain the condition and status of coral reef ecosystems. Previous studies shown the population from roving herbivores are more abundant in unprotected reef sites. In contrast, the piscivores, carnvoires, mobile invertebrate feeders, and territorial herbivores all were significantly more abundant in protected sites (Floeter, Halpern & Ferreira, 2006). The differences in population reef fishes caused by the characteristics of some types of fish in response to anthropogenic pressure.

The biggest challenge at SOI is to protect the area from anthropogenic pressure, including overexploitation and destructive fishing practice. High over exploitation will reduce rapidly of carnivorous fishes as important economic fishes, so the only a small population is found in the unprotected area, whereas if carnivorous fish populations increase (e.g., grouper, snapper, sweetlips) resulted in increased predation rates on small herbivorous as parrotfishes (Burkepile & Hay, 2018) and cause the herbivorous population fishes is low. The opposite even will also happen if following the decline in cover of live coral, the herbivore assemblage on the forereef became increasingly dominated by parrotfish (Adam et al., 2011). After the larger carnivorous animals are depleted, fisheries switch to smaller predators such as groupers and then increase the population of herbivorous fish such as parrotfishes (Burkepile & Hay, 2018). However, the presence of herbivorous fish has indirect positive effects on corals. (Rotjan & Lewis, 2006). The main problem is that the capacity of herbivorous fish that plays a role as coral reef resilience is also limited if anthropogenic pressure or natural disturbance is increasing. Several small outer islands or oceanic islands as Most Caribbean reefs today are also algal-dominated, because of overfishing of herbivorous fishes and the continuing low densities (Hughes et al., 2010). This has become a global problem because reductions in herbivore diversity and abundance via

Table 4. The Percentage of Hard Coral and Dead Coral Algae in Natuna Island

|        | NTNC.01 | NTNC.02 | NTNC.03 | NTNC.04 | NTNC.05 | NTNC.06 |
|--------|---------|---------|---------|---------|---------|---------|
| HC (%) | 36.87   | 33.27   | 14.33   | 30.93   | 23.73   | 19.73   |
| DCA (%)| 58.93   | 48.87   | 48.07   | 56.47   | 51.27   | 63.53   |
overfishing may harm corals directly and may indirectly increase coral susceptibility to other disturbances (Burkepile & Hay, 2010; Hughes et al., 2007).

For herbivorous fishes, besides the several standard protocols has used to measures density, richness, and diversity per family, the measurements of size an essential consideration for (Leão et al., 2019) to determine if the biomass as ecological functions condition. The measurement of biomass, diversity, and abundance of herbivorous fishes that perform these critical functions is becoming a key role in the evaluation of coral reef resilience (Goatley et al., 2016). Large populations of herbivorous fish that have high biomass indicate an ecosystem that protects herbivorous fish. In addition, the ecosystem protection increase size of herbivorous fishes, and the effect of several species of megaherbivorous fishes as B. murex and C. ocellatus can increase growth in maximum size (Johnson, 2009). Our result has shown consistency with previous research from (Beita-Jiménez, A., Alvarado, J.J., Mena, S., et al., 2019) with the biomass of predators, carnivores, and herbivores were greater in areas with protection. Another consistency of herbivorous pattern in our result shown in the protection area has a significantly higher coral cover, and less fleshy algae cover compared to fished reefs area (Ford, Eich, McAndrews, et al., 2018). Other benefits of several large herbivorous fish species with high biomass allow for more bites per individual fish (Burkepile & Hay, 2018) and responsible for reducing algal biomass (Ceccarelli et al., 2011; Paddock, Cowen & Sponaugle, 2006) for help in resilience reef ecosystems rapidly.

In addition, the presence of large herbivorous fishes attracts some top-level predators and helps in restoring ecosystem balance. The high predation mortality of herbivorous fish will increase predator biomass (Rogers et al., 2018). Previous studies from (Floeter et al., 2006) shown roving herbivores showed an overall negative response to more excellent protection, where fish species from higher trophic levels (e.g., piscivores) were among the more positive impact in protection area with growth optimally. Our last result consistent with the high abundance and biomass of herbivores in marine ecosystems may indicate a sign of degradation due to the higher biomass of algae (Medeiros & Grempel, 2007). Our result show, all of the family group herbivorous fishes with small to medium biomass and the mean of length of herbivorous fish under 30 cm of body size. Previous studies shown on Pacific reefs and several reef sites in Great Barrier Reefs, herbivore biomass is dominated by smaller-bodied fishes including acanthurids and smaller parrotfishes (Bellwood et al., 2004; Rogers et al., 2018; Russ, 2003); thus, the presence of several sizeable herbivorous fish species signifies a healthy coral reef ecosystem. Studies from (Wilson, Graham, Fisher, et al., 2012) confirm the good coral ecosystem in the protected area had higher biomass of herbivorous than fishes area.

4. Conclusion

Herbivorous fish biomass patterns are very helpful in understanding the condition of coral reef ecosystems comprehensively. There is a unique pattern of herbivorous fish distribution, where Acanthuridae is dominant in eastern Indonesia (Liki Islands) while Scaridae is dominant in western Indonesia (Natuna Island). The eastern part of Indonesia has a higher herbivorous fish composition, both in density and biomass, compared to the western part of Indonesia. Our result in two locations small outer island Indonesia concludes with the consistency of previous studies shown increased herbivore richness, density, and biomass strongly reduced the cover, biomass, and diversity of fleshy algae (Burkepile & Hay, 2008; Cowburn, Samoilys, Osaka, et al., 2019; Newman, Paredes, Sala, et al., 2006; Sandin, Sampayo & Vermeij, 2008).

Acknowledgment

The authors are deeply grateful to the Research Centre for Oceanography, Indonesia Institute of Science, as organizing committee in Nusa Manggala Expedition. The authors gratefully acknowledge support for this research fund from COREMAP-CTI Program in 2018. Finally, we thank Dr. Ani Suryanti and two anonymous reviewers for their helpful comments that have substantially improved our manuscript.

References

Adam, T.C., Schmitt, R.J., Holbrook, S.J., Brooks, A.J., Edmunds, P.J., Carpenter, R.C., Bernardi, G. 2011. Herbivory, connectivity, and ecosystem resilience: Response of a coral reef to a large-scale perturbation. PLoS ONE 6 (8).

Beita-Jiménez, A., Alvarado, J.J., Mena, S., Guzmán-Mora, A.G. 2019. Benefits of protection on reef fish assemblages in a
human impacted region in Costa Rica. Ocean and Coastal Management 169: 165-170.

Bellwood, D.R., Hoey, A.S., Choat, J.H. 2003. Limited functional redundancy in high diversity systems: Resilience and ecosystem function on coral reefs. Ecology Letters 6 (4): 281-285.

Bellwood, D.R., Hughes, T.P., Folke, C., Nyström, M. 2004. Confronting the coral reef crisis. Nature 429 (6994): 827-33.

Booth, D.J., Beretta, G.A. 2002. Changes in a fish assemblage after a coral bleaching event. Marine Ecology Progress Series. 245: 205-212.

Brown, K.T., Bender-Champ, D., Kubicek, A., van der Zande, R., Achiatis, M., Hoegh-Guldberg, O., Dove, S.G. 2018. The dynamics of coral-algal interactions in space and time on the southern Great Barrier Reef. Frontiers in Marine Science. 5 (MAY): 1-13.

Burkepile, D.E., Hay, M.E. 2008. Herbivore species richness and feeding complementarity affect community structure and function on a coral reef. Proceedings of the National Academy of Sciences. 105 (42): 16201-16206.

Burkepile, D.E., Hay, M.E. 2010. Impact of herbivore identity on algal succession and coral growth on a Caribbean reef. PLoS ONE. 5 (1).

Burkepile, D.E. Hay, M.E. 2018. Coral reefs. Encyclopedia of Ecology. 2: 426-438.

Burkepile, D.E., Allgeier, J.E., Shantz, A.A., Pritchard, C.E., Lemoine, N.P., Bhatti, L.H. Layman, C.A. 2013. Nutrient supply from fishes facilitates macroalgae and suppresses corals in a Caribbean coral reef ecosystem. Scientific Reports. 3: 19-21.

Ceccarelli, D.M., Jones, G.P. McCook, L.J. 2011. Interactions between herbivorous fish guilds and their influence on algal succession on a coastal coral reef. Journal of Experimental Marine Biology and Ecology. 399 (1): 60-67.

Chabanet, P., Bigot, L., Nicet, J.B., Durville, P., Massé, L., Mulochau, T., Russo, C., Tessier, E., et al. 2016. Coral reef monitoring in the Iles Eparses, Mozambique Channel (2011-2013). Acta Oecologica. 72: 62-71.

Clausing, R.J., Annunziata, C., Baker, G., Lee, C., Bittick, S.J. Fong, P. 2014. Effects of sediment depth on algal turf height are mediated by interactions with fish herbivory on a fringing reef. Marine Ecology Progress Series. 517: 121-129.

Comeros-Raynal, M.T., Choat, J.H., Polidoro, B.A., Clements, K.D., Abesamis, R., Craig, M.T., Lazuardi, M.E., McIlwain, J., et al. 2012. The likelihood of extinction of iconic and dominant herbivores and detritivores of coral reefs: The parrotfishes and surgeonfishes. PLoS ONE. 7 (7).

Cowburn, B., Samoilys, M.A., Osuka, K., Klaus, R., Newman, C., Gudka, M. Obura, D. 2019. Healthy and diverse coral reefs in Djibouti - A resilient reef system or few anthropogenic threats? Marine Pollution Bulletin. 148: 182-193.

Davis, T.R. Smith, S.D.A. 2017. Proximity effects of natural and artificial reef walls on fish assemblages. Regional Studies in Marine Science. 9: 17-23.

Dubinsky, Z. 2013. Earth System Monitoring. R.A. Meyers (ed.). New York: Springer.

Dwirama Putra, R., Suryanti, A., Kurniawan, D., Pratomo, A., Irawan, H., Said Raja’l, T., Kurniawan, R., Pratama, G., et al. 2018. Responses of herbivorous fishes on coral reef cover in outer island Indonesia (Study Case: Natuna Island). E3S Web of Conferences. 47.

Edinger, E.N., Jompa, J., Limmon, G. V., Widjatmoko, W. Risk, M.J. 1998. Reef degradation and coral biodiversity in Indonesia: Effects of land-based pollution, destructive fishing practices and changes
over time. Marine Pollution Bulletin. 36 (8): 617-630.

Edwards, C.B., Friedlander, A.M., Green, A.G., Hardt, M.J., Sala, E., Sweatman, H.P., Williams, I.D., Zgliczynski, B., et al. 2013. Global assessment of the status of coral reef herbivorous fishes: Evidence for fishing effects. Proceedings of the Royal Society B: Biological Sciences. 281 (1774): 7-11.

Fabricius, K., De'ath, G., McCook, L., Turak, E., Williams, D.M.B. 2005. Changes in algal, coral and fish assemblages along water quality gradients on the inshore Great Barrier Reef. Marine Pollution Bulletin. 51 (1-4): 384-398.

Ferrigno, F., Bianchi, C.N., Lasagna, R., Morri, C., Russo, G.F., Sandulli, R. 2016. Corals in high diversity reefs resist human impact. Ecological Indicators. 70: 106-113.

Figueroa-Pico, J., Carpio, A.J., Tortosa, F.S. 2020. Turbidity: A key factor in the estimation of fish species richness and abundance in the rocky reefs of Ecuador. Ecological Indicators. 111: 106021.

Floeter, S.R., Halpern, B.S., Ferreira, C.E.L. 2006. Effects of fishing and protection on Brazilian reef fishes. Biological Conservation. 128 (3): 391-402.

Fong, C.R., Frias, M., Goody, N., Bittick, S.J., Clauing, R.J., Fong, P. 2018. Empirical data demonstrates risk-tradeoffs between landscapes for herbivorous fish may promote reef resilience. Marine Environmental Research. 133: 1-5.

Ford, A.K., Eich, A., McAndrews, R.S., Mangubhai, S., Nugues, M.M., Bejarano, S., Moore, B.R., Rico, C., et al. 2018. Evaluation of coral reef management effectiveness using conventional versus resilience-based metrics. Ecological Indicators. 85 (September 2017): 308-317.

Fox, R.J. Bellwood, D.R. 2007. Quantifying herbivory across a coral reef depth gradient. Marine Ecology Progress Series. 339: 49-59.

Goatley, C.H.R., Bellwood, D.R. 2012. Sediment suppresses herbivory across a coral reef depth gradient. Biology Letters. 8 (6): 1016-1018.

Goatley, C.H.R., Bonaldo, R.M., Fox, R.J. Bellwood, D.R. 2016. Sediments and herbivory as sensitive indicators of coral reef degradation. Ecology and Society. 21 (1): 1-17.

Graham, N.A.J., Wilson, S.K., Jennings, S., Polunin, N.V.C., Robinson, J., Bijoux, J.P., Daw, T.M. 2007. Lag effects in the impacts of mass coral bleaching on coral reef fish, fisheries, and ecosystems. Conservation Biology. 21 (5): 1291-1300.

Grimsditch, G., Tamelander, J., Mwaura, J., Zavagli, M., Takata, Y., Gomez, T. 2009. Coral Reef Resilience Assessment of the Pemba Channel Conservation Area, Tanzania.

Guest, J.R., Vergés, A., Bauman, A.G., Campbell, A.H., Chou, L.M., Feary, D.A., Low, J.K.Y., Marzinelli, E.M., et al. 2016. Examining the relationship between fish herbivore biomass, coral and macroalgal cover on Singapore’s heavily disturbed reefs. PeerJ PrePrints.

Harvey, B.J., Nash, K.L., Blanchard, J.L., Edwards, D.P. 2018. Ecosystem-based management of coral reefs under climate change. Ecology and Evolution. 8 (12): 6354-6368.

Hoey, A.S., Bellwood, D.R. 2008. Cross-shelf variation in the role of parrotfishes on the Great Barrier Reef. Coral Reefs. 27 (1): 37-47.

Hopley, D. 2011. Encyclopedia of Modern Coral Reef: Structure, Form and Process. D. Hopley (ed.). (Encyclopedia of Earth Sciences Series). Dordrecht: Springer Netherlands.

Hughes, T.P., Baird, A.H., Bellwood, D.R., Card, M., Connolly, S.R., Folke, C., Grosberg, R. 2003. Climate Change, Human Impacts, and the Science. 301: 929-934.

Hughes, T.P., Rodrigues, M.J., Bellwood, D.R., Ceccarelli, D., Hoegh-Guldberg, O., McCook, L., Moltschanowski, N., Pratchett, M.S., et al. 2007. Phase Shifts, Herbivory, and the Resilience of Coral Reefs to Climate Change. Current Biology. 17 (4): 360-365.

Hughes, T.P., Graham, N.A.J., Jackson, J.B.C., Mumby, P.J., Steneck, R.S. 2010. Rising to the challenge of sustaining coral reef resilience. Trends in Ecology and Evolution. 25 (11): 633-642.

Jayewardene, D. 2009. A factorial experiment quantifying the influence of parrotfish
density and size on algal reduction on Hawaiian coral reefs. Journal of Experimental Marine Biology and Ecology. 375 (1-2): 64-69.

Johnson, C.N. 2009. Ecological consequences of late quaternary extinctions of megafauna. Proceedings of the Royal Society B: Biological Sciences. 276 (1667): 2509-2519.

Jompa, J. McCook, L.J. 2002. Effects of competition and herbivory on interactions between a hard coral and a brown alga. Journal of Experimental Marine Biology and Ecology. 271 (1): 25-39.

Jones, T., Davidson, R.J., Gardner, J.P.A. Bell, J.J. 2015. Evaluation and optimisation of underwater visual census monitoring for quantifying change in rocky-reef fish abundance. Biological Conservation. 186: 326-336.

Kubicek, A. Reuter, H. 2016. Mechanics of multiple feedbacks in benthic coral reef communities. Ecological Modelling. 329: 29-40.

Kulbicki, M., Cornuet, N., Vigliola, L., Wantiez, L., Mouthing, G. Chabane, P. 2010. Counting coral reef fishes: interaction between fish life-history traits and transect design. Journal of Experimental Marine Biology and Ecology. 387 (1-2): 15-23.

Kunzmann, A. Samuadi, 2017. A century of change in an Indonesian coral reef: Sluiert’s Brandewijnsbaai (1890) revisited. Annual Research and Review in Biology. 13 (3): 1-7.

Leão, Z.M.A.N., Kikuchi, R.K.P. Oliveira, M.D.M. 2019. The Coral Reef Province of Brazil. in World Seas: an Environmental Evaluation Second Ed ed. Vol. 1. Elsevier Ltd. 813-833.

Lewis, S.M. Wainwright, P.C. 1985. Herbivore abundance and grazing intensity on a Caribbean coral reef. Journal of Experimental Marine Biology and Ecology. 87 (3): 215-228.

Littler, M.M., Littler, D.S. Brooks, B.L. 2006. Harmful algae on tropical coral reefs: Bottom-up eutrophication and top-down herbivory. Harmful Algae. 5 (5): 565-585.

Marshall, A. Mumby, P.J. 2012. Revisiting the functional roles of the surgeonfish Acanthurus nigrofuscus and Ctenochaetus striatus. Coral Reefs. 31 (4): 1093-1101.

Martin, C.L., Mottam, S., Jordan, A. Moltchanov, N.A. 2016. Exploring recreational fishers’ perceptions, attitudes, and support towards a multiple-use marine protected area six years after implementation. Marine Policy. 73: 138-145.

McClanahan, T. 2002. Ecological states and the resilience of coral reefs. Nicolas Polunin Terry Done. 6 (2): 1-28.

McClanahan, T.R. 2014. Recovery of functional groups and trophic relationships in tropical fisheries closures. Marine Ecology Progress Series. 497: 13-23.

McLeod, E., Anthony, K.R.N., Mumby, P.J., Maynard, J., Beeden, R., Graham, N.A.J., Heron, S.F., Hoegh-Guldberg, O., et al. 2019. The future of resilience-based management in coral reef ecosystems. Journal of Environmental Management. 233: 291-301.

McManus, J.W. Polsenberg, J.F. 2004. Coral-algal phase shifts on coral reefs: Ecological and environmental aspects. Progress in Oceanography. 60 (2-4): 263-279.

Medeiros, P. Grempel, R. 2007. Effects of recreational activities on the fish assemblage structure in a northeastern Brazilian reef. Pan-American Journal 2: 288-300.

Mumby, P.J., Dahl gren, C.P., Harbome, A.R., Kappel, C. V., Michelli, F., Brumbaugh, D.R., Holmes, K.E., Mendes, J.M., et al. 2006. Fishing, trophic cascades, and the process of grazing on coral reefs. Science. 311 (5757): 98-101.

Newman, M.J.H., Paredes, G.A., Sala, E. Jackson, J.B.C. 2006. Structure of Caribbean coral reef communities across a large gradient of fish biomass. Ecology Letters. 9 (11): 1216-1227.

Nyström, M., Folke, C. Moberg, F. 2000. Coral reef disturbance and resilience in a human-dominated environment. Trends in Ecology Evolution. 15 (10): 413-417.

Paddack, M.J., Cowen, R.K. Sponaugle, S. 2006. Grazing pressure of herbivorous coral reef fishes on low coral-cover reefs. Coral Reefs. 25 (3): 461-472.

Patil, L. 2018. ggstatsplot: “ggplot2” Based Plots with Statistical Details. 11.
Pereira, P.H.C., Moraes, R.L., dos Santos, M.V.B., Lippi, D.L., Feitosa, J.L.L. Pedrosa, M. 2014. The influence of multiple factors upon reef fish abundance and species richness in a tropical coral complex. Ichthyological Research. 61 (4): 375-384.

Pinault, M., Bissery, C., Gassiole, G., Magalon, H., Quod, J.P., Galzin, R. 2014. Fish community structure in relation to environmental variation in coastal volcanic habitats. Journal of Experimental Marine Biology and Ecology. 460: 62-71.

Polunin, N.V.C. Klumpp, D.W. 1989. Ecological correlates of foraging periodicity in herbivorous reef fishes of the Coral Sea. Journal of Experimental Marine Biology and Ecology. 126 (1): 1-20.

Pratchett, M.S., Bridge, T.C.L., Brodie, J., Cameron, D.S., Day, J.C., Emslie, M.J., Grech, A., Hamann, M., et al. 2019. Australia’s Great Barrier Reef. in World Seas: an Environmental Evaluation Second Edi ed. Elsevier Ltd. 333-362.

Putra, M.I.H., Afatta, S., Wilson, J., Muljadi, A. Yusidarta, I. 2015. Coral Reef Resilience in 17 Islands Marine Recreation Park, Riung - An Assessment of Functional Groups of Herbivorous Fish and Benthic Substrate. Procedia Environmental Sciences. 23: 230-239.

Putra, R.D., Suhana, M.P., Kurniawn, D., Abrar, M., Siringoringo, R.M., Sari, N.W.P., Irawan, H., Prayetno, E., et al. 2019. Detection of reef scale thermal stress with Aqua and Terra MODIS satellite for coral bleaching phenomena. AIP Conference Proceedings. 2094.

Quimbayo, J.P., Dias, M.S., Kulbicki, M., Mendes, T.C., Lamb, R.W., Johnson, A.F., Aburto-Oropeza, O., Alvarado, J.J., et al. 2019. Determinants of reef fish assemblages in tropical Oceanic islands. Ecography. 42 (1): 77-87.

Ricart, A.M., Sanmartí, N., Pérez, M. Romero, J. 2018. Multilevel assessments reveal spatially scaled landscape patterns driving coastal fish assemblages. Marine Environmental Research. 140: 210-220.

Rice, M.M., Ezzat, L. Burkepilie, D.E. 2019. Corallivory in the anthropocene: Interactive effects of anthropogenic stressors and corallivory on coral reefs. Frontiers in Marine Science. 5: 1-14.

Robinson, J.P.W., Baum, J.K. Giacomini, H. 2016. Trophic roles determine coral reef fish community size structure. Canadian Journal of Fisheries and Aquatic Sciences. 73 (4): 496-505.

Rogers, A., Blanchard, J.L., Newman, S.P., Dryden, C.S. Mumby, P.J. 2018. High refuge availability on coral reefs increases the vulnerability of reef-associated predators to overexploitation. Ecology. 99 (2): 450-463.

Rongo, T. van Woesik, R. 2013. The effects of natural disturbances, reef state, and herbivorous fish densities on ciguatera poisoning in Rarotonga, southern Cook Islands. Toxicon. 64: 87-95.

Roth, F., Stuhlbreder, I., Sánchez-Noguera, C., Morales-Ramírez, T., Wild, C. 2015. Effects of simulated overfishing on the succession of benthic algae and invertebrates in an upwelling-influenced coral reef of Pacific Costa Rica. Journal of Experimental Marine Biology and Ecology. 468: 55-66.

Roth, F., Saalmann, F., Thomson, T., Coker, D.J., Villalobos, R., Jones, B.H., Wild, C. Carvalho, S. 2018. Coral reef degradation affects the potential for reef recovery after disturbance. Marine Environmental Research. 142: 48-58.

Rotjan, R.D. Lewis, S.M. 2006. Parrotfish abundance and selective corallivory on a Belizean coral reef. Journal of Experimental Marine Biology and Ecology. 335 (2): 292-301.

Ruppert, J.L.W., Travers, M.J., Smith, L.L., Fortin, M.J. Meekan, M.G. 2013. Caught in the Middle: Combined Impacts of Shark Removal and Coral Loss on the Fish Communities of Coral Reefs. PLoS ONE. 8 (9): 1-9.

Russ, G.R. 2003. Grazer biomass correlates more strongly with production than with biomass of algal turfs on a coral reef. Coral Reefs. 22 (1): 63-67.

Sandin, S.A., Sampaio, E.M. Vermeijij, M.J.A. 2008. Coral reef fish and benthic community structure of bonaire and Curaçao, Netherlands Antilles. Caribbean Journal of Science. 44 (2): 137-144.

Sheppard, C.R.C., Spalding, M., Bradshaw, C. Wilson, S. 2002. Erosion vs. recovery of coral reefs after 1998 El Niño: Chagos
Putra et al., 2020, The Pattern of Herbivorous Fish Assemblages

Sheppard, C.R.C.C., Ateweberhan, M., Bowen, B.W., Carr, P., Chen, C.A., Clubbe, C., Craig, M.T., Ebinghaus, R., et al. 2012. Reefs and islands of the Chagos Archipelago, Indian Ocean: why it is the world’s largest no-take marine protected area. Aquatic Conservation: Marine and Freshwater Ecosystems. 22 (2): 232-261.

Souter, D.W. Lindén, O. 2000. The health and future of coral reef systems. Ocean and Coastal Management. 43 (8-9): 657-688.

Teixeira-Neves, T.P., Neves, L.M. Araújo, F.G. 2016. The development of a preliminary rock reef fish multimetric index for assessing thermal and urban impacts in a tropical bay. Marine Pollution Bulletin. 109 (1): 290-300.

Vergés, A., Bennett, S. Bellwood, D.R. 2012. Diversity among Macroalgae-Consuming Fishes on Coral Reefs: A Transcontinental Comparison. PLoS ONE. 7 (9).

Wen, C.K.C., Chen, K.S., Hsieh, H.J., Hsu, C.M. Chen, C.A. 2013. High coral cover and subsequent high fish richness on mature breakwaters in Taiwan. Marine Pollution Bulletin. 72 (1): 55-63.

Wenger, A.S., Fabricius, K.E., Jones, G.P. Brodie, J.E. 2015. Effects of sedimentation, eutrophication, and chemical pollution on coral reef fishes. Ecology of Fishes on Coral Reefs. (May 2017): 145-153.

Wild, C., Hoegh-Guldberg, O., Naumann, M.S., Colombo-Pallotta, M.F., Ateweberhan, M., Fitt, W.K., Iglesias-Prieto, R., Palmer, C., et al. 2011. Climate change impedes scleractinian corals as primary reef ecosystem engineers. Marine and Freshwater Research. 62 (2): 205-215.

Williams, D.M. 1991. Patterns and Processes in the Distribution of Coral Reef Fishes. ACADEMIC PRESS, INC.