ECOLOGICAL STUDIES OF CORAL REEF ECOSYSTEM AT MALVAN, WEST COAST OF INDIA

Thesis submitted to the

Goa University

For the award of the degree of

Doctor of Philosophy

In

Marine Sciences

By

Kalyan De

Under the supervison of

Dr. Baban Ingole

School of Earth, Ocean, and Atmospheric Sciences,

Goa University

(Biological Oceanography Division, CSIR – National Institute of Oceanography, Goa)

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Dedicated to my Parents

and

Gurus

गुरुर्ब्रह्मा गुरुर्विष्णुः गुरुर्देवो महेश्वरः ।

गुरुःसाक्षात् परब्रह्म तस्मै श्रीगुरवे नमः ॥

(आपदशंकराचायव गुरुस्तोत्रम्)

Declaration

As required under the university ordinance OA.19, I hereby state that the present thesis entitled "**Ecological Studies of Coral Reef Ecosystem at Malvan, West Coast of India**" is my original contribution and the same has not been submitted for any other degree, diploma, associate-ship, fellowship or similar titles in any universities or institutions on any previous occasion. To the best of my knowledge, the present study is the first comprehensive work of its kind from the area mentioned. Literature related to the scientific objectives has been cited. Due acknowledgments have been made wherever facilities and suggestions have been availed of.

Place: Goa University, Goa **_Kalyan De_**

Date: 10th August 2020

Certificate

This is to certify that the thesis entitled "**Ecological Studies of Coral Reef Ecosystem at Malvan, West Coast of India**" submitted by Mr. Kalyan De to the School of Earth, Ocean, and Atmospheric Sciences, Goa University for the degree of Doctor of Philosophy in Marine Sciences, is based on original work carried out by him under my supervision. The thesis, or any part thereof has not been previously submitted for any other degree or diploma in any university or institution.

Dr. Baban Ingole
Former Chief Scientist and Head,
Biological Oceanography Division,
CSIR-National Institute of Oceanography,
Dona Paula, Goa-403004

Place: Goa University
Date: 10th August 2020

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Kalyan De

Preface

Coral reefs are one of the most diverse and complex ecosystems in tropical seas, providing sustainable ground to millions of marine biotas to settle, forage, spawn and nurture their juveniles (Moberg & Folke, 1999, Hoegh-Guldberg *et al.,* 2018). The ecosystem services from corals have been valued in the billions of dollars and mainly include nutrient cycling, food supply, shoreline protection, erosion regulation, a source of income by boosting fishery and tourism industries (Hoegh-Guldberg *et al.,* 2018). Coral reefs directly support over 500 million people worldwide, mostly in developing countries (IUCN 2017).

India is blessed with four major coral reefs in the Gulf of Kachchh, Lakshadweep archipelago, Gulf of Mannar, and the Andaman and Nicobar Islands. Apart from these reefs, there are several small reefs are there like Netrani Island in Karnataka coast (Zacharia *et al.,* 2008), Grande Islands in Goa coast (Rodriques,1998), Gavesani bank in off Malpe, Kerala coast (Nair, 1978) and the Malvan Marine Sanctuary (MMS) along the Maharashtra coast. Biodiversity and ecological studies from these reefs, especially the Malvan and Goa, are poorly described.

The MMS is the only Marine Protected Area (MPA) in the Central West Coast of India and harbours coral reef ecosystem. The MMS is spread over 29.12 km^2 with a core zone of 3.182 km^2, and the buffer zone is comprising of an area of 25.94 km^2 between $16^0\ 02'00$ N-$16^0 03'90$ N and $73^0 25'00$ E-$73^0 29'25$E. Submerged and exposed rocks protect corals from strong wave action and endow with ideal substratum for coral settlement. Most of the coral species have grown in patches on the subtidal reef flat. The MMS significantly contributes to the livelihood of the local fishermen population. Apart from the marine fishery, eco-tourism is the fastest growing economic sector associated with coral reefs in this region. However, due to the lack of a management plan and opposition of the local community, the MMS is not operational MPA in the real sense (Rajagopalan, 2008).

Despite the presence of coral reef in the MMS, very little is known so far on coral reef ecology, health status, i.e., coral disease, bleaching, and influence of different environmental drivers on the reef ecosystem. Therefore, a detailed study on reef biodiversity, the extent of the reef formation, the health status of reef-forming corals, and the impact of coastal pressure and changing climatic conditions is carried out through this Ph.D. research work. The present thesis forms a comprehensive study on the biodiversity of scleractinian corals and associated biota, ecology, and present health status of the coral reefs in the MMS.

Objectives:

The following objectives were set for doctoral work:

- To document the distribution and abundance of corals and associated biota.
- To study the seasonal variation in coral-associated species composition.
- To assess the response of selected species to anthropogenic factors.

Structure of the thesis

The thesis is presented in the following eleven chapters:

Chapter-1: This chapter provides a general introduction to the thesis and comprises the rationale of the scientific problem.

Chapter-2: This chapter includes a literature review on coral reef research in India. Wherein I reviewed the documented scleractinian fauna from the Indian water and presented a revised checklist and biogeography of scleractinian from the major Indian reefs.

Chapter-3: This chapter describes details of geomorphology, environmental setup, reef biodiversity, and economic status of the Malvan region on the Central West Coast of India.

Chapter-4: This chapter describes the distribution and abundance of corals of the MMS reefs. The detailed taxonomic identification, distribution, and photographic illustration of the coral species described in this chapter.

Chapter-5 This chapter investigates the distribution and abundance of coral reef-associated fishes.

Chapter-6: This chapter illustrates seasonal variation (pre-monsoon and post-monsoon) in coral reef-associated macroalgal and algal turfs abundance.

Chapter-7: This chapter assesses the seasonal variation (pre-monsoon and post-monsoon) in coral-boring sponges.

Chapter-8: This chapter focuses on the impact of elevated sea surface temperature (SST) on coral species**.** SST induced coral bleaching and bleaching associated mortality was investigated in this chapter.

Chapter-9: This chapter consists of five years of comprehensive observation on the occurrence of several coral diseases (white syndrome, tissue necrosis, infestations of coral boring molluscs, Trematodiasis), and disease impact on corals documented in this chapter.

Chapter-10: Tourism activity related to the beautiful coral reef is rapidly emerging in the MMS. However, unregulated tourism has been proven detrimental to the coral reef in the MMS. Therefore, an attempt to quantify the impact of rapidly growing recreational tourism activity on reef-building corals was made in this chapter.

Chapter-11: This chapter discusses the inference of the doctoral study and recommendations for management practice and the future research scope. This chapter is followed by the references cited in the thesis.

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Chapter 1

1. Introduction

1.1. Rationale

Coral reefs are the wonderland of the underwater world. The word ‘coral’ is originated from the ancient Greek word ‘korallion’, which refers to the valuable red coral of the Mediterranean, known as *Corallum rubrum* (Linnaeus, 1758) (Etnoyer & Morgan, 2003). Reefs are formed by hard corals or Scleractinians, a tiny multicellular invertebrate animal that belongs to the phylum Cnidaria also comprises other marine invertebrates like sea anemones, hydroids, jellyfishes, etc. These organisms are radially symmetrical with absent of head, usually have a crown of tentacles around the mouth, and live in the skeleton, secreted by their tissues. They are some of the oldest animals on the planet and have been surviving on the Earth by building limestone ($CaCO_3$) reef structures. Fossil coral record dated back 450-500 million years to the Ordovician Period of the Paleozoic Era, and the modern-day Scleractinian corals evolved during the Mesozoic era, 245 million years ago, and diversified into today’s reef-building corals in the marine environment (Etnoyer & Morgan, 2003). They are one of the most dynamic, highly sensitive, complex, biologically diverse, highly productive, magnificently beautiful ecosystems on the earth and mostly found in the tropical coastal environment between 30N and 30S latitudes, which constitute half of the world’s coastlines (Fig.1.1). Instead of spreading only 0.00063 percent of Earth’s surface, coral reefs have had significant contributions to the atmospheric and ocean chemistry, the geomorphology of ocean floor and Islands, the diversity and biogeography of life (Birkeland, 2015). Coral reefs support at least 500 million people worldwide through services like fisheries and tourism and provide a substantial amount of goods and services to the world community (Moberg & Folke, 1999). In 1997, global estimation of the goods and services value of coral reefs to average \$6,075 ha^{-1} $year^{-1}$ and worldwide about \$375 billion $year^{-1}$ (Costanza *et al.,* 1997). However, re-estimation in 2014 by further considerations of services such as storm protection, erosion protection, and tourism revealed the annual average per hectare value of coral reefs is about \$352,000 ha^{-1} $year^{-1}$ (Costanza *et al.,* 2014).

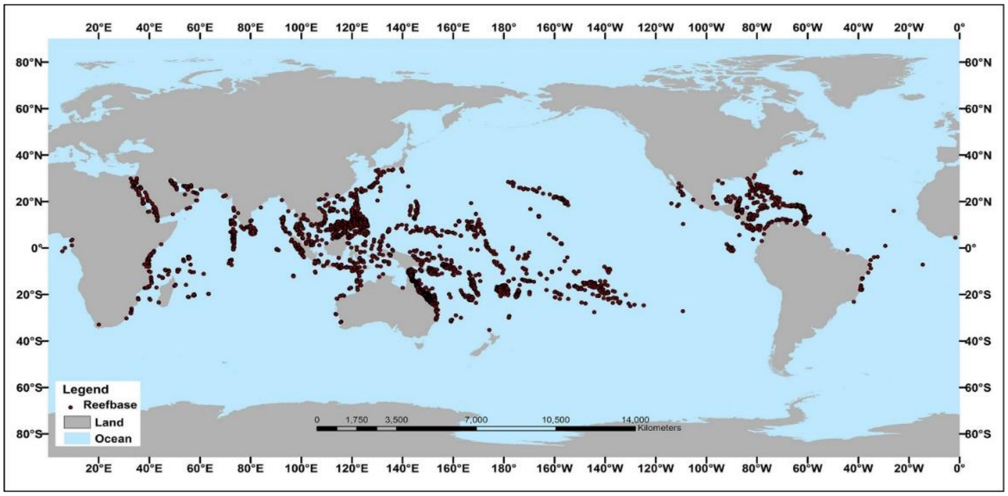

Fig. 1.1. Global distribution of coral reefs (map Sources: http://reefgis.reefbase.org and NOAA, 2017)

Unfortunately, most of the coral reef habitat plummeted significantly, facing a high risk of extinction due to climate change and human disturbances. Recent scientific studies have projected that if the current trend of coral degradation continues uninterruptedly, coral reef ecosystems will shift towards systems that are dominated by other organisms such as cyanobacteria, algae, sponges and other invertebrates (Bellwood *et al.,* 2004; Bell *et al.*, 2013). It is also expected that coral reefs will decline by 70-90% relative to their current abundance by 2050 if the emission of greenhouse gases (GHS) continued under the ‘business-as-usual scenario’ (Hoegh-Guldberg *et al.*, 2018). Moreover, based on current evidence, projection estimated if not by 2050, certainly by the 22nd century, coral reefs possibly succumbed to an anthropogenic extinction (Bowen, 2015).

1.2. Coral reefs in India

The Indian Ocean (IO) is characterized by a lot of highly diverse reefs in its tropical and subtropical regions. India has a long coastline of 8228 Km, and Indian EEZ comprises an area of 2,289,197 Km^2 (http://www.seaaroundus.org). Major Indian reefs have been studied moderately well and are still being considered as a priority area of scientific investigators, which is undoubtedly a good sign. Most of them are under the Marine Protected Areas (MPA) and are thus protected by laws. However, intensive long-term studies on their ecology, biology, microbiology, and bleaching status of these submerged reefs are absent from the literature. This may be because of their remoteness and their submerged presence far away from the nearest land, making their access difficult without specialized research vessels updated with the recent technology; similarly, lack of trained marine biologists, SCUBA divers, and equipment, etc. Development of technology and human resources through rigorous training with funding support would help in carrying forward the coral reef research in India.

Generally, three types of coral reef are present in the Indian water (Fig. 1.2)- Fringing reef (Andaman & Nicobar Islands 1021.46 km^2, Gulf of Kutch 352.50 km^2, Gulf of Mannar, 75.93 km^2); Barrier reef (Gulf of Mannar, Andaman & Nicobar Islands); Atoll (Lakshadweep Islands, 933.7 km^2) (SAC, 2010). Apart from these three main types, there are other types, such as patch reefs, with low generic diversity, which are small structures within the lagoon of other reef and bank reefs. Coral reefs in Indian EEZ comprises 2.48% of the total tropical coral reefs of the world (http://www.seaaroundus.org). Studies have found a living coral bank, about 100 km off the coast of Malpe (North of Mangalore), named as Gaveshani bank (Nair & Qasim, 1978). Hermatypic corals along the shore are reported from Quilon in the Kerala coast to Enayem in Tamilnadu (Pillai, 1996). Qasim and Wafar (1979) documented living corals from the intertidal region of the Central-West coast of India (Ratnagiri, Malvan & Redi). Ramaiyan and Adiyapatham (1985) found evidence of coral on the east coast between Parangipettai (Porto Novo), south of Cuddalore (10°50'N, 79°80'E) and Pondicherry. Vora and Almeida, 1990 noted the presence of a submerged reef system on the continental shelf of the same coastal region between Vengurla and Vijaydurga, parallel to the shoreline at water depths of 60 to 110m.

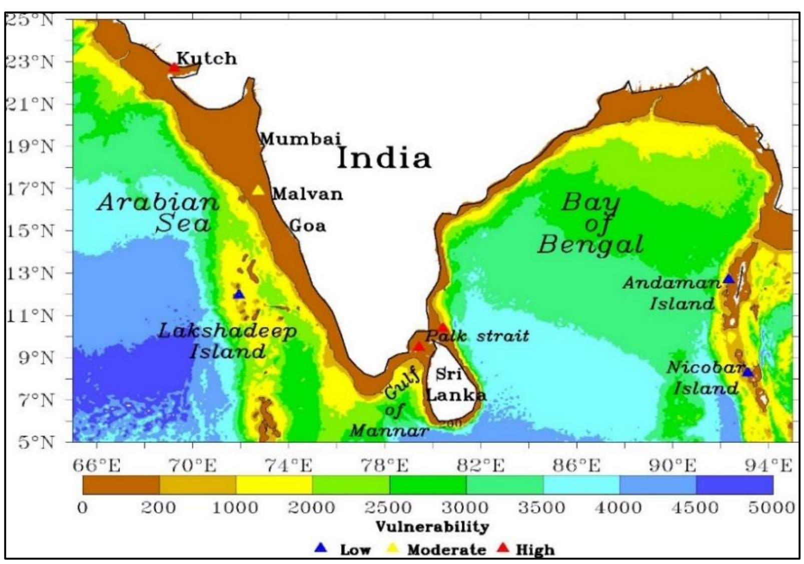

Fig. 1.2. Map representing coral reefs around India, colour scale indicating water depths

Each discovery of a coral island, atoll, or a submerged reef system should be followed with quantitative and qualitative surveys, biodiversity assessment, detailed studies of its health and management, sustainable use, and conservation. The use of new coral reef survey technologies such as remotely operated vehicles (ROVs) and submersibles could help to monitor remote and deep-water reefs. Moreover, it is expected that there would be many more yet to be discovered patches of reefs, scattered across the Indian waters and detailed study for the conservation of these just discovered, yet poorly known coral reefs are need of the hour. However, further research needs to be encouraged by providing the necessary funding and infrastructure.

1.3. Coral reef taxonomy and biodiversity research in India

Speaking about the biodiversity of Indian coral reefs, it is undoubtedly high. Flora and fauna reported so far from Indian water include 1284 species of fish, 3271 species of mollusks, 765 species of echinoderms, 519 species of sponges, 274 species of coral, 607 species of crustaceans and 624 species of algae (Venkataraman *et al.*, 2012a). Additionally, (Venkataraman *et al.*, 2003) documented around 345 species of 87 genera of corals from the reefs of India. The faunal richness in Andaman and Nicobar Islands is remarkable. Around 6000 species were recorded so far, amounting to 7.5% of total Indian fauna, which constitutes approximately 3% of the terrestrial fauna and 4.6 % of marine fauna (Jeyabaskaran & Rao 2007). In Lakshadweep, Rodrigues, 1996 reported 86 species of macrophytes, ten species of Anomuran crabs, 81 species of Brachyuran crabs, 155 species of Gastropods, 24 species of bivalves, 13 species of sea stars, six species of brittle stars, 23 species of sea cumbers, 15 species of sea urchins and 120 species of fish. On comparing coral species diversity of India with neighbouring Indo-Pacific, the recorded more than 208 species number is much poorer than the recorded number of Indo-Pacific, which is around 581. This enlightened the inevitability of more focused studies in the documentation of all coral species from Indian waters (Venkataraman *et al.*, 2003).

Coral reef-dwelling fishes are one of the most diverse and conspicuous constituents of the reef fauna, and because of their broader ecological significance, their monitoring is also necessary. Reef fish diversity survey in a patchy coral reef of Netrani, located near Murudeshwar, Karnataka, founded high fish diversity, which is linked with the complex coral reef system present in the study area (Thomas *et al.*, 2011). Around 600 species of reef fishes were reported from Andaman and Nicobar Islands and 600 species from Lakshadweep (Pillai & Pillai, 2010). Overfishing, especially trawling and bycatch causes the removal of herbivorous fishes that encourage excessive algal growth in the absence of these grazers. In such a way, the coral reef ecosystem is shifted to algal dominated barren without any fish and corals. As a result, many Indian known and unknown reefs might have changed or been turning into algal dominated barren (Ravindran *et al.*, 2012). Further research on such and other ecological themes is required for a better understanding of reef ecology and ecosystem dynamics, which in turn will be useful for the conservation and management of the reef.

1.4. The alarming situation for coral reef ecosystem in a changing ocean

Coral reefs foster rich biodiversity compared to any other ecosystem globally (Done *et al.*, 1996; Fisher *et al.,* 2015). They provide sustainable ground for millions of marine biotas to settle, forage, spawn, and nurture their juveniles (Moberg & Folke, 1999; Hoegh-Guldberg *et al.*, 2018). The ecosystem services from corals have been valued in the billions of dollars and mainly include nutrient cycling, food supply, shoreline protection, erosion regulation, a source of income by boosting fishery and tourism industries (Moberg & Folke, 1999; McCook *et al.*, 2010; Garren & Azam, 2012). Similar to all aquatic and terrestrial habitats, coral reefs are hierarchically organized and are mainly dependent on the presence of the foundation species, i.e., scleractinians or hard corals those generate the robust reef framework by accretion of Calcium carbonate ($CaCO_3$) skeleton. However, the inadvertent climate change condition, mainly ocean warming (OW) and ocean acidification (OA), are severe threats to coral reefs. In recent years, back-to-back elevated temperature-driven coral bleaching resulted in high coral mortality at a global-scale caused far-reaching environmental implications (Hughes *et al.*, 2017). Notably, with OA, coral reefs have been estimated to experience a transition from net precipitation of Calcium carbonate to net dissolution by 2050 (Eyre *et al.,* 2018). The emerging environmental stressors have been plummeting coral reproduction and recruitment capacity leading to gradual impairment of coral resilience capacity (Done *et al.,* 2010; Hughes *et al.*, 2010). The sensitivity or restricted resilience capacity of corals to climate change impacts can be determined from the fact that coral cover began to decline globally in the late 1970s to early 1980s (Gardner *et al.,* 2003; Bruno & Selig, 2007; Schutte *et al.,* 2010). For instance, the average Indo-Pacific coral cover declined from 42.5% during the early 1980s to 22.1% by 2003 (Bruno & Selig, 2007). Also, coral cover is estimated to be dropping by 1-2% $year^{-1}$ (or 3,168 km^2 $year^{-1}$) across the Indo-Pacific (Bruno & Selig, 2007). A recent estimate has suggested that the combined impact of local and global stressors has resulted in a gradual decline of 50 percent of reef-forming corals from across most of the 'world's biologically diverse marine ecosystem', the Great Barrier Reef (Lough *et al.,* 2018). Across the Caribbean, coral reefs have already declined by an average of ~80% (Gardner *et al.,* 2003), and also their recruitment has declined dramatically since the mid-1970s (e.g., Tanner & Hughes, 2000). In a span of 26-years, the coral reef in the Florida Keys underwent a dramatic (>75%) loss of coral cover, particularly Acroporids (Dustan, 2003; Alevizon & Porter, 2015). In Key Largo Dry Rocks, stony coral declined from 57 to 14% between 1974 and 2000 (Alevizon & Porter, 2015), and at Carysfort Reef, living coral cover declined by 92% between 1974 and 1999 (Dustan, 2003). A recent study documented 52% coral cover loss in lagoons of the Florida Keys (McClenachan *et al.,* 2017).

With the advent of increasing anthropogenic CO_2 emissions, the oceans will be a vastly affected ecosystem by the mid to end of this century (Gattuso *et al.,* 2015). Rising levels of CO_2 in the atmosphere resulting in ocean scale warming and OA pushing coral reefs towards a tipping point beyond which the existence of corals is uncertain over the coming years (Kamenos & Hennige, 2018). Future projections estimated the decline in coral cover by 70-90% relative to their current abundance by 2050 if present CO_2 emissions are anticipated to continue as 'business-as-usual scenario' (Beyer *et al.,* 2018; Hoegh-Guldberg *et al.*, 2018) and, even if the goals of the Paris Climate Agreement are achieved (Hoegh-Guldberg *et al.,* 2018). This degradation of corals will increase coastal ecosystem vulnerability by manifold. The coral loss will disturb the ecosystem holistically and negatively affect all the trophic levels from micro-to-macro-to-mega, which is unpredictable. The direct impact of coral loss will be in the form of loss of revenue from tourism and fishing. Further, reduced coastal buffering from extreme weather conditions like tropical cyclones and storm surge will be other inevitable impacts from coral loss (Brander *et al.,* 2007; Harris *et al.,* 2018).

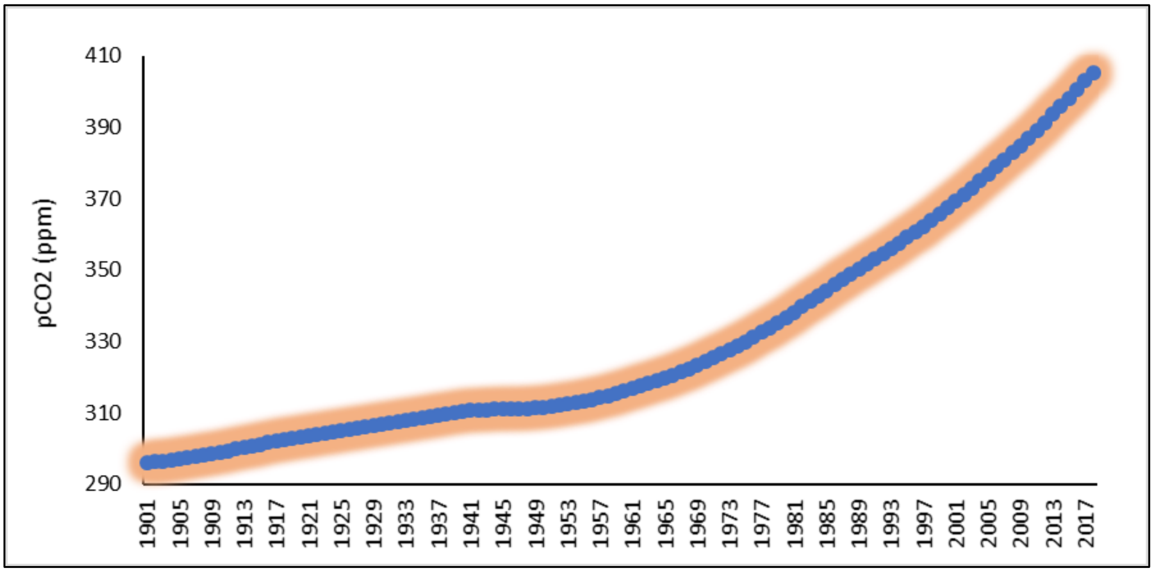

Fig. 1.3. Long-term trends in atmospheric CO_2 concentration (parts per million) (data source: NOAA)

1.5. Coral Bleaching – global crisis incurred by global climate change!

Warming of the global ocean due to caused widespread damage to the coral reefs. Coral bleaching, typically caused by thermal stress, is broadly recognized as one of the significant contributors to the extensive damage of coral reefs (Normile, 2016). The global average temperature has been rising alarmingly during the past few decades, as shown in Fig. 1.3. Moreover, in the past five years, consecutively, turned out as the warmest years in the modern record. Wherein, the year 2016 was the hottest on record in 136 years by breaking the previous record in 2015, 2014. Elevated Sea Surface Temperature (SST), the major component of global climate change, is known as the primary cause of mass coral bleaching events. Coral bleaching events occur when surface waters become so warm and remain high for more than 28 days than there is a breakdown or expulsion of the symbiosis between corals and the microalgae that live within their cells. Generally, the zooxanthellae that inhabit within the gastrodermal tissue of corals fix carbon through photosynthesis process (Muscatine & Porter, 1977). The organic carbons produced through photosynthesis are supplied to the host coral for meeting its nutrient requirements, hence, assist the growth and persistence of tropical corals (Muscatine & Porter, 1977; Yellowlees *et al.*, 2008). The loss of these symbiotic single-celled dinoflagellate *Symbiodinium* or zooxanthellae cause bleaching of the corals and leads to starvation, sickness, and, in many cases, death of the corals. Bleached corals, without in house supply of sugars (through algal photosynthesis), cannot get the resources necessary for their growth, reproduction, and reef construction. This causes coral bleaching or ‘whitening of coral tissue,’ which leads to large-scale mortality of coral species and poses a major threat to the persistence of coral reefs. Coral reefs are the most sensitive of all coastal ecosystems to SST changes and are declining at a disturbingly rapid pace. The combined land and ocean temperature have increased at an average rate of 0.17°C (0.31°F) per decade since 1881 (NOAA, 2018). Nine out of 10 of the warmest years recorded since 2005, and the last five years (2014–2018) ranking as the five warmest years on history (NOAA, 2018). The frequency of coral bleaching events and mortalities have increased since late 1970 and affected coral reefs at a regional scale. Globally, mass coral bleaching, the whitening of the entire reef, occurred in 1998, 2010 and 2016. During 1997–1998, 2010, and 2015–2016, the extreme climate change events El Nino Southern Oscillations (ENSO) events have occurred in tropical oceans, which also raised sea surface temperature in the Indian Seas. Higher SST anomaly with prolong heating events caused worldwide coral bleaching events during these years, Fig. 1.4 shows the range of SST, annual SST anomaly and coral bleaching alert on the IO and coral triangle during the last three mass coral bleaching years (1998, 2010, 2016), based on satellite SST data.

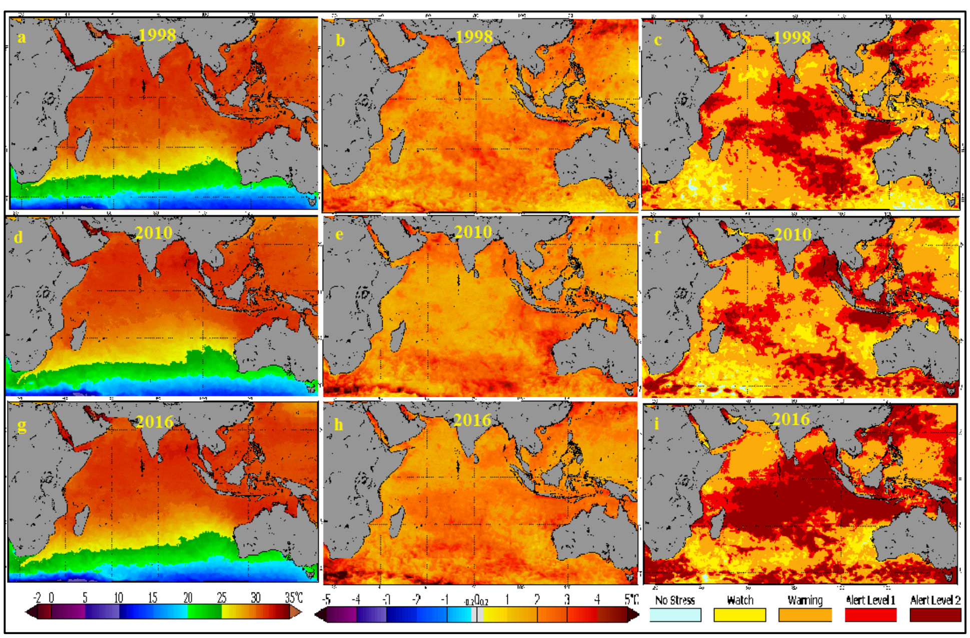

Fig. 1.4. Ranges of annual maximum SST, annual SST anomaly and coral bleaching alert on Indian Ocean coral reefs during the mass coral bleaching years in 1998 (a,b,c); 2010 (d,e,f); 2016

(g,h, i) respectively (Data source: NOAA-CRW 5km SST data; data were visualised using NOAA-CRW platform version 3.1, https://coralreefwatch.noaa.gov/**)**

Although corals can recover themselves after mass bleaching, sometimes it may take decades for the ecosystem to return to the pre-bleaching state. However, in the recent era, an increase in the frequency and severity of bleaching events and growing human pressure could overwhelm the ability of coral reefs to recover between events.

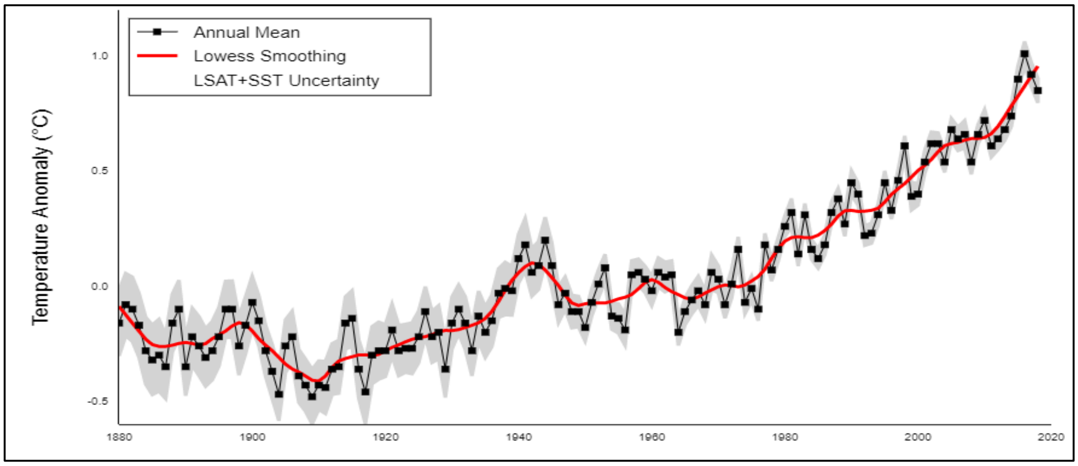

Fig. 1.5. Land-ocean temperature index, 1880 to present, with base period 1951-1980. The solid black line is the global annual mean, and the solid red line is the five-year lowest. The grey shading represents the total (LSAT and SST) annual uncertainty at a 95% confidence interval. (data source NASA-Goddard Institute for Space Studies, Surface Temperature Analysis (NASA/GISS/GISTEMPv4), data visualised through https://data.giss.nasa.gov/gistemp/graphs_v4/#**)**

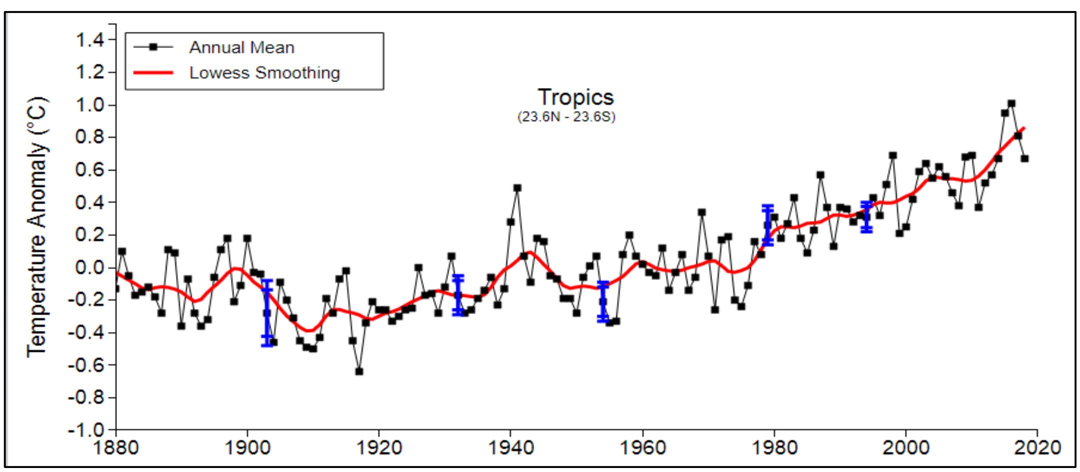

Fig. 1.6. Annual and five-year lowest smoothed temperature changes use land and ocean data, with the base period 1951-1980, the tropics cover 40% of the global area and most of the coral reefs located in this area. Uncertainty bars (95% confidence limits) for the annual (outer) and five-year smooth (inner) are based on a spatial sampling analysis. (data source NASA-Goddard Institute for Space Studies, Surface Temperature Analysis (NASA/GISS/GISTEMPv4), data visualised through https://data.giss.nasa.gov/gistemp/graphs_v4/#**)**

In Indian reefs, mass coral bleaching events caused mass mortality of corals. Six major bleaching events with the associated coral mortality have occurred since 1979, affecting more or less all reefs of the world. The bleaching event of 1997-98 was unprecedented and worst of an ever-recorded mass bleaching event, affecting almost all reefs of the world, including very remote ones (Hoegh-Guldberg, 1999). Arthur (2002) reported 11% coral bleaching with no apparent bleaching-related mortality from the Gulf of Kutch, 82% coral bleaching, and 26% bleaching related mortality from Lakshadweep and 89% coral bleaching and 23% bleaching related mortality from Gulf of Mannar; during April to July 1998. Harithsa *et al.,* (2005) noted temperatures beyond the bleaching threshold in the region of Kavaratti atoll, Lakshadweep, during April-May 2002 with consequent coral bleaching. Recently, Ravindran *et al.*, (2012) studied how coral bleaching prevalence in combination with other stressors are destroying reefs of Palk Bay at an alarming rate. They reported that elevated sea surface temperature in combination with increased irradiance caused coral bleaching in Palk Bay during April 2010. Severe coral bleaching of several key coral species in the South Andaman Islands during the same bleaching event of 2010 (Marimuthu *et al.*, 2013) with little evidence of recovery of corals following the bleaching event of 2010 (Marimuthu *et al.*, 2013).

Sea surface temperatures are expected to increases by 1-2°C over the next century, and even this much of change in global climate may increase the incidences of mass coral bleaching like 1998 one (Hoegh-Guldberg, 1999). Unfortunately, this speculation is appearing to be right with the very recent evidence of bleaching events of 2010, 2016 (Hughes *et al.*, 2017b; Skirving *et al.*, 2019).

Table 1.1. List of the ten warmest years on record globally (1880–2018), represents global annual-average combined land and ocean temperature rank and temperature anomaly (°C) (data source NOAA; https://www.ncdc.noaa.gov/sotc/global/201813#gprcp)

Rank 1 = Warmest period of record: 1880–2018	Year	Anomaly °C
1	2016	0.95
2	2015	0.91
3	2017	0.85
4	2018	0.79
5	2014	0.75
6	2010	0.7
7	2013	0.67
8	2005	0.66
9	2009	0.64
9	1998	0.64

Despite having around 3000 sq.km of coral reef across Indian water, and knowing the fact that the reefs in the Indian ocean are severely impacted by coral bleaching (Bhandhari and Sharma 2010); very few scientific reports could be found from India that have investigated the impact of coral bleaching; which has severe detrimental effects not only on the oceanic biological system but also on the socio-economy of the country.

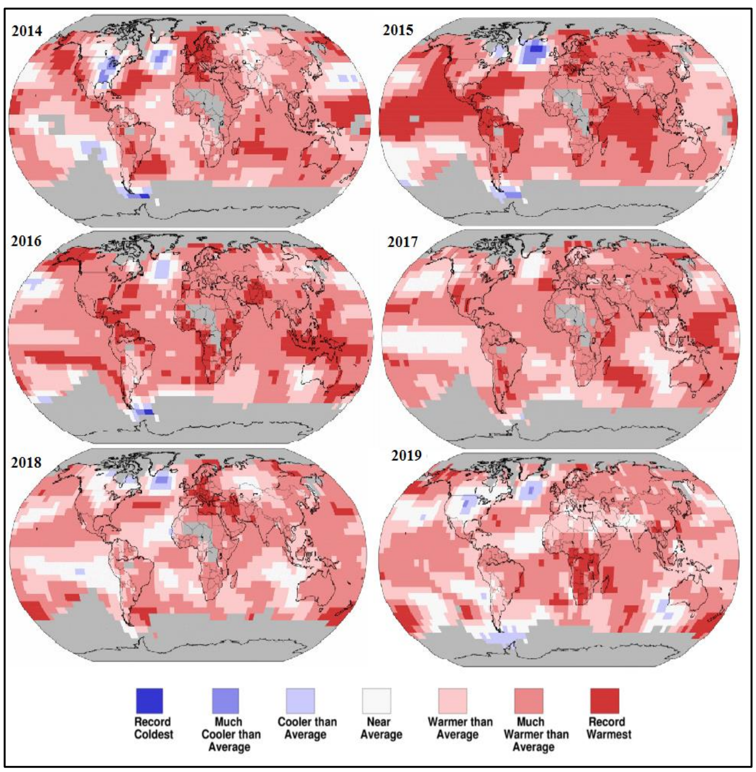

Fig. 1.7. Annual land and ocean temperature percentile since 2014. 2019 data represent the spring months March-May; data source: NOAAGlobalTemv5.0.0-20190610, data visualised using NOAA-NCEI platform (https://www.ncdc.noaa.gov**)**

1.6. Ocean acidification and the future of the coral reef

Human activities have increased atmospheric concentrations of carbon dioxide by 36%, and the pH of ocean surface waters has already declined by about 0.1 units, 8.2 to 8.1, since the beginning of the industrial era (Caldeira & Wickett, 2003). The CO_2 taken up by the ocean reduces the pH and concentration of carbonate ions (CO_3^{2-}) and results in a combination of chemical changes collectively known as ocean acidification (OA) (Orr *et al.,* 2005). The effect of elevated pCO_2 on coral has received particular interest (Marubini & Atkinson, 1999; Kleypas *et al.,* 2001) because calcification rates in corals (which secrete a form of $CaCO_3$, aragonite) decline under increased pCO_2 environment. It may result in decreasing future coral cover and degradation of the reef biodiversity (Kleypas *et al.,* 2001; Hoegh-Guldberg *et al.,* 2018; Mcleod *et al.,* 2019).

Langdon *et al.,* (2003) predicted that coral calcification rates might decrease by 21–40% over the period 1880–2065 in response to changes in atmospheric CO_2 concentrations. Based on the amount of coral reef that has died in the Caribbean in the past few years, studies predicted that over 65% of the earth's coral reefs would be at very high to critical high risk by 2050 if global warming and local human impact continues unchecked (Burke *et al.,* 2011). Even if climate mitigation efforts achieve to draw down atmospheric CO_2 levels to pre-industrial level, the time required for the regaining of seawater chemistry will be detrimental for corals and recovery process (Hoegh-Guldberg *et al.*, 2017).

In a nutshell, there are several confounding aspects and theories proposed so far that are involved in some way in the mechanisms that lead to bleaching, bleaching related mortality, the physiological response of holobiont against bleaching, and the effect of predicted ocean acidification. This area is still under continuous active research.

1.7. Coral physiological performance in the changing ocean

The global concentration of atmospheric CO_2 will increase from a present-day partial pressure (pCO_2) of 405.0±0.1 ppm (Le Quéré *et al.*, 2018) to between 730-1088 ppm by 2100 (Meehl *et al.,* 2007). The advent of increasing atmospheric CO_2 levels invariably increasing the rate of CO_2 uptake by the ocean, resulting in a 30% increase in ocean acidity since preindustrial times and could hasten up to 170% by 2100 unless CO_2 emissions decline (Ciais *et al.*, 2013). This increase CO_2 dissolution rate significantly changes the seawater carbonate chemistry, causing a reduction in pH and carbonate saturation, and an increase in dissolved inorganic carbon availability (Caldeira & Wickett, 2003). This condition mainly affects carbonate-accreting organisms like reef-building corals (Anthony *et al.*, 2008).

Likewise, elevated Sea Surface Temperature (SST) prodigiously detrimental to corals in several ways, most importantly, the thermal stress breakdown crucial Coral–algal (*Symbiodinium* spp.) symbiosis which leads to coral bleaching followed by coral mortality (e.g., Sully *et al.*, 2019) and have contributed to drastic drops in coral cover worldwide (~50–80%) since the 1970s (Gardner *et al.*, 2003; Bruno & Selig, 2007). Additionally, coral bleaching often causes a far-reaching effect on the post-bleaching metabolic and demographic condition, such as a reduction in reproduction, larval survival, growth rates, and resulted in reduced species richness (Anthony *et al.*, 2008; Albright, 2011; Albright & Mason, 2013; Prada *et al.*, 2017; Fordyce *et al.*, 2019). Moreover, the experimental study also confirmed a decline in coral fertilization success by reducing sperm concentrations with an increase in the levels of temperature and pCO_2 (Albright & Mason, 2013). The combinatorial impact of temperatures and OA inhibits calcification rate, impair skeleton formation or decline in growth, and also cost additional energy, which may hamper other critical metabolic functions like feeding, digestion, reproduction, and immunity (Wall *et al.*, 2019). Reef accretion may be further compromised by the dissolution of exiting calcium carbonate reef framework in high CO_2 state (Manzello *et al.*, 2008; Comeau *et al.*, 2016). A recent year-long experimental high CO_2 adaptation study on *Pocillopora verrucosa*, *Psammocora profundacella*, *Acropora pulchra*, *Porites* spp. and calcifying macroalgal species *Lithophyllum kotschyanum*, *Halimeda minimato* by determining calcifying fluid chemistry revealed no evidence of acclimatization to elevated pCO_2, which indicates looming threats of OA for reef calcification and ecosystem function (Comeau *et al*., 2019).

Furthermore, Knutson *et al.*, (2010) demonstrated that ocean warming could increase the physical damage of corals by driving more frequent cyclones. It has also reported that tropical cyclones associated with high rainfall and runoff will negatively impact the structural persistence of coral reefs by $CaCO^{-3}$ undersaturation (e.g., Manzello *et al.*, 2013). On the other hand, an average increase of 3.2 mm $year^{-1}$ from 1993 to 2010 in average global sea levels was documented because of the OW and the melting of land and sea ice (Ciais *et al.,* 2013). Consequently, tropical coral reefs in Southeast Asia and Northern Australia have reported rates of sea-level rise of around 10mm year^{-1} (Hoegh-Guldberg *et al.*, 2017), is predicted that ongoing sea-level rise will increase coastal wave exposure, which may further degrade the reef framework and limit vertical reef growth (Perry *et al.,* 2013). Considering the consequences of ongoing climate change, i.e., i) OW, ii) OA and iii) sea-level rise, it is projected that coral reefs are about to decline by a further 70–90% at 1.5^{0}C warming with substantial losses (>99%) at 2^{0}C by the end of this century (IPCC, 2018).

1.8. How are the coral reefs transitioning to the macroalgal forest?

Macroalgae are the major competitors of corals for space on tropical reefs (Rasher & Hay, 2010; Rasher *et al.,* 2011), and are taking over reefs that have been weakened by multiple stressors including of recurrent bleaching events, sedimentation, eutrophication, overfishing, and storms (Mccook *et al.,* 2001; Bellwood *et al.,* 2004). Being an opportunistic species with high reproductive capacity and fast growth macroalgae often outcompete slow-growing coral species. The term 'Coral-algal phase shifts' coined as the coral cover declines to their low levels and is replaced by algae in many reefs across worldwide. Studies demonstrated that macroalgae would either benefit from or remain relatively unaffected by projected OW and OA, which is detrimental to the coral reefs (Hughes *et al.*, 2010). Evidence of coral-macroalgae regime shifts from field data have been supplemented by experimental laboratory investigation (e.g., Hughes *et al.,* 2007), theoretical (e.g., Knowlton, 1992; Mcmanus & Polsenberg, 2004) and through modelling (e.g., Mumby *et al.,* 2007). Studies demonstrated that nutrients enrichment in the coastal ocean from terrigenous sources promotes algal growth, and subsequent fishing of the key herbivores has resulted in the proliferation of macroalgae (Rasher *et al.*, 2012; Bozec *et al.*, 2016).

Additionally, some macroalgal species like *Laurencia dendroidea*, *Fucus vesiculosus* can produce chemical defenses in response to herbivory, further resulted reduce grazing by herbivore (Jormalainen & Honkanen, 2008; Sudatti *et al.,* 2018). Studies showed that when protected from herbivores, approximately 40 to 70% of common macroalgae cause bleaching and death of coral tissue when in direct contact (Rasher & Hay, 2010). Further, macroalgae can cause immediate knock-off effect to corals by transfer of hydrophobic allelochemicals present on algal surfaces leading to coral bleaching, inhibiting photosynthesis, localized tissue death and occasionally death of corals (Rasher & Hay, 2010; Rasher *et al.*, 2011). Moreover, a recent study has demonstrated that increasing ocean acidification benefits some macroalgae over corals by enhancing the allelopathy (Del Monaco *et al.*, 2017). Additionally, benthic algae also host a variety of different potential coral pathogens and could transmit coral diseases like White Syndrome and Yellow Band Disease (Sweet *et al.*, 2013). These shifts from a coral to an algae-dominated state "phase-shifts," are likely to be accelerated and expected to occur more frequently (Fong & Paul, 2011). Moreover, this algal dominated alternative stable state is difficult to be reversed in the coral-dominated state (Briggs *et al.,* 2018; Schmitt *et al.,* 2019). In a nutshell, these phenomena will lead to an increasing frequency of coral-macroalgae contacts, increasing allelopathic suppression of remaining corals, and a continuing decline of reef corals.

Furthermore, recent studies predicted that global environmental change is going to increase the frequency of extreme weather events like tropical storms and cyclones (Gutmann *et al.,* 2018). These natural calamities are widely known for mechanically disturbing the coral reefs (Hoegh-Guldberg *et al.*, 2018). Also, these events will cause a higher terrigenous nutrient influx in coastal reefs, which significantly supports the growth of algal biomass (Mejia *et al.,* 2012). This antagonistic impact over two communities will cause dramatic ecosystem changes (Hannah, 2015; Welsh & Bellwood, 2015).

1.9. Threats, Conservation & Management

Growing concern for the coral reef is driving the need for broader research to inform conservation management all over the world. India is blessed with around 5,790 km^2 of the coral reef region, but like other parts of the world, Indian reefs are also in colossal peril (De *et al.*, 2017). Consumerist anthropogenic impacts on coral reefs have augmented dramatically for various reasons since the last century. By the end of the previous century, except reefs of Andaman & Nicobar, hardly any reefs of mainland India were pristine. These small islands are facing multiple challenges to protect their natural resources and for resilience. In India, for better management of its reef ecosystem, identification of future research is needed, topics including taxonomy, population biology, ecosystem functioning, environmental effects, and human impacts. Let us look into some major threats to Indian reefs:

- There is a range of natural calamities causing immense destruction of corals to include cyclone, local tectonic upheavals, tsunami, siltation and pests and predators like coral-eating Crown of Thorns Starfish (COTS) *Acanthaster plansi* and outbreaks of different diseases including White pox, White band, and Black band causes extensive damage to reefs.
- Coral mining for cement industries has been continuously affecting Indian coral reefs, in the past, it was open business, and currently, it is being practiced illegally as national parks covering the reef area have banned these practices. It was estimated that annually about 15,000 tons of coral stones are removed from four islands near Tuticorin alone (Venkataramanujam *et al.*, 1981).
- Uncontrolled tourism is the other major problem responsible for the degradation of Indian reefs. Tourists hanging out, swimming, snorkeling, and diving in unmanaged areas result in significant damage to shallow water corals by unintentionally or intentionally trampling and breaking coral fragments.
- Souvenir hunting, uncontrolled harvesting, smuggling various exotic marine organisms and products derived from the reef is still a serious concern for the reef managers. These sorts of damages are mechanical damages caused by direct human involvement.
- Destructive fishing practices are also responsible for such type of damage. Many banned malpractices like blast fishing are still going on in many coral reef regions like the Gulf of Mannar, which is believed to be massively damaged and destroyed reef region of India (Venkataraman, 2003). Removal of crucial herbivores fishes by overfishing caused grave impacts on coral reefs; hence, the reef-fishing practice needs to be managed to make it sustainable. Monitoring of fishing trawlers is required to minimize by-catches, which could have severely impacted the submerged coral reef ecosystem, which is still unknown and not even intensively studied.
- Other human impacts that arise through land runoff, waste disposal, shipping, and such other activities are of pollution, which is a widespread and well-known problem for Indian coral reefs (De *et al.,* 2017; Nanajkar *et al.,* 2019). Conservation of Indian coral reefs are enforced under the Wild Life (Protection) Act (WLPA) 1972, Marine Fishing Regulation Act (MFRA) 1983, 2000, and the Coastal Regulation Zone (CRZ) Notification, 2011; under which coral reefs of India are protected by law. There are a total of 31 Marine Protected Areas (MPAs) in India. Among them, five coral reef regions have been surveyed and identified for protection. Pillai

(1996) recommended halting all human intervention and malpractices that are responsible for mechanical damage and removal of corals and bringing numerous marine organisms except scientific research. To replenish the degraded coral reef regions in the Lakshadweep Islands, Venkatesh *et al.,* (2008) successfully transplanted scleractinian species *Acropora* and *Pocillopora* on artificial metal frames placed on a sandy bottom in the degraded lagoon. In the same study, they noted 90% survival and establishment of associated fauna such as reef fish along with the transplanted corals. Based on this experience, they suggested that such re-introduction of coral species is a practical and cost-effective strategy for conserving and restoring the degraded areas (Chavanich *et al.,* 2015; Nanajkar *et al.,* 2019).

In a nutshell, the focus, particularly in developing countries, is mainly on food species production and economic growth, whereas conservation of biodiversity and living resources is ignored to a certain extent. Fortunately, it appears that realizing the problem, efforts have been put for conservation and management of abundant coral reef resources. Implementation of robust laws, conservation strategies like integrated watershed management connecting land and sea, appropriate management and use of biological resources, modern technology, expert marine biologists, indispensable further scientific research are crucial in fulfilling this goal.

Though efforts are being made at the governmental and non-governmental levels to conserve effectively and sustainably the exploitation of Indian corals, the efforts are not sufficient. Moreover, there is a dire need for effective communication between the scientific community, stakeholders, and managers and strict implication of conservation practice to protect the spectacular coral reef ecosystem (De *et al.*, 2017). However, there are many challenges, political back up, the courage of local stakeholders and scientific knowledge, integration, and positive alliance among the central government organization, state departments, NGOs, scientists, and educators are essential for the betterment of the situation. Ontogeny of artificial reef restoration methods such as transplantation and underwater silviculture should be heightened. Future directions should be set up accordingly in meeting these challenges.

Chapter 2

2. Review of literature

2.1. The hard corals (Scleractinia) of India: a revised checklist

Coral reefs are incredibly diverse, valuable ecosystems and millions of people depend on the coral reef for their livelihood and food security (Hughes *et al.*, 2017). Nevertheless, these are facing a bleak future worldwide due to unprecedented climate change and rapid coastal development (Hughes *et al.*, 2017; Hughes *et al.*, 2018; lough *et al.*, 2018). Coral reefs are distributed on the East coast (Bay of Bengal) and the west coast (Arabian Sea) (Fig.2.1). Being less than one percent, i.e., 2383.87 km^2 of the total reef formation of the world (DOD & SAC, 1997; Venkataraman *et al.*, 2003), coral reefs in the Indian water is highly crucial in respect of ecosystem service and economy. These reefs vary from a small patch reef in the Eastern Arabian Sea to extensive barrier reefs in the Andaman Sea, and shallow water fringing reefs to deep water corals. Some of the Indian reefs show unique phenomena of elasticity and resilience. Gulf of Kachchh (GoK) shows unique resistance to some extreme climatic conditions like high SST and sedimentation.

On the other hand, reefs in Lakshadweep and Andaman Islands are some of the near-pristine reefs due to remoteness and less human perturbation. Moreover, most reefs are also subjected to ongoing climate change and elevated SST since the last decade resulted in several bleaching events in all the reefs in India (De *et al.*, 2017). Some of them severely degraded, significantly lost species richness, and structural reef complexity. Most of the coral species representing Indian reefs belong to the widespread Indo-pacific species group. However, some are endemic to Indian water (Venkataraman *et al.*, 2003), viz. *Montipora jonesi* Pillai 1969; *Montipora manauliensis* Pillai 1969; *Porites exserta* Pillai 1969; *Porites mannarensis* Pillai 1969; *Porites minicoiensis* Pillai, 1969; *Alveopora superficialis* Pillai and Scheer, 1976; and *Favites monticularis* Mondal, Raghunathan and Venkataraman 2013. *Ctenactis triangularis* Mondal, Raghunathan 2013 (taxon inquirendum). Reefs in the Andaman and Nicobar Islands (A&N) are biologically diverse due to its geographic proximity and connectivity to the Indo-Pacific coral triangle (Venkataraman *et al.*, 2003).

Taxonomical studies on coral in India is dated back to 1847 by Rink from the Nicobar Islands (Venkataraman *et al.*, 2003). After a prolonged gap, Pillai (1967) conducted an extensive study on the coral fauna of the Gulf of Mannar and the Lakshadweep, and he listed a total of 125 species of corals of 34 genera and one subgenus (Pillai, 1967). In a series of publications, Pillai demonstrated and documented species richness and coral community structure in the Gulf of Mannar, the Lakshadweep islands and the Andaman and Nicobar Islands. A comprehensive account of the coral fauna by Pillai (1987) included 155 species of hermatypic corals belonging to 50 genera and 44 species and ahermatypic corals distributed among 21 genera constitute 135 species of 59 genera from the A&N Islands, 78 species of corals belonging to 31 genera from the Lakshadweep, 94 species allocated among 37 genera from the Southeast coast of India and 37 species belonging to 24 genera from the GoK. After the pioneering work of Pillai, the Zoological Survey of India (ZSI) has initiated the coral reef research and has documented the hidden biodiversity in different coral reefs in Indian water, especially from the Andaman and Nicobar Islands. The effort of ZSI has yielded several new records of Scleractinans from Indian water. Venkataraman *et al.,* (2003) documented 208 species of Scleractinans belonging to 60 genera and 15 families from India, of which 177 species were from A&N, 91 species from Lakshadweep, 82 species from GoMBR, and 36 were from GoK. Subsequently, Turner *et al.,* (2009) reported 234 species of scleractinian coral from the Andaman and Nicobar Islands with several new reports for the first time from the Andaman Islands as well as from India. A considerable effort by various researchers has significantly increased the total species number of scleractinian in the Indian reefs for the last two decades, and most of the species are reported in the Andaman and Nicobar Islands.

2.2. Methodology

The objective of the present article is to document the diversity of Scleractinian faunas in Indian reefs and to present an updated checklist. In the present study, we referred to the latest taxonomic nomenclature presented in the *World List of Scleractinia,* accessed through the World Register of Marine Species (WoRMS) database (http://www.marinespecies.org) to update the taxonomic status. The species distribution range of the reported species was confirmed using Coral of the World, (accessed through http://coralsoftheworld.org). The species list compiled and curated here is based on the extensive literature search and data mining of all the available published literature (scientific reports, journal articles, thesis) on scleractinian coral diversity from Indian reefs using an online database, includes Web of Science, Google Scholar, digital archives of Zoological Survey of India (ZSI) and Central Marine Fisheries Research Institute (CMFRI). Based on the occurrence of scleractinian species in different Indian reefs, a reef-wise species list was prepared to show the diversity of each reef. Additionally, we provided an annotated list of species that were erroneously reported by the previous studies. Besides the species distribution list, we compared the unique and cosmopolitan scleractinian species distribution across the GoK, LKD, GoMBR, and A&N Islands using Venny 2.1 (Fig. 2.2).

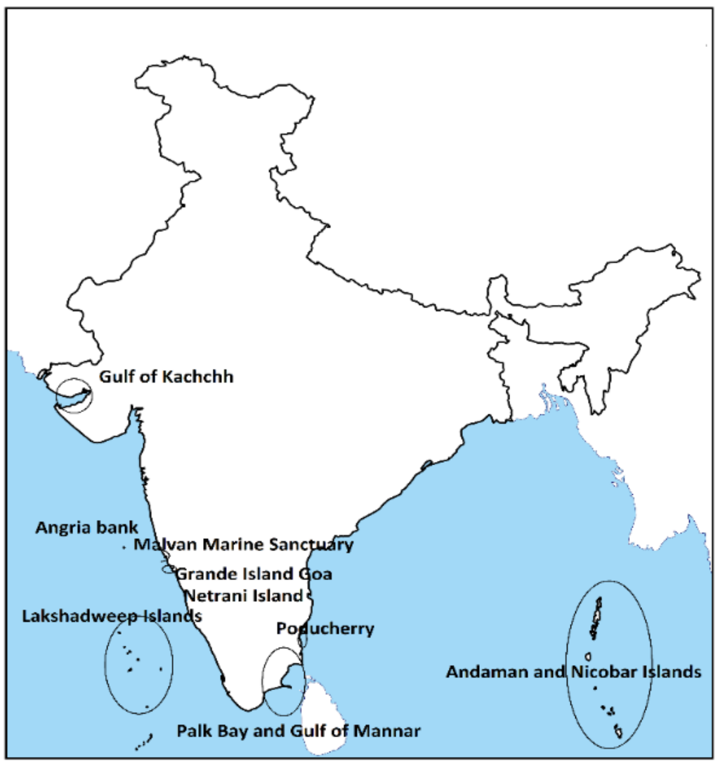

Fig. 2.1. Distribution of major coral reefs in Indian water.

2.3.Result

The present checklist consists of a total of 589 species belonging to 108 genera and 22 families of scleractinian fauna. Maximum species diversity is recorded in the Andaman and Nicobar Islands 525 species of 92 genera and 22 families followed by Lakshadweep Islands 167 species of 56 genera and 18 families, Gulf of Mannar Biosphere reserve 168 species of 47 genera and

16 families and Gulf of Kachchh 78 species of 30 genera and 12 families. Overall species assemblage, Acroporids show the highest number of species diversity of 185 species belongs to six genera. Then, the Merulinidae family includes 102 species of 20 genera, followed by the Poritidae family consist of 52 species of four genera. The most commonly occurring genera of corals are *Acropora* (104 species), *Montipora* (56 species), *Porites* (30), *Dipsastrea* (20 species), *Goniopora* (20 species), *Favites* (18 species), *Lobophyllia* and *Pavona* both contribute 16 species. Reefs species list provided in table 2.1.

Corals from the Merulinidae family are the most common in the GoK (26 species of nine genera), followed by Acroporidae (13 species of two genera). Family Poritidae and Dendrophillidae, both represented by ten species belonging to three genera. However, dead and fossilized *Acropora* are found in different patches in the GoK, but, report of live specimen needs to be confirmed (Dixit *et al.,* 2010; Satyanarayana & Ramakrishna, 2009).

In the Lakshadweep Islands, Acroporidae (51 species of five genera), Merulinidae (34 species of 14 genera), and Poritidae (17 species of two genera) form the significant species assemblage. At the genera level, *Acropora* represented by 36 species, followed by *Porites* (14 species) and *Montipora* (nine species).

In the GoMBR, we recorded occurrence reports of 57 species Acroporidae belonging to three genera, Merulinidae (45 species of 14 genera), Poritidae (19 species of four genera), are the most commonly occurring family. The most common genera are *Acropora*, represented by 33 species, followed by *Montipora* (22 species).

Acroporids formed the dominant species assemblage in the A&N Islands, consist of a total of 164 species belonging to six genera. Followed by the Merulinidae family, they contribute 130 species distributed between 19 genera. Forty-three species represented by Fungiidae belongs to 15 genera. Lobophylliidae was the next dominant family, consist of 32 species and nine genera. Dendrophyllidae were contributed by 29 species belong to nine genera. At the genera level, *Acropora* is the most common genus, consist of 97 species, followed by *Montipora* and *Porites*, contribute 48 and 25 species, respectively. Moreover, most of the studies in the A&N Islands are limited to the Mahatma Gandhi Marine National Park and a few other islands, whereas, many Islands (there are 572 islands in A&N Islands) are mostly unexplored (Ramakrishna *et al.*, 2010).

The presence of 301 species only found in the reefs of the A&N Islands. Whereas, the GoMBR serves as a home for 27 unique scleractinian species. The distribution of 10 species of scleractinians was only restricted in the LKD Islands. GoK has two unique species, namely *Acanthastrea simplex* and *Erythrastrea flabellata*. We found 36 common scleractinian species present across the reefs in A&N Islands, GoMBR, LKD, and GoK. Names of these species are present in table 2.1 with *mark.

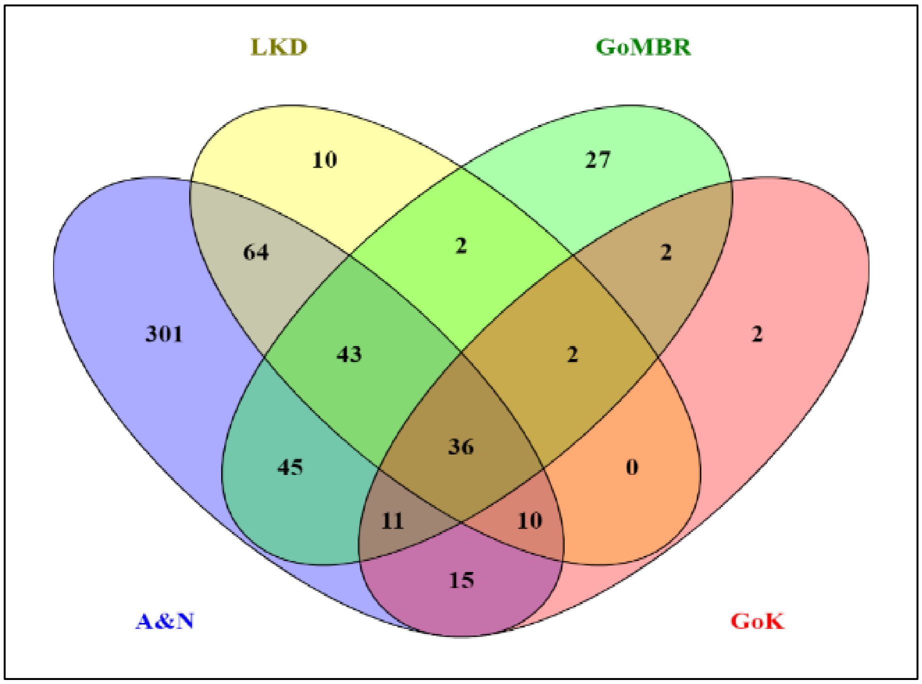

Fig. 2.2. Comparative analysis for occurrence similarity and uniqueness of reported coral species (percentage and number) across the major Indian reefs (A&N=Andaman and Nicobar Islands; LKD=Lakshadweep Islands; GoMBR: Gulf of Mannar Biodiversity Reserve; GoK: Gulf of Kachchh)

Apart from the four main coral reefs of India, the occurrence of patch reefs has been recorded from a few locations on the central west coast of the country. These reefs are characterized by rocky substratum, high turbidity due to land-based runoff, and corals can be found from intertidal rock pools to 15 m subtidal region[28-29]. Information on the biodiversity of these reefs is still sparse, therefore more detailed study required to elucidate the faunal diversity.

Distribution of scleractinian fauna were reported from Ratnagiri, Redi, south of Bombay, Malvan Marine Sanctuary (Qasim &Wafar, 2979) in Maharashtra coast, Grande Islands in Goa coast (Singarayan & Rethnaraj, 2016; Manikandan *et al.*, 2016), Netrani Island in Karwar coast (Zacharia *et al.*, 2008), and Angria bank off Malvan coast (Ingole, 2017). The presence of hard coral species is also reported from Quilon in the Kerala coast to Enayem in Tamilnadu (Pillai, 1996). Pillai and Jasmine (1995) reported the occurrence of 13 species of hermatypic corals belonging to six genera and 16 species of ahermatypic corals belonging to 11 genera from a depth of 40 to 100 meters in the southwest coast (Kerala, Tamilnadu) of India (Pillai & Jasmin, 1995). Here, we enlisted the hard-coral species documented so far from these reefs. Reports on Scleractinian fauna from the coral reefs of the West coast of India were mostly from the Malvan Marine Sanctuary (MMS), Grande Islands, Netrani Island, and the Angria bank. In the MMS, reported species includes *Porites lichen* Dana, 1846, *Porites lutea* (Quoy & Gaimard, 1833); *Goniopora pedunculata* Quoy & Gaimard, 1833, *Goniopora* sp., *Coscinaraea monile* (Forskål, 1775), *Pseodosiderastrea tayami* Yabe & Sugiyama, 1935, *Siderastrea savignyana* Milne Edwards and Haime, 1850, *Cyphastrea* sp., *Turbinaria* sp., *Synarea* sp., *Montastrea* sp., *Leptastrea* sp., *Pavona* sp., *Goniastrea retiformis* (Lamarck, 1816), *Favites halicora* (Ehrenberg, 1834), *Favites* sp., *Leptastrea purpurea* (Dana, 1846), *Tubastraea coccinea* Lesson, 1829, *Polycyathus verrilli* Duncan, 1889, *Pavona bipartita* Nemenzo, 1979 (Qasim & wafer, 1979, Parulekar, 1981, Raj *et al.*, 2017).

In the Grande Islands, presence of *Porites* sp., *Goniopora* sp., *Coscinaraea* sp., *Pocillopora* sp., *Siderastrea* sp., *Turbinaria* sp., *Montastrea* sp., *Leptastrea* sp., *Goniastrea* sp., *Favites* sp, *Favia* sp., *Plesiastrea* sp., *Balanophyllia cumingii* Milne Edwards and Haime, 1848, *Dendrophyllia indica* Pillai, 1969, *Paracyathus profundus* Duncan, 1889 were recorded (Manikandan *et al.*, 2016, Singarayan & Rethnaraj, 2016).

Zacharia *et al.*, (2008) reported occurrence of *Porites* sp., *Goniopora* sp., *Coscinaraea* sp., *Coscinaraea monile* (Forskål, 1775), *Pocillopora verrucosa* (Ellis and Solander, 1786), *Pocillopora* sp., *Turbinaria* sp., *Symphyllia* sp., *Leptastrea* sp., *Dendrophyllia* sp., *Goniastrea retiformis* (Lamarck, 1816), *Goniastrea pectinate* (Ehrenberg, 1834), *Favia favus, Plesiastrea versipora* (Lamarck, 1816) in the Netrani Island, Karnataka coast.

Among these reefs, an underwater survey by Ingole (2017) in the Angria bank revealed the highest number of species assemblage, includes *Acanthastrea sp., Sclerophyllia sp., Lobophyllia corymbosa* (Forskål, 1775), *Dipsastraea sp., Dipsastraea speciosa* (Dana, 1846), *Echinophyllia* sp., *Echinophyllia pectinata* Veron, 2000, *Mycedium* sp., *Scolymia sp., Fungia* sp., *Ctenactis* sp., *Echinopora sp., Galaxea sp., Favites* sp., *Goniastrea* sp., *Paragoniastrea* sp., *Leptastrea* sp., *Psammocora* Sp., *Plesistrea versipora* (Lamarck, 1816), *Astreopora* sp., *Euphyllia ancora* Veron & Pichon, 1980, *Coelastrea* sp., *Pachyseris speciosa* Dana, 1846, *Platygyra* sp., *Leptoseris* sp, *Pocillopora* sp., *Porites lobata* Dana, 1846, *Porites solida* Forskal, 1775, *Goniopora* sp., *Symphyllia* sp, *Turbinaria mesenterina* (Lamarck, 1816), *Turbinaria peltata* Esper, 1794.

More recently, Laxmilata *et al.,* (2019) documented mesophotic coral reef-associated biota from Puducherry. They reported the occurrence of 12 species belonging to ten genera and seven families, viz. *Leptoseris explanata* Yabe & Sugiyama, 1941, *Pavona minuta* Wells, 1954, *Pavona maldivensis* (Gardiner, 1905), *Tubastraea micranthus* (Cairns and Zibrowius, 1997), *Tubastraea coccinea* Lesson, 1829, *Euphyllia ancora* Veron and Pichon, 1980, *Hydnophora rigida* (Dana, 1846), *Goniastrea pectinata* (Ehrenberg, 1834), *Dipsastraea favus* (Forskål, 1775), *Psammocora haimeana* Milne Edwards & Haime, 1851, *Pachyseris speciosa* (Dana, 1846), and *Cycloseris* sp.

2.4.Discussion

Reefs in the Gulf of Kachchh (GoK) are in the located north-western part of the Indian Arabian Sea and home of some of the most northern reefs in the world (Kelleher *et al.,* 1995). Patel (1976), Pillai *et al.,* (1979), and Pillai & Patel (1988) presented the comprehensive account of coral diversity and distribution in the GoK (Patel, 1976; 1978). Further, Singh *et al.,* (2004) reported 42 species of hard-coral belonging to seven families and 24 genera (Singh *et al.,* 2003). Satyanarayana & Ramakrishna (2009) documented 49 species of corals with new records of *Barabattoia amicorum, Favia lacuna, Favites flexuosa,* and *Turbinaria frondens* from Indian water (Satyanarayana & Ramakrishna, 2009). Raghuraman *et al.,* (2012) reported the occurrence of 49 species belonging to 27 genera and ten families (Raghuraman *et al.,* 2012). In a detailed study on reef ecology in the GoK, Sreenath (2015) noted the occurrence of 31 species of hard corals belonging to 20 genera and nine families and mentioned the new record of

Goniopora djiboutiensis, *G. stokesi*, *Hydnophora pilosa* from the GoK (Sreenath, 2015). Further, Kumar *et al.*, (2017) presented an updated checklist showing the presence of 56 species belonging to 27 genera and ten families (Kumar *et al.*, 2017); however, they left out the new records described by Sreenath (2015). Moreover, a recent study from GoK reported the presence of 53 species of hard coral based on available literature[43]. However, in the present checklist, we compiled all the occurrence records and the number of the total hard coral fauna of the GoK represented by 78 species of 31 genera and 12 families, which is comparatively higher than earlier checklists.

In the south-western part of India, Lakshadweep reef archipelago located 200-400km away from the Indian mainland and formed by a series of coral atolls. Pillai and Jasmine (1989) reported 104 species of scleractinians, of which 26 species were a new record to the Lakshadweep (Pillai, 1989). Suresh (1991) recorded 105 species of scleractinian fauna, with a new record of 22 species and four genera (*Herpolitha, Leptoseris, Oulophyllia,* and *Pachiseris*) (Suresh, 1991). Caeiro (1999) studied coral fauna of the Lakshadweep Islands and reported the occurrence of 96 species of corals belonging to 34 genera and listed 28 new records for Lakshadweep (Caeiro, 1999). Moreover, Jeyabaskaran (2007) reported an additional occurrence of 20 species under 13 genera from this region (Jeyabaskaran, 2009). Additionally, Raghuraman *et al.*, 2012 recorded presences of 104 species (37 genera and 13 families) from the LKD Islands (Raghuraman *et al.*, 2012).

Coral reefs in the Gulf of Mannar Marine Biosphere reserve (GoMBR) are the southernmost reefs of India, located in Tamil Nadu, the Southeast coast of India. The presence of diverse types of reef forms such as fringing, shore platform, patch, and coral pinnacles was found in the Gulf of Mannar and Palk Bay. Pillai (1986) described 94 species in 37 genera. Patterson *et al.*, (2007; 2008) provided a comprehensive account of the coral fauna from this region and reported the presence of 117 species. Furthermore, Raghuraman *et al.*, 2012 enumerated 117 species belonging to 40 genera and 14 families from these reefs. Additionally, Venkataraman & Rajan (2013) reported the occurrence of 34 species from this region with 16 new distribution records, which is lower than the earlier finding of 63 species by Pillai (1969). A recent study has identified 51 species from the GoMBR with 17 new distribution records from this region (Krishna *et al.*, 2018).

Coral reefs in the Andaman and Nicobar Islands in the Bay of Bengal are known for remarkable faunal diversity. Scheer & Pillai (1974) and Pillai (1977, 1978, 1983) documented the diversity and distribution of corals of Andaman and Nicobar Islands. In Andaman and Nicobar Islands, the detailed taxonomic study of scleractinian fauna accelerated by the Zoological Survey of India (ZSI), Port Blair. Turner *et al.,* (2001) recorded a total of 198 species of scleractinian coral from different islands of Andaman, of which 111 were new records to India. Subsequently, Venkataraman *et al.,* (2003) described 208 species of hard coral species with detailed taxonomic descriptions and discussion on different coral reefs of India.

Additionally, Ramakrishna *et al.,* (2010) described 419 species of hard corals from the Andaman and Nicobar Islands with a new occurrence record of 85 species of scleractinia. Thereafter, another attempt to compose a checklist of corals from the significant reefs was made by Raghuraman *et al.,* (2012); they reported 478 species under 89 genera and 19 families, of which 424 species (86 genera and 19 families) from Andaman and Nicobar Islands. Subsequently, Mondal *et al.,* (2016) also presented an account of 173 species (48 genera and 14 families) from the Great Nicobar Island.

Moreover, extensive exploratory surveys and taxonomic work in recent times unveiled hundreds of scleractinians in a series of publications in the A&N Islands, including a description of a novel species *Favites monticularis* Mondal, Raghunathan and Venkataraman (2013). Mondal *et al.,* (2017) reported the occurrence of a total of 628 species of hard corals from Indian reefs, and out of these, 588 species were from Andaman and Nicobar Islands. However, after detailed literature searches, we enlisted in 532 species of 96 genera and 22 families in the present checklist.

Furthermore, we found some of these records are based on erroneous identification; for example, the occurrence report of the Caribbean species *Porites porites* from Andaman (Ramakrishna *et al.,* 2010), wherein the photographs in the same description resemble *Heliopora* sp.; an Octocoral species (personal communication with Dr. Douglas Fenner). We also noticed that multiple occasions, species were reported from the Indian coral reefs are endemic to the Atlantic Ocean, or in the Caribbean and other geographical areas. Such as *Halomitra clavator* (Höeksema, 1989) is native to Indonesia, Philippines, Malaysia, and Papua New Guinea, but reported from the A&N Islands (Mondal & Raghunathan, 2012). Similarly, *Diploria clivosa* (Ellis & Solander, 1786) reported from the GoMBR (Krishnan *et al.,* 2018), is a Caribbean species. Likewise, *Mussismilia braziliensis* (Verrill, 1868) is endemic to Brazilian water but reported from the Andaman (Mondal *et al.,* 2015). Similarly, *Cantharellus noumeae* Hoeksema and Best, 1984, is an endemic species of New Caledonia and does not occur elsewhere. Additionally, several species were reported erroneously from the A&N Islands by different authors are generally native to the Caribbean and the Atlantic Ocean. For example, *Mycetophyllia danaana* Milne Edwards and Haime, 1849 (Ramakrishna *et al.*, 2010), *Pseudodiploria strigosa* (Dana, 1846) (Ramakrishna *et al.*, 2010), *Agaricia fragilis* Dana, 1848 (Ramakrishna *et al.,* 2010), *Favia fragum* (Esper, 1797) (Mondal *et al.*, 2015), *Mussa angulosa* (Pallas 1766) (Reddiah, 1977), *Solenastrea bournoni* Milne Edwards and Haime, 1849 (Ramakrishna *et al.,* 2010; Mondal *et al.,* 2010), *Siderastrea radians* (Pallas, 1766) (Raghuraman *et al.,* 2012), *Siderastrea siderea* (Ellis and Solander, 1786) (Mondal *et al.,* 2011), *Leptoseris cucullata* (Ellis and Solander, 1786) (Ramakrishna *et al.*, 2010; Raghuraman *et al.*, 2012), *Porites porites* (Pallas, 1766) (Mondal *et al.*, 2010), *Mycetophyllia lamarckiana* Milne Edwards and Haime, 1848 (Mondal & Raghunathan, 2010), *Leptoseris cucullata* (Ellis & Solander, 1786). Therefore, these species were excluded from the present checklist. In another instance, the occurrences of *Montastrea annularis* (Ellis and Solander, 1786) were reported from the A&N Islands (Mondal *et al.*, 2013) and GoMBR (Krishnan *et al.,* 2018), which is s previous combination, wrong genus spelling of *Orbicella annularis* (Ellis and Solander, 1786) and is native to Atlantic water (Hoeksema & Cairns, 2019; IUCN, 2018), hence, we excluded this record from this checklist. In a few instances, we also found that some species reported in different synonymy claiming new occurrences from Indian water. For example, *Acropora cytherea* (Dana, 1846) was reported from the Lakshadweep by Pillai (1971), and in the A&N Islands by Mondal *et al.*, (2014) as *Acropora efflorescens* (Dana, 1846), whereas, same species was recorded from the A&N Island *as Acropora armata* (Brook, 1892) by Reddiah (1977), also as *Acropora corymbosa* (Lamarck, 1816) by Reddiah (1977), and again as *Acropora reticulata* (Brook, 1892) by Pillai (1971) from the Lakshadweep Islands. To mitigate such ambiguity, we used the WoRMS database (Hoeksema & Cairns, 2019) to identify the synonymous entries and excluded the synonymous records and present the current valid species name. Considering, these occasions, we have added a list of 198 species of scleractinian along with detailed remarks, those were reported in different literature, but have been excluded in the present checklist.

Coral identification solely based on morphological observation and underwater monitoring comes with a certain amount of uncertainties, as corals show phenotypic plasticity and intraspecific variation in appearance and skeletal characteristics across the habitat and geographic location (Veron *et al.*, 2019). Coral taxonomy research in India is so far mostly based on morphological identification or the underwater observation. This problem further aggravates as the coral collection is legally restricted in India; hence, researchers need to rely on field identification (Manikandan *et al.,* 2016). Moreover, identification of coral species solely based on underwater field observation and underwater photographs often leads to erroneous identification, and unfortunately, those misidentifications have been used in over subsequent other publications. We also admit that some of the entries in the present checklist are based on the list of coral species recorded in a different publication, and we could not verify these reports, as are lacking taxonomic details and photographs of the species. Some of the species discussed in this work were reported as synonyms by some and subsequently as misidentifications, and only a detailed taxonomic study with a more comprehensive geographical range, preferably comparison of the coral skeleton with the other holotype sample, is essential to delineate the Indian scleractinian fauna accurately. Inclusion of synonyms and endemic species of the Atlantic waters by some workers brought the taxonomic ambiguity in some of the previous occurrence reports from Indian water and, therefore, need an urgent revision of the voucher specimen and application of advanced molecular tools for confirmation of species in India. These limitations triggered the preparation of the present checklist to document the valid species in Indian reefs. Hence, a dedicated taxonomical research program with a combination of standard morphological identification keys and incorporation of molecular phylogenetic techniques, along with inter-institute or international collaboration, would be desirable to unveil new coral records and rectification of earlier erroneous reports, which will be helpful to underline conservation policies.

Table 2.1. The revised checklist of hard coral from India (x: not reported)

Sr. No.	Species Name	Gulf of Kachchh (GoK)	Gulf of Mannar Marine Biosphere Reserve (GoMBR)	Lakshadweep Islands (LKD)	Andaman & Nicobar Islands (A&N)
Family ACROPORIIDAE Verrill, 1902 Genus *Acropora* Oken, 1815					
1.	*Acropora abrotanoides* (Lamarck, 1816)	x	Pillai 1967	Pillai 1972	Reddiah 1977
2.	*Acropora abrolhosensis* Veron, 1985	x	Edward *et al.*, 2007	x	x
3.	*Acropora acuminata* (Verrill, 1864)	x	x	x	Mondal *et al.*, 2015
4.	*Acropora anthocercis* (Brook, 1893)	x	x	x	Venkataraman *et al.*, 2003; Venkataraman *et al.*, 2012
5.	*Acropora arabensis* Hodgson & Carpenter, 1995	x	Geetha & Kumar 2012	x	x
6.	*Acropora aspera* (Dana, 1846)	x	x	Pillai 1989	Pillai 1967
7.	*Acropora austera* (Dana, 1846)	x	x	Suresh 1991	Turner *et al.*, 2009; Venkataraman *et al.*, 2012
8.	*Acropora awi* Wallace & Wolstenholme, 1998	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2011
9.	*Acropora batunai* Wallace, 1997	x	x	x	Mondal *et al.*, 2014
10.	*Acropora bifurcata* Nemenzo, 1971	x	x	x	Mondal *et al.*, 2014
11.	*Acropora branchi* Riegl,1995	x	Geetha & Kumar 2012	x	x
12.	*Acropora capillaris* (Klunzinger, 1879	x	x	Suresh 1991	x
13.	*Acropora carduus* (Dana,1846)	x	x	x	Turner *et al.*, 2009
14.	*Acropora caroliniana* Nemenzo, 1976	x	x	x	Venkataraman *et al.*, 2003; Ramakrishna *et al.*, 2010
15.	*Acropora cerealis* (Dana, 1846)	x	x	Caeiro 1999	Turner *et al.*, 2009
16.	*Acropora cervicornis* (Lamarck, 1816),	x	x	Sreenath *et al.*, 2015	Mondal *et al.*, 2015; Mondal *et al.*, 2017 Mondal *et al.*, 2015
17.	*Acropora chesterfieldensis* Veron & Wallace, 1984	x	Krishnan *et al.*, 2018	x	Turner *et al.*, 2009
18.	*Acropora clathrata* (Brook, 1891)	x	x	x	Reddiah 1977
19.	*Acropora cophodactyla* (Brook,1892)	x	x	x	Turner *et al.*, 2009
20.	*Acropora cytherea* (Dana, 1846)	x	Pillai 1967	Pillai 1971	Reddiah 1977; Mondal *et al.*, 2014
21.	*Acropora dendrum* (Bassett-Smith, 1890)	x	x	x	Mondal *et al.*, 2014

22.	*Acropora desalwii* (Wallace, 1994)	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2014
23.	*Acropora divaricata* (Dana,1846)	x	x	Suresh 1991	Turner *et al.*, 2009
24.	*Acropora digitifera* (Dana,1846)	x	Pillai 1967e	x	Reddiah 1977
25.	*Acropora donei* Veron & Wallace, 1984	x	x	x	Turner *et al.*, 2009
26.	*Acropora echinata* (Dana, 1846)	x	Pillai 1967e	Pillai 1971	Pillai 1967e
27.	*Acropora elseyi* (Brook, 1892)	x	x	x	Mondal *et al.*, 2013
28.	*Acropora exigua* (Dana, 1846)	x	Pillai 1967e	x	x
29.	*Acropora exquisita* Nemazo, 1971 (*nomen dubium*)	x	x	x	Mondal *et al.*, 2013; Raghunathan 2015
30.	*Acropora fastigata* Nemenzo, 1967	x	x	x	Mondal *et al.*, 2011
31.	*Acropora florida* (Dana,1846)	x	Pillai 1967e	Suresh 1991	Reddiah 1977
32.	*Acropora forskali* (Ehrenberg, 1834) (*nomen dubium*)	x	Krishnan *et al.*, 2018	Pillai 1971	Mondal *et al.*, 2011; Venkataraman *et al.*, 2012
33.	*Acropora gemmifera* (Brook, 1892)	x	Venkataraman & Rajan, 2013	Sreenath *et al.*, 2015	Turner *et al.*, 2009
34.	*Acropora glauca* (Brook,1893)	x	x	x	Venkataraman *et al.*, 2003; Venkataraman *et al.*, 2012
35.	*Acropora globiceps* (Dana, 1846)	x	x	x	Turner *et al.*, 2009
36.	*Acropora gomezi* Veron, 2000	x	x	x	Ramakrishna *et al.*, 2010; Raghuraman *et al.*, 2012
37.	*Acropora grandis* (Brook, 1892)	x	x	x	Reddiah 1977
38.	*Acropora granulosa* (Milne Edwards, 1860)	x	x	Pillai 1989	Turner *et al.*, 2009
39.	*Acropora haimei* Edwards,1860 (taxon inquirendum)	x	Pillai 1967	Pillai 1971	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2011
40.	*Acropora hemprichii* (Ehrenberg, 1834)	x	Sukumaran *et al.*, 2007	Pillai 1971	Turner *et al.*, 2009
41.	*Acropora hoeksemai* Wallace, 1997	x	x	x	Mondal *et al.*, 2014
42.	*Acropora horrida* (Dana, 1836)	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
43.	*Acropora humilis* (Dana, 1846)*	Pillai & Patel 1988	Pillai 1967	Pillai 1989	Pillai 1969
44.	*Acropora hyacinthus* (Dana, 1846)	x	Pillai 1967	Pillai 1971	Pillai 1972
45.	*Acropora indica* (Brook, 1893) (*nomen dubium*)	x	Brook 1893	Pillai 1971	x
46.	*Acropora intermedia* (Brook, 1891)	x	Pillai 1967e	Pillai 1971	Reddiah 1977
47.	*Acropora insignis* Nemenzo, 1967 (*nomen dubium*)	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2010; Mondal *et al.*, 2017
48.	*Acropora kimbeensis* Wallace, 1999	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*,

					2011; Mondal *et al.*, 2014
49.	*Acropora kosurini* Wallace, 1994	x	x	x	Turner *et al.*, 2009
50.	*Acropora latistella* (Brook, 1892)	x	x	x	Mondal *et al.*, 2014
51.	*Acropora lamarcki* Veron 2000	x	Sukumaran *et al.*, 2007	Sreenath *et al.*, 2015	x
52.	*Acropora loisetteae* Wallace, 1994	x	x	x	Turner *et al.*, 2009
53.	*Acropora longicyathus* (Milne Edwards, 1860)	x	x	Sreenath *et al.*, 2015	Pillai 1967e
54.	*Acropora lovelli* Veron & Wallace, 1984	x	x	x	Mondal & Raghunathan 2016
55.	*Acropora loripes* (Brook, 1892)	x	x	x	Venkataraman *et al.*, 2003 Turner *et al.*, 2009
56.	*Acropora lutkeni* Crossland, 1952	x	x	x	Turner *et al.*, 2009
57.	*Acropora microclados* (Ehrenberg, 1834)	x	x	x	Venkataraman *et al.*, 2003; Ramkrishna *et al.*, 2010; Venkataraman *et al.*, 2012
58.	*Acropora microphthalma* (Verrill, 1859)	Satyanarayana & Ramakrishna 2009	x	Sreenath *et al.*, 2015	Venkataraman *et al.*, 2003; Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
59.	*Acropora millepora* (Ehrenberg,1834)	x	x	Caeiro 1999	Tikader *et al.*, 1986
60.	*Acropora minuta* Veron, 2000	x	x	x	Raghuraman *et al.*, 2012#
61.	*Acropora mirabilis* (Quelch,1886) (*nomen dubium*)	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
62.	*Acropora monticulosa* (Brüggemann, 1879)	x	x	Pillai 1971	Reddiah 1977
63.	*Acropora multiacuta* Nemenzo, 1967	x	x	x	Tikader *et al.*, 1986
64.	*Acropora muricata* (Linnaeus, 1758)	x	Venkataraman *et al* 2003	Venkataraman *et al* 2003	Venkataraman *et al.*, 2003
65.	*Acropora nana* (Studer, 1877)	x	x	x	Venkataraman *et al.*, 2012
66.	*Acropora natalensis* Riegl, 1995	x	x	x	Mondal *et al.*, 2013
67.	*Acropora nasuta* (Dana, 1846)	x	Edward *et al.*, 2007	Pillai 1989	Reddiah 1977
68.	*Acropora palmerae* Wells, 1954	x	x	x	Reddiah 1977; Venkataraman *et al.*, 2003
69.	*Acropora paniculata* Verrill, 1902	x	x	x	Turner *et al.*, 2009
70.	*Acropora papillare* Latypov, 1992	x	x	x	Venkataraman *et al.*, 2003; Venkataraman *et al.*, 2012
71.	*Acropora pectinata* Veron, 2000	x	x	x	Mondal *et al.*, 2015b
72.	*Acropora pharaonis* (Milne Edwards, 1860)	x	Pillai 1967	Pillai 1971	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
73.	*Acropora plantaginea* (Lamarck,1816) (*nomen dubium*)	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012

74.	*Acropora polystoma* (Brook, 1891)	x	Krishnan *et al.*, 2018	x	Venkataraman *et al.*, 2003; Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
75.	*Acropora proximalis* Veron, 2002	x	x	x	Turner *et al.*, 2009
76.	*Acropora pulchra* (Brook, 1891)	x	x	Caeiro 1999	Reddiah 1977
77.	*Acropora pruinosa* (Brook, 1893)	Raghuraman *et al.*, 2013 (location not mentioned)			
78.	*Acropora retusa* (Dana, 1846)	x	Sukumaran *et al.*, 2007	x	x
79.	*Acropora robusta* (Dana,1846)	x	Pillai 1967	Pillai 1989	Pillai 1967
80.	*Acropora roseni* Wallace, 1999	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
81.	*Acropora rudis* (Rehberg,1892)	x	Venkataraman *et al.*, 2003	x	Turner *et al.*, 2009
82.	*Acropora samoensis* (Brook,1891)	x	Sukumaran *et al.*, 2007	x	Ramakrishna *et al.*, 2010, Mondal *et al.*, 2014
83.	*Acropora sarmentosa* (Brook, 1892)	x	x	x	Turner *et al.*, 2009
84.	*Acropora secale* (Studer, 1878)	x	Pillai 1967 e	x	Pillai 1967e
85.	*Acropora selago* (Studer,1878)	x	x	Suresh 1991	Turner *et al.*, 2009
86.	*Acropora solitaryensis* Veron & Wallace, 1984	x	x	x	Venkataraman *et al.*, 2003 Turner *et al.*, 2009
87.	*Acropora spicifera* (Dana, 1846)	x	Pillai 1967	x	Turner *et al.*, 2009
88.	*Acropora speciosa* (Quelch, 1886)	x	x	Pillai 1971	Tikader *et al.*, 1986
89.	*Acropora squarrosa* (Ehrenberg,1834)*	Pillai & Patel 1988	Pillai 1967e	Pillai 1971	Reddiah 1977
90.	*Acropora subglabra* (Brook, 1891)	x	x	x	Venkataraman *et al.*, 2003; Venkataraman *et al.*, 2012
91.	*Acropora subulata* (Dana, 1846)	x	x	x	Ramakrishna *et al.*, 2010; Mondal & Raghunathan 2017
92.	*Acropora striata* (Verrill, 1866)	x	x	x	Mondal *et al.*, 2014
93.	*Acropora tenuis* (Dana, 1846)	x	x	Caeiro 1999	Turner *et al.*, 2009
94.	*Acropora teres* (Verrill, 1866) (*nomen dubium*)	x	x	Pillai 1989	Mondal *et al.*, 2014
95.	*Acropora tanegashimensis* Veron, 1990	x	x	x	Mondal *et al.*, 2010
96.	*Acropora thurstoni* (Brook, 1893) (*nomen dubium*)	x	Pillai 1967	x	x
97.	*Acropora turaki* Wallace, 1994	x	x	x	Raghuraman *et al.*, 2012; Mondal *et al.*, 2014
98.	*Acropora valenciennesi* (Milne Edwards & Haime, 1860)	x	Pillai 1967; Venkataraman *et al.*, 2003	Suresh 1991	Turner *et al.*, 2009
99.	*Acropora valida* (Dana, 1846)	Pillai 1972	x	Caeiro 1999	Pillai 1972

100.	*Acropora variolosa* (Klunzinger, 1879)	x	x	x	Ramakrishna *et al.,* 2010; Venkataraman *et al.,* 2012; Mondal *et al.,* 2014
101.	*Acropora vaughani* Wells, 1954	x	x	x	Turner *et al.,* 2009
102.	*Acropora verweyi* Veron & Wallace,1984	x	Geetha & Kumar 2012	x	Ramakrishna *et al.,* 2010; Venkataraman *et al.,* 2012
103.	*Acropora yongei* Veron & Wallace, 1984	x	x	x	Mondal & Raghunathan 2016
104.	*Acropora willisae* Veron & Wallace, 1984	x	x	x	Mondal *et al.,* 2014
Genus *Alveopora* Blainville, 1830					
105.	*Alveopora allingi* Hoeffmeister, 1925	x	x	x	Mondal *et al.,* 2013; Mondal *et al.,* 2014; Mondal *et al.,* 2015
106.	*Alveopora catalai* Wells, 1968	x	x	x	Ramakrishna *et al.,* 2010; Mondal & Raghunathan 2017
107.	*Alveopora daedalea* (Forskål, 1775)	x	x	x	Tikader *et al.,* 1986
108.	*Alveopora gigas* Veron, 1985	x	x	x	Mondal *et al.,* 2013; Mondal *et al.,* 2015
109.	*Alveopora marionensis* Veron & Pichon, 1982	x	x	x	Sadhukhan & Raghunathan 2012
110.	*Alveopora verrilliana* Dana, 1846	x	x	x	Venkataraman *et al.,* 2003; Venkataraman *et al.,* 2012
111.	*Alveopora superficialis* Pillai & Scheer, 1976	x	x	Pillai 1989	Venkataraman *et al.,* 2003
Genus *Anacropora* Ridley, 1884					
112.	*Anacropora forbesi* Ridley, 1884	x	x	x	Mondal *et al.,* 2012
113.	*Anacropora pillai* Veron, 2000	Raghuraman *et al.,* 2013 (location not mentioned)			
114.	*Anacropora reticulata* Veron & Wallace, 1984	x	x	x	Ramakrishna *et al.,* 2010; Venkataraman *et al.,* 2012
115.	*Anacropora spinosa* Rehberg, 1892	x	x	x	Mondal *et al.,* 2015
Genus *Astreopora* Blainville, 1830					
116.	*Astreopora cucullata* Lamberts, 1980	x	x	x	Venkataraman *et al.,* 2003; Venkataraman *et al.,* 2012
117.	*Astreopora gracilis* Bernard, 1896	x	x	Suresh 1991	Turner *et al.,* 2009
118.	*Astreopora incrustans* Bernard, 1896	x	x	x	Ramakrishna *et al.,* 2010; Venkataraman *et al.,* 2012
119.	*Astreopora listeri* Bernard, 1896	x	x	Pillai 1989	Tikader *et al.,* 1986
120.	*Astreopora myriophthalma* (Lamarck,1816)	x	x	Pillai 1989	Turner *et al.,* 2009; Venkataraman *et al.,* 2012
121.	*Astreopora ocellata* Bernard, 1896	x	x	Caeiro 1999	Ramakrishna *et al.,* 2010; Venkataraman *et al.,* 2012
122.	*Astreopora randalli* Lamberts, 1980	x	x	x	Turner *et al.,* 2009; Venkataraman *et al.,* 2012

123.	*Astreopora scabra* Lamberts, 1982	x	x	x	Mondal *et al.*, 2015
124.	*Astreopora suggesta* Wells, 1954	x	x	x	Turner *et al.*, 2009; Venkataraman *et al.*, 2012
Genus *Isopora* Studer, 1878					
125.	*Isopora brueggemanni* (Brook, 1893)	x	Krishnan *et al.*, 2018	x	Reddiah 1977
126.	*Isopora cuneata* (Dana, 1846)	x	Pillai (1967)	x	Raghunathan 2015; Mondal *et al.*, 2019
127.	*Isopora crateriformis* (Gardiner, 1898)	x	x	x	Mondal *et al.*, 2014
128.	*Isopora elizabethensis* (Veron, 2000)	x	x	x	Raghuraman *et al.*, 2012, 2013
129.	*Isopora palifera* (Lamarck, 1816)	x	x	Pillai 1971; Sreenath *et al.*, 2015	Reddiah 1977; Venkataraman *et al.*, 2003
Genus *Montipora* Blainville, 1830					
130.	*Montipora aequituberculata* Bernard,1897	x	Pillai 1967e	x	Reddiah 1977
131.	*Montipora angulata* Lamarck, 1816	x	x	x	Venkataraman *et al.*, 2003; Venkataraman *et al.*, 2012
132.	*Montipora caliculata* (Dana, 1846)	x	x	x	Turner *et al.*, 2009
133.	*Montipora capitata* (Dana, 1846)	x	x	x	Turner *et al.*, 2009
134.	*Montipora capricornis* Veron, 1985	x	x	x	Mondal *et al.*, 2014
135.	*Montipora cebuensis* Nemenzo,1976	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012; Mondal *et al.*, 2017
136.	*Montipora circumvallata* (Ehrenberg, 1834)	x	x	x	Mondal *et al.*, 2014
137.	*Montipora cocosensis* Vaughan, 1918	x	x	x	Tikader *et al.*, 1986
138.	*Montipora confusa* Nemenzo, 1967	x	x	x	Mondal *et al.*, 2013
139.	*Montipora corbettensis* Veron & Wallace, 1984	x	x	x	Raghunathan *et al.*, 2015; Mondal *et al.*, 2015
140.	*Montipora crassituberculata* Bernard, 1897	x	x	x	Mondal *et al.*, 2012
141.	*Montipora danae* Milne Edwards & Haime, 1851	Singh *et al.*, 2003	x	x	Rajan *et al.*, 2010
142.	*Montipora delicatula* Veron, 2000	x	x	x	Ramakrishna *et al.*, 2010; Raghunathan *et al.*, 2013
143.	*Montipora digitata* (Dana, 1846)*	Singh *et al.*, 2003	Venkataraman *et al.*, 2003	Venkataraman *et al.*, 2003	Pillai 1967
144.	*Montipora edwardsi* Bernard, 1897	x	Pillai 1967e	x	x
145.	*Montipora efflorescens* Bernard, 1897	x	x	x	Mondal *et al.*, 2013
146.	*Montipora effusa* (Dana, 1846)	x	x	x	Mondal *et al.*, 2012
147.	*Montipora elschneri* Vaughan, 1918 (*taxon inquirendum*)	x	Pillai 1967	x	x
148.	*Montipora explanata* Brüggemann, 1879	Pillai & Patel 1988	Pillai 1967	Pillai 1989	x

149.	*Montipora exserta* Quelch, 1886 (*taxon inquirendum*)	x	Pillai 1967	x	x
150.	*Montipora feveolata* (Dana,1846)	x	x	Jeyabaskaran 2009	x
151.	*Montipora flabellata* Studer,1901	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
152.	*Montipora florida* Nemenzo, 1967	x	x	x	Reddiah 1977
153.	*Montipora foliosa* (Pallas,1766)*	Pillai & Patel 1988	Bernard 1897	Pillai 1989	Pillai 1967
154.	*Montipora foveolata* (Dana, 1846)	x	x	Caeiro 1999	Turner *et al.*, 2009
155.	*Montipora friabilis* Bernard, 1897	x	Geetha & Kumar 2012	x	x
156.	*Montipora gaimardi* Bernard, 1897	x	x	x	Mondal *et al.*, 2011
157.	*Montipora granulosa* Bernard, 1897	x	Pillai 1967	x	x
158.	*Montipora grisea* Bernard, 1897	x	x	x	Ramakrishna *et al.*, 2010; Mondal & Raghunathan 2017
159.	*Montipora hemispherica* Veron, 2000	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
160.	*Montipora hispida* (Dana, 1846)	Pillai & Patel 1988	x	Suresh 1991	Reddiah 1977
161.	*Montipora informis* Bernard, 1897	x	Pillai 1967e	x	Venkataraman *et al.*, 2012; Mondal *et al.*, 2015, 2017
162.	*Montipora jonesi* Pillai, 1986	x	Pillai 1983, 1986	x	x
163.	*Montipora manauliensis* Pillai, 1967	x	Pillai 1967a	x	x
164.	*Montipora meandrina* (Ehrenberg, 1834)	x	x	x	Turner *et al.*, 2009
165.	*Montipora millepora* Crossland, 1952	x	Pillai 1967e	x	Mondal *et al.*, 2013
166.	*Montipora mollis* Bernard, 1897	x	x	x	Mondal *et al.*, 2013; 2015
167.	*Montipora monasteriata* (Forskål, 1775)	Pillai & Patel 1988	Pillai 1967e, Venkataraman & Rajan, 2013	x	x
168.	*Montipora nodosa* (Dana, 1846)	x	x	x	Mondal *et al.*, 2014
169.	*Montipora palawanensis* Veron, 2000	x	x	x	Ramakrishna *et al.*, 2010
170.	*Montipora peltiformis* Bernard, 1897	x	Sukumaran *et al.*, 2007	x	Tikader *et al.*, 1986
171.	*Montipora porites* Veron, 2000	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2015
172.	*Montipora samarensis* Nemenzo, 1967	x	Krishnan *et al.*, 2018	x	Mondal *et al.*, 2012
173.	*Montipora spongiosa* (Ehrenberg, 1834)	x	x	x	Mondal *et al.*, 2013
174.	*Montipora spumosa* (Lamarck, 1816)	x	Pillai 1967e	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2015

175.	*Montipora taiwanensis* Veron, 2000	x	x	x	Ramakrishna *et al.*, 2010, Mondal *et al.*, 2015
176.	*Montipora tortuosa* Dana, 1846	x	x	x	Tikader *et al.*, 1986
177.	*Montipora turgescens* Bernard, 1897*	Pillai & Patel 1988	Pillai 1967e	Pillai 1989	Tikader *et al.*, 1986; Turner *et al.*, 2009
178.	*Montipora tuberculosa* (Lamarck,1816)	x	Sukumaran *et al.*, 2007	Pillai 1989	Turner *et al.*, 2009
179.	*Montipora turtlensis* Veron & Wallace, 1984	x	x	x	Mondal *et al.*, 2012; 2017
180.	*Montipora undata* Bernard, 1897	x	x	x	Ramakrishna *et al.*, 2010; Raghunathan *et al.*, 2015; Mondal *et al.*, 2017
181.	*Montipora verrucosa* (Lamarck, 1816)	Singh *et al.*, 2003	Pillai 1967e	x	Turner *et al.*, 2009; Venkataraman *et al.*, 2012
182.	*Montipora verruculosa* Veron, 2000	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
183.	*Montipora venosa* (Ehrenberg, 1834)*	Pillai & Patel 1988	Pillai 1967e	Pillai 1989	Turner *et al.*, 2009; Venkataraman *et al.*, 2012
184.	*Montipora verrilli* Vaughan, 1907	x	Pillai 1967e	x	Venkataraman *et al.*, 2012
185.	*Montipora vietnamensis* Veron, 2000	x	x	x	Turner *et al.*, 2009; Venkataraman *et al.*, 2012; Mondal *et al.*, 2019
Family ASTROCOENIIDAE Koby, 1890					
Genus *Madracis* Milne Edwards & Haime, 1849					
186.	*Madracis interjecta* Marenzeller, 1906	x	Edward *et al.*, 2008	x	x
187.	*Madracis kirbyi* Veron & Pichon, 1976	x	Venkataraman *et al.*, 2003	x	Venkataraman *et al.*, 2003; Venkataraman *et al.*, 2012
Genus *Stylocoeniella* Yabe & Sugiyama, 1935					
188.	*Stylocoeniella armata* (Ehrenberg, 1834)	x	x	x	Turner *et al.*, 2009
189.	*Stylocoeniella guentheri* (Bassett-Smith, 1890)	x	x	x	Venkataraman *et al.*, 2003; Turner *et al.*, 2009
Family AGARICIIDAE Grey, 1847					
Genus: *Agaricia* Lamarck, 1801					
190.	*Agaricia grahamae* Wells 1973	x	x	x	Mondal & Raghunathan 2012
Genus *Coeloseris* Vaughan, 1918					
191.	*Coeloseris mayeri* Vaughan, 1918	x	x	x	Tikader *et al.*, 1986
Genus *Gardineroseris* Scheer & Pillai, 1974					
192.	*Gardineroseris planulata* (Dana, 1846)	x	x	Pillai 1989	Venkataraman *et al.*, 2003; Turner *et al.*, 2009
Genus *Leptoseris* Milne Edwards & Haime, 1849					
193.	*Leptoseris explanata* Yabe & Sugiyama,1941	x	x	x	Turner *et al.*, 2009; Venkataraman *et al.*, 2012
194.	*Leptoseris foliosa* Dinesen, 1980	x	x	x	Mondal *et al.*, 2013

195.	*Leptoseris fragilis* Milne Edwards & Haime, 1849	x	x	x	Matthai 1924a
196.	*Leptoseris gardineri* Van der Horst, 1921	x	x	x	Mondal *et al.*, 2012
197.	*Leptoseris incrustans* (Quelch, 1886)	x	x	x	Turner *et al.*, 2009; Venkataraman *et al.*, 2012
198.	*Leptoseris hawaiiensis* Vaughan, 1907	x	x	x	Matthai 1924a; Venkataraman *et al.*, 2012
199.	*Leptoseris mycetoseroides* Wells, 1954	x	x	x	Turner *et al.*, 2009; Venkataraman *et al.*, 2012
200.	*Leptoseris papyracea* (Dana, 1846)	x	x	x	Matthai 1924a; Venkataraman *et al.*, 2012
201.	*Leptoseris scabra* Vaughan, 1907	x	x	Suresh 1991	Turner *et al.*, 2009; Venkataraman *et al.*, 2012
202.	*Leptoseris solida* (Quelch, 1886)	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012; Mondal *et al.*, 2013
203.	*Leptoseris striata* Fenner & Veron, 2000	x	x	x	Raghuranathan 2015; Mondal *et al.*, 2013, 2019
204.	*Leptoseris tubulifera* Vaughan, 1907	x	x	x	Mondal *et al.*, 2019
205.	*Leptoseris yabei* (Pillai & Scheer, 1976)	x	x	x	Turner *et al.*, 2009; Mondal *et al.*, 2013
Genus *Pavona* Lamarck, 1801					
206.	*Pavona bipartita* Nemenzo, 1980	x	x	x	Turner *et al.*,2009; Venkataraman *et al.*, 2012
207.	*Pavona cactus* (Forskål, 1775)	x	x	x	Matthai 1924, Pillai 1972
208.	*Pavona clavus* (Dana, 1846)	x	x	x	Tikader *et al.*, 1986; Venkataraman *et al.*, 2012
209.	*Pavona danai* Milne Edwards 1860	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2011
210.	*Pavona decussata* (Dana, 1846)	x	Pillai 1967	x	Matthai 1924
211.	*Pavona divaricata* Lamarck, 1816	x	Sukumaran *et al.*, 2007	x	Mondal *et al.*, 2013
212.	*Pavona diffluens* (Lamarck, 1816)	x	x	x	Mondal *et al.*, 2015; Raghunathan 2015
213.	*Pavona duerdeni* Vaughan, 1907	x	x	Pillai 1989	Tikader *et al.*, 1986
214.	*Pavona explanulata* (Lamarck, 1816)	x	x	Jeyabaskaran 2009	Pillai 1967
215.	*Pavona frondifera* (Lamarck, 1816)	x	x	x	Mondal *et al.*, 2012
216.	*Pavona gigantea* Verrill, 1896	x	x	x	Venkataraman *et al.*, 2012; Mondal & Raghunathan 2017; Mondal *et al.*, 2019
217.	*Pavona maldivensis* (Gardiner, 1905)	x	Pillai 1967	Pillai 1971	Mondal *et al.*, 2013
218.	*Pavona minuta* Wells, 1954	x	x	Suresh 1991	Mondal *et al.*, 2013

219.	*Pavona varians* Verrill, 1864)	x	Pillai 1967	Pillai 1967	Tikader *et al.*, 1986
220.	*Pavona venosa* (Ehrenberg, 1834)	x	Pillai 1967	Caeiro 1999	Reddiah 1977
221.	*Pavona xarifae* Scheer & Pillai, 1974	x	x	x	Tikader *et al.*, 1986
Family CARYOPHYLLIIDAE Dana. 1846					
Genus *Caryophyllia* Lamarck, 1801					
222.	*Caryophyllia* (*Caryophyllia*) *ambrosia* Alcock, 1898	x	Alcock (1898)	x	x
223.	*Caryophyllia* (*Caryophyllia*) *cintinculata* (Alcock, 1898)	x	x	x	Horst 1931, Pillai 1972
224.	*Caryophyllia* (*Caryophyllia*) *ephyla* Alcock, 1891	Alcock (1898) reported from Chennai coast			
225.	*Caryophyllia* (*Caryophyllia*) *gr&is* Gardiner & Waugh, 1938	x	x	Venkataraman 2007	Venkataraman 2007
226.	*Caryophyllia* (*Acanthocyathus*) *grayi* (Milne Edwards & Haime, 1848)	x	x	x	Alcock 1998; Venkataraman 2006
227.	*Caryophyllia* (*Caryophyllia*) *paradoxus* Alcock, 1898	Alcock (1898) reported from Kerala coast			
228.	*Caryophyllia* (*Caryophyllia*) *smithii* Stokes & Broderip, 1828	x	x	Alcock (1898)	Alcock (1898)
Genus *Desmophyllum* Ehrenberg, 1834					
229.	*Desmophyllum dianthus* (Esper, 1794)	x	x	x	Mondal *et al.*, 2017
Genus *Heterocyathus* Milne Edwards & Haime, 1848					
230.	*Heterocyathus aequicostatus* Milne Edwards & Haime, 1848	x	Venkataraman 2007	x	Alcock (1993); Pillai (1972)
231.	*Heterocyathus alternatus* Verrill, 1865	Venkataraman 2007 Chennai coast			
232.	*Heterocyathus sulcatus* (Verrill, 1866)	Venkataraman 2007 Chennai coast			
Genus *Paracyathus* Milne Edwards & Haime, 1848					
233.	*Paracyathus indicus* Duncan, 1899	x	x	x	Horst 1931; Pillai 1972
234.	*Paracyathus parvulus* Gardiner, 1899	x	Pillai 1967	x	x
235.	*Paracyathus profundus* Duncan, 1889	x	Pillai, 1986	x	x
236.	*Paracyathus pruinosus* Alcock, 1902	x	x	x	Raghuraman & Raghunathan 2015
237.	*Paracyathus stokesii* Milne Edwards & Haime, 1848	Pillai & Patel 1988	Venkataraman 2006	x	Venkataraman 2006; Mondal *et al.*, 2015
238.	*Paracyathus rotundatus* Semper, 1872	x	x	x	Mondal *et al.*, 2014
Genus *Polycyathus* Duncan, 1876					
239.	*Polycyathus verrilli* Duncan, 1889	Pillai & Patel 1988	Sukumaran *et al.*, 2007	x	Tikader *et al.*, 1986
240.	*Polycyathus andamanensis* Alcock, 1893	x	x	x	Alcock 1993, Pillai 1972
241.	*Polycyathus fuscomarginatus* (Klunzinger, 1879)	Venkataraman 2007 Chennai coast			
Genus *Solenosmilia* Duncan, 1873					
242.	*Solenosmilia variabilis* Duncan, 1873	Alcock (1898) from Kerala coast (as per pillai, 1967)			

Genus *Stephanocyathus* Seguenza, 1864					
243.	*Stephanocyathus* (*Odontocyathus*) *nobilis* (Moseley in Thompson, 1873)	x	Alcock (1898), also from off goa coast (as per pillai, 1967)	Venkataraman 2007	x
Genus *Trochocyanthus* Milne Edwards & Haime, 1848					
244.	*Trochocyanthus* sp.	x	x	Pillai 1967	x
Family COSCINARAEIDAE Benzoni, Arrigoni, Stefani & Stolarski, 2012					
Genus *Coscinaraea* Milne Edwards & Haime, 1848					
245.	*Coscinaraea columna* (Dana, 1846)	Satyanarayana & Ramakrishna 2009	x	*x*	Turner *et al.*, 2009
246.	*Coscinaraea crassa* Veron & Pichon,1980	x	Rajan & Venkataraman, 2010 Chennai coast	x	Turner *et al.*, 2009
247.	*Coscinaraea monile* (Forskål, 1775)	Pillai & Patel 1988	Pillai 1967	x	Mondal & Raghunathan 2017; Mondal *et al.*, 2015, 2019
248.	*Coscinaraea exesa* (Dana, 1846)	x	x	Pillai 1971	Mondal *et al.*, 2012
Family DELTOCYATHIDAE Cairns, Stolarski & Miller, 2012					
Genus *Deltocyathus* Milne Edwards & Haime, 1848					
249.	*Deltocyathus andamanicus* Alcock, 1898	x	x	x	Alcock (1898), Tikader *et al.*,1986
250.	*Deltocyathus rotulus* (Alcock, 1898)	Venkataraman 2007 Chennai coast			
Family DENDROPHYLLIIDAE Grey, 1847					
Genus *Balanophyllia* Wood, 1844					
251.	*Balanophyllia* (*Balanophyllia*) *cumingii* Milne Edwards & Haime, 1848	x	Pillai & Jasmine 1995 off Quilon	x	x
252.	*Balanophyllia* (*Balanophyllia*) *galapagensis* Vaughan, 1907	x	x	x	Mondal *et al.*, 2017
253.	*Balanophyllia* (*Eupsammia*) *imperialis* Kent, 1871	x	x	x	Venkataraman 2007
254.	*Balanophyllia* (*Balanophyllia*) *merguiensis* Duncan, 1889	x	x	x	Mondal *et al.*, 2014
255.	*Balanophyllia* (*Balanophyllia*) *parallela* Samper, 1889	x	x	x	Alcock 1893
256.	*Balanophyllia* (*Balanophyllia*) *scabra* Alock,1983	x	x	x	Alcock 1893, Pillai 1972
257.	*Balanophyllia* (*Eupsammia*) *stimpsonii* (Verrill, 1865)	x	Venkataraman 2007	x	Venkataraman 2007
258.	*Balanophyllia (Balanophyllia) vanderhorsti* Cairns, 2001	x	x	x	Mondal *et al.*, 2014
Genus *Cladopsammia* Lacaze-Duthiers, 1897					
259.	*Cladopsammia gracilis* (Milne Edwards & Haime, 1848)	x	Pillai 1986	x	x
260.	*Cladopsammia eguchii* (Wells, 1982)	x	x	x	Mondal *et al.*, 2017
Genus *Dendrophyllia* de Blainville, 1830					
261.	*Dendrophyllia arbuscula* van der Horst, 1922	x	x	x	Tikader *et al.*, 1986
262.	*Dendrophyllia cornigera* (Lamarck, 1816)	x	Pillai & Jasmine	x	x

			1995 off Quilon		
263.	*Dendrophyllia indica* Pillai, 1969	x	Pillai 1967c; Venkataraman *et al.,* 2002	x	x
264.	*Dendrophyllia minuscula* Bourne, 1905	Pillai & Patel 1988	Pillai & Jasmine 1995 off Quilon	x	Sudarshan & Mukhopadhyay 1967, Tikader *et al.,* 1986; Venkataraman *et al.,* 2012
265.	*Dendrophyllia robusta* (Bourne,1905)	x	Pillai 1967	x	Ramakrishna *et al.,* 2010; Venkataraman *et al.,* 2012
Genus *Enallopsammia* Sismonda, 1871					
266.	*Enallopsammia rostrata* (Pourtalès, 1878)	x	Venkataraman 2007	x	Pillai 1983
267.	*Enallopsammia pusilla* (Alcock, 1902)	x	x	x	Pillai 1983
Genus *Eguchipsammia* Cairns, 1994					
268.	*Eguchipsammia gaditana* (Duncan, 1873)	x	Venkataraman 2007	x	x
Genus *Endopachys* Milne Edwards & Haime, 1848					
269.	*Endopachys grayi* Milne Edwards & Haime, 1848	Pillai & Jasmine 1995 off Quilon			
Genus *Endopsammia* Milne Edwards & Haime, 1848					
270.	*Endopsammia philippensis* Milne Edwards & Haime, 1848	x	Venkataraman 2007	Pillai 1967	Venkataraman & Satyanarayana. 2012
Genus *Heteropsammia* Milne Edwards & Haime, 1848					
271.	*Heteropsammia cochlea* (Spengler, 1781)	x	Pillai & Jasmine 1995 off Quilon, Venkataraman 2007 Chennai coast	x	Tikader *et al.,* 1986
272.	*Heteropsammia eupsammides* (Gray, 1849)	x	x	x	Alcock 1893, Pillai 1972
Genus *Rhizopsammia* Verrill, 1869					
273.	*Rhizopsammia verrilli* van der Horst, 1922	x	x	x	Mondal *et al.,* 2012
Genus *Tubastraea* Lesson, 1829					
274.	*Tubastraea coccinea* Lesson, 1829	Pillai & Patel 1988	x	Pillai 1989	Tikader *et al.,* 1986
275.	*Tubastraea diaphana* Dana,1846	x	x	X	Venkataraman *et al.,* 2003; Ramakrishna *et al.,* 2010; Mondal & Raghunathan 2017
276.	*Tubastraea faulkneri* Wells, 1982	Singh *et al.,* 2003	x	x	Ramakrishna *et al.,* 2010; Raghunathan *et al.,* 2015
277.	*Tubastraea micranthus* (Ehrenberg, 1834)	Singh *et al.,* 2003	x	Suresh 1991	Tikader *et al.,* 1986; Venkataraman *et al.,* 2003
Genus *Turbinaria* J. V. Lamouroux, 1825					

278.	*Turbinaria crater* (Pallas, 1766) (*nomen dubium*)*	Pillai & Patel 1988	Pillai 1967	Pillai 1989	Tikader *et al.*, 1986
279.	*Turbinaria frondens* (Dana, 1846)*	Satyanarayana & Ramakrishna 2009	Venkataraman & Rajan, 2013	Caeiro 1999	Mondal *et al.*, 2015
280.	*Turbinaria irregularis* Bernard, 1896	x	x	x	Mondal *et al.*, 2015
281.	*Turbinaria mesenterina* (Lamarck, 1816)	Kumar *et al.*, 2014	Pillai 1967	Pillai 1989	x
282.	*Turbinaria mollis* Bernard, 1896 taxon inquirendum	Raghuraman *et al.*, 2013 (location was not mentioned)			
283.	*Turbinaria patula* (Dana, 1846)	x	Geetha & Yogesh Kumar, 2012; Mathews *et al.*, 2017	x	x
284.	*Turbinaria peltata* (Esper, 1794)	Pillai & Patel 1988	Pillai 1967	x	Tikader *et al.*, 1986
285.	*Turbinaria quincuncialis* Ortmann, 1889 (*nomen dubium*)	x	Pillai 1967	x	x
286.	*Turbinaria radicalis* Bernard, 1896	x	x	x	Mondal *et al.*, 2013
287.	*Turbinaria reniformis* Bernard, 1896	Kumar *et al.*, 2014	x	x	Tikader *et al.*, 1986
288.	*Turbinaria stellulata* (Lamarck, 1816)	x	x	x	Turner *et al.*,2009; Venkataraman *et al.*, 2003
289.	*Turbinaria undata* Bernard, 1896 (*nomen dubium*)	Pillai & Patel 1988	x	x	Mondal *et al.*, 2012
Family DIPLOASTREIDAE Chevalier & Beauvais, 1987					
Genus *Diploastrea* Matthai, 1914					
290.	*Diploastrea heliopora* (Lamarck,1816)	Singh *et al.*, 2003	x	Pillai 1971	Reddiah 1977
Family EUPHYLLIDAE Veron, 2000					
Genus *Euphyllia* Dana, 1846					
291.	*Euphyllia ancora* Veron & Pichon, 1980	x	x	x	Turner *et al.*, 2009
292.	*Euphyllia cristata* Chevalier, 1971	x	x	x	Mondal *et al.*, 2012
293.	*Euphyllia divisa* Veron & Pichon, 1979	x	x	x	Turner *et al.*, 2009
294.	*Euphyllia glabrescens* (Chamisso & Eysenhardt, 1821)	x	x	Pillai 1967	Reddiah 1977
295.	*Euphyllia paraglabrescens* Veron, 2000	x	x	x	Mondal *et al.*, 2015; Raghuraman *et al.*, 2015
296.	*Euphyllia yaeyamaensis* (Sirai, 1980)	x	x	x	Turner *et al.*, 2009
Genus *Catalaphyllia* Wells, 1971					
297.	*Catalaphyllia jardinei* (Saville-Kent, 1893)	x	x	x	Mondal *et al.*, 2015; Raghuraman *et al.*, 2015
Family FLABELLIDAE Bourne, 1905					
Genus *Flabellum* Lesson, 1831					
298.	*Flabellum* (*Flabellum*) *pavoninum* Lesson, 1831	Alcock 189) as *F. paripavonium* as of Pillai 1967, Venkataraman 2007			
299.	*Flabellum* (*Ulocyathus*) *deludens* Marenzeller, 1904	Alcock 1898 as *F. lacinatum from the Bay of Bengal* as of Pillai 1967, Venkataraman 2007			

300.	*Flabellum* (*Ulocyathus*) *japonicum* Mosley, 1881	Alcock 1898 from Indian seas (as of Pillai, 1967)			
Genus *Javania* Duncan, 1876					
301.	*Javania cailleti* (Duchassaing & Michelotti, 1864)	Alcock 1898 Kerala coast & Alcock 1893 off Konkon coast, as of Pillai 1967			
Genus *Placotrochus* Milne Edwards & Haime, 1848					
302.	*Placotrochus laevis* Milne Edwards & Haime, 1848	x	Venkataraman 2007	x	Venkataraman 2007
Genus *Truncatoflabellum* Cairns, 1989					
303.	*Truncatoflabellum crassum* (Milne Edwards & Haime 1848)	x	Bourne 1905	x	x
304.	*Truncatoflabellum paripavoninum* (Alcock, 1894)	x	x	Venkataraman 2007	Venkataraman 2007
305.	*Truncatoflabellum spheniscus* (Dana, 1846)	x	x	x	Mondal *et al.*, 2017
306.	*Truncatoflabellum stokesii* (Milne Edwards & Haime, 1848)	x	Pillai & Jasmine 1995 off Quilon	x	Venkataraman 2007
Genus *Rhizotrochus* Milne Edwards & Haime, 1848					
307.	*Rhizotrochus typus* Milne Edwards & Haime, 1848	Venkataraman 2007 Bay of Bengal			
Family FUNGIACYATHIDAE Chavalier, 1987					
Genus *Fungiacyathus* Sars, 1872					
308.	*Fungiacyathus* (*Bathyactis*) *symmetricus* (Pourtalès, 1871)	x	x	x	Alcock 1898 as of Pillai 1972
309.	*Fungiacyathus* (*Fungiacyathus*) *stephanus* (Alcock, 1893)	x	x	Venkataraman 2007	x
Family FUNGIIDAE Dana, 1846					
Genus *Cantharellus* Hoeksema & Best, 1984					
310.	*Cantharellus doederleini* (von Marenzeller, 1907)	x	x	x	Mondal *et al.*, 2013; Raghunathan *et al.*, 2015
311.	*Cantharellus jebbi* Hoeksema, 1993	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2011; Mondal & Raghunatha 2017
Genus *Ctenactis* Verrill, 1864					
312.	*Ctenactis albitentaculata* (Hoeksema, 1989)	x	x	x	Ramakrishna *et al.*, 2010; Raghuraman *et al.*, 2012; 2015
313.	*Ctenactis crassa* (Dana, 1846)	x	x	x	Tikader *et al.*, 1986; Ramakrishna *et al.*, 2010
314.	*Ctenactis echinata* (Pallas, 1766)	x	x	x	Pillai 1967
315.	*Ctenactis triangularis* Mondal & Raghunathan, 2013 (taxon inquirendum)	x	x	x	Mondal & Raghunathan, 2013
Genus *Cycloseris* Milne Edwards & Haime, 1849					
316.	*Cycloseris cyclolites* (Lamarck, 1801)	x	Pillai 1967 Tuticorin, Kumar *et al.*, 2019	Jeyabaskaran 2009	Tikader *et al.*, 1986; Ramakrishna *et al.*, 2010

317.	*Cycloseris costulata* (Ortmann, 1889)	x	x	Jeyabaskaran 2009	Tikader *et al.*, 1986; Ramakrishna *et al.*, 2010
318.	*Cycloseris curvata* (Hoeksema, 1989)	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2019
319.	*Cycloseris distorta* (Michelin, 1842)	x	x	x	Alcock 1893; Mondal *et al.*, 2019
320.	*Cycloseris explanulata* (Van der Horst, 1922)	x	x	x	Turner *et al.*, 2009
321.	*Cycloseris hexagonalis* (Milne Edwards & Haime, 1848)	x	x	x	Alcock 1893
322.	*Cycloseris fragilis* (Alcock, 1893)	x	x	x	Alcock 1893; Ramakrishna *et al.*, 2010
323.	*Cycloseris sinensis* Milne Edwards & Haime, 1851	x	x	x	Alcock 1893; Ramakrishna *et al.*, 2010
324.	*Cycloseris somervillei* (Gardiner, 1909)	x	x	Pillai 1971	Pillai 1972; Mondal *et al.*, 2019
325.	*Cycloseris tenuis* (Dana, 1846)	x	x	Jeyabaskaran 2009	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2015
326.	*Cycloseris vaughani* (Boschman, 1923)	x	x	x	Mondal *et al.*, 2015, 2019
327.	*Cycloseris wellsi* Veron & Pichon, 1980	x	x	x	Mondal *et al.*, 2015; Mondal & Raghunathan 2017
Genus *Danafungia* Wells, 1966					
328.	*Danafungia scruposa* (Klunzinger, 1879)	x	x	Caeiro 1999	Matthai 1924; Venkataraman *et al.*, 2012
329.	*Danafungia horrida* (Dana, 1846)	x	x	Pillai 1971	Matthai 1924; Turner *et al.*, 2009
Genus *Fungia* Lamarck, 1801					
330.	*Fungia fungites* (Linnaeus, 1758)	x	x	Pillai 1967	Pillai 1972
Genus *Halomitra* Dana, 1846					
331.	*Halomitra pileus* (Linnaeus, 1758)	x	x	x	Venkataraman *et al.*, 2003; Ramakrishna *et al.*, 2010
Genus *Heliofungia* Wells, 1966					
332.	*Heliofungia fralinae* (Nemenzo, 1955)	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012; Raghuraman *et al.*, 2013
Genus *Herpolitha* Eschscholtz, 1825					
333.	*Herpolitha limax* (Esper, 1797)	x	x	Suresh 1991	Pillai 1972; Rajan *et al.*, 2010
Genus *Lithophyllon* Rehberg, 1892					
334.	*Lithophyllon concinna* (Verrill, 1864)	x	x	Caeiro 1999	Matthai 1924
335.	*Lithophyllon repanda* (Dana, 1846)	x	x	Sreenath *et al.*, 2015	Tikader *et al.*, 1986
336.	*Lithophyllon scabra* (Döderlein, 1901)	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2011, 2015

337.	*Lithophyllon spinifer* (Claereboudt & Hoeksema, 1987)	x	x	x	Ramakrishna *et al.*, 2010; Venkataranman *et al.*, 2012; Raghuraman *et al.*, 2013
338.	*Lithophyllon undulatum* Rehberg, 1892	x	x	x	Turner *et al.*, 2009
Genus *Lobactis* Verrill, 1864					
339.	*Lobactis scutaria* (Lamarck, 1801)	x	x	Pillai 1967	Tikader *et al.*,1986
Genus *Pleuractis* Verrill, 1864					
340.	*Pleuractis granulosa* (Klunzinger, 1879)	x	x	Jeyabaskaran 2009	Turner *et al.*, 2009; Rajan *et al* 2010
341.	*Pleuractis moluccensis* (Van der Horst, 1919)	x	x	x	Turner *et al.*, 2009
342.	*Pleuractis paumotensis* (Stutchbury, 1833)	x	x	x	Mathai 1924; Venkataraman *et al.*, 2012
343.	*Pleuractis seychellensis* (Hoeksema, 1993)	x	x	Jeyabaskaran 2009	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
Genus *Podabacia* Milne Edwards & Haime, 1849					
344.	*Podabacia crustacea* (Pallas, 1766)	x	x	Pillai 1967	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
345.	*Podabacia lankaensis* Veron, 2000	x	x	x	Turner *et al.*, 2009; Ramakrishna *et al.*, 2010
346.	*Podabacia motuporensis* Veron, 1990	x	x	x	Mondal *et al.*, 2012
347.	*Podabacia sinai* Veron, 2000	x	x	x	Mondal *et al.*, 2011
Genus *Polyphyllia* Blainville, 1830					
348.	*Polyphyllia talpina* (Lamarck, 1801)	x	x	Pillai 1989	Tikader *et al.*, 1986
Genus *Sandalolitha* Quelch, 1884					
349.	*Sandalolitha dentata* Quelch, 1884	x	x	x	Turner *et al.*, 2009
350.	*Sandalolitha robusta* (Quelch, 1886)	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
Genus *Zoopilus* Dana, 1846					
351.	*Zoopilus echinatus* Dana, 1846	x	x	x	Mondal *et al.*, 2012
Family LOBOPHYLLIIDAE Dai & Horng, 2009					
Genus *Acanthastrea* Milne Edwards & Haime, 1848					
352.	*Acanthastrea brevis* Milne Edwards & Haime, 1849	x	x	x	Mondal *et al.*, 2013, 2015; Raghunathan 2015; Mondal & Raghuraman *et al.*, 2017
353.	*Acanthastrea echinata* (Dana, 1846)*	Singh *et al.*, 2003	Edward *et al.*, 2007	Pillai 1971	Turner *et al.*, 2009; Venkataraman *et al.*, 2012
354.	*Acanthastrea hemprichii* (Ehrenberg, 1834)	x	x	x	Turner *et al.*, 2009; Venkataraman *et al.*, 2012
355.	*Acanthastrea pachysepta* (Chevalier, 1975)	x	x	x	Mondal *et al.*, 2013, 2015; Raghunathan 2015

356.	*Acanthastrea simplex* (Crossland, 1848) (*nomen dubium*)	Pillai & Patel 1988	x	x	x
Genus *Cynarina* Brüggemann, 1877					
357.	*Cynarina lacrymalis* (Milne Edwards & Haime, 1849)	x	x	x	Turner *et al.*, 2009
Genus *Echinomorpha* Veron, 2000					
358.	*Echinomorpha nishihirai* (Veron, 1990)	x	x	x	Mondal *et al.*, 2015; Raghunathan 2015
Genus *Echinophyllia* Klunzinger, 1879					
359.	*Echinophyllia aspera* (Ellis & Solander, 1786)	Satyanarayana & Ramakrishna 2009	x	x	Reddiah 1977
360.	*Echinophyllia echinata* (Saville-Kent, 1871)	x	x	x	Turner *et al.*, 2009
361.	*Echinophyllia echinoporoides* Veron & Pichon,1980	x	x	x	Turner *et al.*, 2009
362.	*Echinophyllia orpheensis* Veron & Pichon, 1980	x	x	x	Mondal *et al.*, 2013, 2015; Mondal & Raghunathan 2017
Genus *Homophyllia* Brüggemann, 1877					
363.	*Homophyllia australis* (Milne Edwards & Haime, 1849)	x	x	x	Mondal *et al.*, 2011
364.	*Homophyllia bowerbanki* (Milne Edwards, 1857)	Kumar *et al.*, 2014	x	x	Venkataraman *et al.*, 2012; Mondal *et al.*, 2019
Genus *Lobophyllia* de Blainville, 1830					
365.	*Lobophyllia agaricia* (Milne Edwards & Haime, 1849)	x	x	x	Venkataraman *et al.*, 2003, 2012
366.	*Lobophyllia diminuta* Veron, 1985	x	x	x	Mondal *et al.*, 2011
367.	*Lobophyllia corymbosa* (Forskål, 1775)	x	Edward *et al.*, 2007	Pillai 1971	Matthai 1924, Pillai 1972
368.	*Lobophyllia dentata* Veron, 2000	x	x	x	Mondal et aḷ. 2013
369.	*Lobophyllia erythraea* (Klunzinger, 1879)	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2011, 2015, 2019
370.	*Lobophyllia flabelliformis* Veron, 2000	x	x	x	Mondal *et al.*, 2015; Mondal & Raghunathan 2017
371.	*Lobophyllia hassi* (Pillai & Scheer, 1976)	x	x	x	Mondal *et al.*, 2015, 2017; Raghunathan 2015
372.	*Lobophyllia hemprichii* (Ehrenberg, 1834)	Kumar *et al.*, 2017	x	Suresh 1991	Reddiah 1977
373.	*Lobophyllia ishigakiensis* (Veron, 1990)	x	x	x	Turner *et al.*, 2009
374.	*Lobophyllia radians* (Milne Edwards & Haime, 1849)*	Pillai & Patel 1988	Matthai 1924	Pillai 1971	Tikader *et al.*, 1986
375.	*Lobophyllia recta* (Dana, 1846)*	Singh *et al.*, 2003	Matthai 1924, Venkataraman & Rajan, 2013	Pillai 1971	Matthai 1924, Pillai 1972
376.	*Lobophyllia robusta* Yabe & Sugiyama,1936	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012

377.	*Lobophyllia rowleyensis* (Veron, 1985)	x	x	x	Turner *et al.*, 2009
378.	*Lobophyllia serrata* Veron, 2000	x	x	Jeyabaskaran 2009	x
379.	*Lobophyllia valenciennesii* (Milne Edwards & Haime, 1849)	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012; Mondal & Raghunathan 2017
380.	*Lobophyllia vitiensis* (Brüggemann, 1877)	x	x	x	Venkataraman *et al.*, 2012; Mondal *et al.*, 2013, 2015
Genus *Micromussa* Veron, 2000					
381.	*Micromussa regularis* (Veron, 2000)	x	x	x	Mondal *et al.*, 2011
Genus *Oxypora* Saville-Kent, 1871					
382.	*Oxypora crassispinosa* Nemenzo, 1979	x	x	x	Turner *et al.*, 2009; Venkataraman *et al.*, 2012
383.	*Oxypora glabra* Nemenzo, 1959	x	x	x	Venkataraman *et al.*, 2012; Mondal & Raghunathan 2017; Mondal *et al.*, 2013, 2019
384.	*Oxypora lacera* (Verrill, 1864)	x	x	x	Turner *et al.*, 2009; Venkataraman *et al.*, 2012
Genus *Sclerophyllia* Klunzinger, 1879					
385.	*Sclerophyllia maxima* (Sheppard & Salm, 1988)	x	x	x	Mondal *et al.*, 2015
Family MERULINIDAE Verrill, 1865					
Genus *Astrea* Lamarck, 1801					
386.	*Astrea annuligera* Milne Edwards & Haime, 1849	Singh *et al.*, 2003	x	x	Mondal *et al.*, 2013
387.	*Astrea curta* Dana, 1846	x	x	Caeiro 1999	Turner *et al.*, 2009
Genus *Caulastraea* Dana, 1846					
388.	*Caulastraea curvata* Wijsman-Best, 1972	x	x	x	Mondal *et al.*, 2013
389.	*Caulastraea echinulata* (Milne Edwards & Haime, 1849)	x	x	x	Mondal *et al.*, 2012
390.	*Caulastraea furcata* Dana, 1846	x	x	x	Mondal *et al.*, 2015
Genus *Coelastrea* Verrill, 1866					
391.	*Coelastrea aspera* (Verrill, 1866)	x	Pillai 1967	Caeiro 1999	Mondal *et al.*, 2013, 2015
392.	*Coelastrea palauensis* (Yabe & Sugiyama, 1936)	x	x	x	Mondal *et al.*, 2012
Genus *Cyphastrea* Milne Edwards & Haime, 1848					
393.	*Cyphastrea agassizi* (Vaughan, 1907)	x	x	x	Mondal *et al.*, 2013
394.	*Cyphastrea chalcidicum* (Forskal, 1775)	x	Pillai 1967	Suresh 1991	Mondal *et al.*, 2015; Raghunathan 2015
395.	*Cyphastrea japonica* Yabe & Sugiyama, 1932	x	Edward *et al.*, 2007	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2015, 2019; Raghunathan 2015
396.	*Cyphastrea microphthalma* (Lamarck, 1816)	x	Pillai 1967, Venkataram	Pillai 1989	Tikader *et al.*, 1986

			an & Rajan 2013		
397.	*Cyphastrea ocellina* (Dana,1864)	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2015; Raghunathan 2015
398.	*Cyphastrea serailia* (Forskål, 1775)*	Pillai & Patel 1988	Pillai 1967	Pillai 1989	Mondal *et al.*, 2015, 2019; Raghunathan 2015
Genus *Dipsastraea* Blainville, 1830					
399.	*Dipsastraea amicorum* (Milne Edwards & Haime, 1849)	Satyanarayana & Ramakrishna 2009	x	x	Mondal *et al.*, 2011, 2013, 2015, 2019
400.	*Dipsastraea albida* (Veron 2000)	x	Venkataraman & Rajan 2013	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012; Mondal *et al.*, 2015
401.	*Dipsastraea danai* (Milne Edwards, 1857)	x	x	x	Mondal *et al.*, 2015, 2019
402.	*Dipsastraea faviaformis* (Veron, 2000)	x	x	x	Mondal *et al.*, 2015, 2019; Raghunathan 2015
403.	*Dipsastraea favus* (Forskål, 1775)*	Pillai & Patel 1988	Pillai 1967	Pillai 1971	Pillai 1972
404.	*Dipsastraea helianthoides* (Wells, 1954)	x	x	x	Mondal *et al.*, 2011
405.	*Dipsastraea lacuna* (Veron, Turak & DeVantier, 2000)	Satyanarayana & Ramakrishna 2009	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
406.	*Dipsastraea laddi* (Wells, 1954)	x	x	x	Mondal *et al.*, 2013, 2015, 2019; Raghunathan 2015
407.	*Dipsastraea laxa* (Klunzinger, 1879)	x	x	x	Mondal *et al.*, 2015, 2019; Raghunathan 2015
408.	*Dipsastraea lizardensis* (Veron, Pichon & Wijsman-Best, 1977)	x	Venkataraman & Rajan 2013	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012; Mondal *et al.*, 2015, 2019
409.	*Dipsastraea marshae* (Veron, 2000)	x	x	x	Mondal *et al.*, 2013
410.	*Dipsastraea matthaii* (Vaughan, 1918)	x	Krishnan *et al.*, 2018	x	Mondal *et al.*, 2011, 2013, 2019
411.	*Dipsastraea maxima* (Veron, Pichon & Wijsman-Best, 1977)	Singh *et al.*, 2003	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012; Mondal & Raghunathan 2017, Mondal *et al.*, 2019
412.	*Dipsastraea pallida* (Dana, 1846)*	Pillai *et al.*, 1979	Pillai 1967	Pillai 1971	Matthai 1924; Tikader *et al.*, 1986; Venkataraman *et al.*, 2012
413.	*Dipsastraea speciosa* (Dana, 1846)*	Pillai & Patel 1988	Pillai 1967	Pillai 1971	Reddiah 1977; Tikader *et al.*, 1986; Venkataraman *et al.*, 2012
414.	*Dipsastraea rotumana* (Gardiner, 1899)	x	Venkataraman & Rajan 2013	x	Tikader *et al.*, 1986; Venkataraman *et al.*, 2012

415.	*Dipsastraea rosaria* (Veron, 2000)	x	Krishnan *et al.*, 2018	x	Mondal *et al.*, 2012
416.	*Dipsastraea truncata* (Veron, 2000)	x	x	x	Turner *et al.*, 2009; Venkataraman *et al.*, 2012
417.	*Dipsastraea veroni* (Moll & Best, 1984)	x	Krishnan *et al.*, 2018	x	Mondal *et al.*, 2013, 2015
418.	*Dipsastraea vietnamensis* (Veron, 2000)	x	x	x	Mondal *et al.*, 2015
Genus *Echinopora* Lamarck, 1816					
419.	*Echinopora forskaliana* (Milne Edwards & Haime, 1849)	x	x	x	Mondal *et al.*, 2015; Raghunathan 2015
420.	*Echinopora fruticulosa* Klunzinger, 1879	x	x	x	Venkataram *et al.*, 2012; Mondal *et al.*, 2013, 2015; Raghunathan 2015
421.	*Echinopora gemmacea* (Lamarck, 1816)	x	Pillai 1967	x	Turner *et al.*, 2009; Venkataram *et al.*, 2012
422.	*Echinopora horrida* Dana, 1846	x	x	x	Tikader *et al.*, 1986; Venkataram *et al.*, 2012
423.	*Echinopora hirsutissima* Milne Edwards & Haime, 1849	x	x	x	Turner *et al.*, 2009; Venkataram *et al.*, 2012
424.	*Echinopora lamellosa* (Esper, 1795)	x	Pillai 1967	Pillai 1989	Matthai 1924; Venkataram *et al.*, 2012
425.	*Echinopora pacificus* Veron, 1990	x	x	x	Mondal *et al.*, 2014, 2015; Mondal & Raghunathan 2017
Genus *Erythrastrea* Pichon, Scheer & Pillai, 1983					
426.	*Erythrastrea flabellata* Pichon, Scheer & Pillai, 1983	DeVantier *et al.*, 2008	x	x	x
Genus *Favites* Link, 1807					
427.	*Favites abdita* (Ellis & Solander 1786)*	Singh *et al.*, 2003	Pillai 1967	Pillai 1967	Pillai 1972; Venkataraman *et al.*, 2012
428.	*Favites acuticollis* (Ortmann,1889)	x	x	x	Turner *et al.*, 2009; Venkataraman *et al.*, 2012
429.	*Favites chinensis* (Verrill, 1866)	Satyanarayana & Ramakrishna 2009	Krishnan *et al.*, 2018	x	Venkataraman *et al.*, 2012; Raghunathan 2015, 2019
430.	*Favites colemani* (Veron, 2000)	x	Venkataraman & Rajan 2013	x	Turner *et al.*, 2009
431.	*Favites complanata* (Eherenberg, 1834)*	Pillai & Patel 1988	Venkataraman & Rajan 2013	Pillai 1989	Tikader *et al.*, 1986; Venkataraman *et al.*, 2012
432.	*Favites flexuosa* (Dana, 1846)*	Satyanarayana & Ramakrishna 2009	Krishnan *et al.*, 2018	Pillai 1989	Tikader *et al.*, 1986; Venkataraman *et al.*, 2012
433.	*Favites halicora* (Ehrenberg, 1834)*	Satyanarayana & Ramakrishna 2009	Pillai 1967	Pillai 1971	Pillai 1972; Venkataraman *et al.*, 2012
434.	*Favites magnistellata* (Chevalier, 1971)	Pillai & Patel 1988	x	Caeiro 1999	Mondal *et al.*, 2013
435.	*Favites melicerum* (Ehrenberg, 1834)*	Pillai & Patel 1988	Pillai 1967	Pillai 1971	Mondal *et al.*, 2013

436.	*Favites micropentagona* Veron, 2002	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
437.	*Favites monticularis* Mondal, Raghunathan & Venkataraman, 2013	x	x	x	Mondal *et al.*, 2013
438.	*Favites paraflexuosus* Veron, 2000	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2010a
439.	*Favites pentagona* (Esper, 1795)*	Satyanarayana & Ramakrishna 2009	Pillai 1967	Pillai 1971	Turner *et al.*, 2009
440.	*Favites rotundata* Veron, Pichon & Wijsman-Best, 1977	x	x	x	Turner *et al.*, 2009
441.	*Favites spinosa* (Klunzinger, 1879)	x	Krishnan *et al.*, 2018	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2010a; Venkataraman *et al.*, 2012
442.	*Favites stylifera* Yabe & Sugiyama, 1937	x	x	Pillai 1989	Mondal *et al.*, 2015; Mondal & Raghunathan 2017
443.	*Favites vasta* (Klunzinger, 1879)	x	Venkataraman & Rajan 2013	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2015, 2019
444.	*Favites valenciennesi* (Milne Edwards & Haime, 1849)	x	Pillai 1972, Venkataraman & Rajan 2013	Pillai 1989)	Pillai 1972
Genus *Goniastrea* Milne Edwards & Haime, 1848					
445.	*Goniastrea edwardsi* Chevalier,1971	x	Venkataraman & Rajan 2013	Caeiro 1999	Venkataraman *et al.*, 2003; Turner *et al.*, 2009
446.	*Goniastrea favulus* (Dana, 1846)	x	x	x	Mondal *et al.*, 2015, 2019; Mondal & Raghunathan 2017
447.	*Goniastrea minuta* Veron, 2000	x	Krishnan *et al.*, 2018	x	Turner *et al.*, 2009; Venkataraman, 2012
448.	*Goniastrea retiformis* (Lamarck, 1816)	x	Pillai 1967	Pillai 1971	Reddiah 1977; Venkataraman, 2012
449.	*Goniastrea pectinata* (Ehrenberg, 1834)	Pillai & Patel 1988	Pillai 1967	x	Pillai 1967, Reddiah 1977
450.	*Goniastrea stelligera* (Dana, 1846)*	Venkataraman *et al.*, 2004	Pillai 1967	Pillai 1971	Tikader *et al.*, 1986
Genus *Hydnophora* Fischer von Waldheim, 1807					
451.	*Hydnophora bonsai* Veron, 1990	x	x	x	Mondal *et al.*, 2012
452.	*Hydnophora exesa* (Pallas, 1766)*	Pillai & Patel 1988	Pillai 1967	Suresh 1991	Mathai 1924
453.	*Hydnophora grandis* Gardiner, 1904	x	Pillai 1967	x	Rajan *et al.*, 2010
454.	*Hydnophora microconos* (Lamarck, 1816)	x	Pillai 1967	Pillai 1967	Tikader *et al.*,1986
455.	*Hydnophora pilosa* Veron, 1985	Sreenath 2015	x	x	Turner *et al.*, 2009
456.	*Hydnophora rigida* (Dana, 1846)	x	x	x	Reddiah 1977
Genus *Leptoria* Milne Edwards & Haime, 1848					
457.	*Leptoria irregularis* Veron, 1990	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2013, 2019

458.	*Leptoria phrygia* (Ellis & Solander, 1786)	x	Pillai 1967	Pillai 1971	Tikader *et al.*,1986
Genus *Merulina* Ehrenberg, 1834					
459.	*Merulina ampliata* (Ellis & Solander, 1786)	x	Pillai 1967	Pillai 1971	Pillai 1972
460.	*Merulina scabricula* Dana, 1846	x	x	x	Turner *et al.*, 2009
Genus *Mycedium* Milne Edwards & Haime, 1851					
461.	*Mycedium elephantotus* (Pallas, 1766)	Pillai & Patel 1988	Pillai 1967	x	Tikader *et al.*, 1986
462.	*Mycedium robokaki* Moll & Borel-Best, 1984	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012; Mondal & Raghunathan 2017
Genus *Oulophyllia* Milne Edwards & Haime, 1848					
463.	*Oulophyllia bennettae* (Veron, Pichon, & Best, 1977)	x	x	Jeyabaskaran 2009	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
464.	*Oulophyllia crispa* (Lamarck, 1816)	x	Krishnan *et al.*, 2018	Suresh 1991	Venkataraman *et al.*, 2003, 2012; Mondal *et al.*, 2015
465.	*Oulophyllia levis* (Nemenzo, 1959)	x	x	x	Venkataraman *et al.*, 2012; Mondal *et al.*, 2015, 2017
Genus *Paragoniastrea* Huang, Benzoni & Budd, 2014					
466.	*Paragoniastrea australensis* (Milne Edwards, 1857)	x	x	Pillai 1989	Reddiah 1977
467.	*Paragoniastrea russelli* (Wells, 1954)	x	Venkataraman & Rajan 2013	Caeiro 1999	Mondal *et al.*, 2013; Raghunathan 2015
Genus *Paramontastraea* Huang & Budd, 2014					
468.	*Paramontastraea peresi* (Faure & Pichon, 1978)	x	Krishnan *et al.*, 2018	x	Venkataraman *et al.*, 2012; Mondal & Raghunathan 2017
469.	*Paramontastraea salebrosa* (Nemenzo, 1959)	x	x	x	Mondal *et al.*, 2013
Genus *Pectinia* Blainville, 1825					
470.	*Pectinia alcicornis* (Saville-Kent, 1871)	x	x	x	Turner *et al.*, 2009
471.	*Pectinia lactuca* (Pallas, 1766)	x	x	Jeyabaskaran 2009	Tikader *et al.*,1986
472.	*Pectinia paeonia* (Dana, 1846)	x	x	x	Venkataraman *et al.*, 2003
473.	*Pectinia teres* Nemenzo & Montecillo, 1981	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012; Mondal *et al.*, 2019
Genus *Platygyra* Ehrenberg, 1834					
474.	*Platygyra acuta* Veron, 2000	x	x	x	Turner *et al.*, 2009
475.	*Platygyra carnosus* Veron, 2000	x	x	x	Mondal *et al.*, 2011; Mondal *et al.*, 2019
476.	*Platygyra crosslandi* (Matthai, 1928)	x	x	x	Ramakrishna *et al.*, 2010; Mondal & Raghunathan 2017; Mondal *et al.*, 2019
477.	*Platygyra contorta* Veron, 1990	x	x	x	Mondal *et al.*, 2012

478.	*Platygyra daedalea* (Ellis & Solander, 1786)*	Singh *et al.*, 2003	Venkataraman & Rajan 2013	Pillai 1989	Reddiah 1977
479.	*Platygyra lamellina* (Ehrenberg, 1834)*	Singh *et al.*, 2003	Matthai 1924	Pillai 1967	Pillai 1967
480.	*Platygyra pini* Chevalier, 1975	Satyanarayana & Ramakrishna 2009	x	Jeyabaskaran 2009	Turner *et al.*, 2009
481.	*Platygyra ryukyuensis* Yabe & Sugiyama, 1936	x	x	x	Ramakrishna *et al.*, 2010; Mondal & Raghunathan 2017
482.	*Platygyra sinensis* (Milne Edwards & Haime,1849)*	Pillai & Patel 1988	Venkataraman & Rajan 2013	Pillai 1989	Reddiah 1977
483.	*Platygyra verweyi* Wijsman-Best, 1976	x	Krishnan *et al.*, 2018	x	Turner *et al.*, 2009
484.	*Platygyra yaeyamaensis* (Eguchi & Shirai, 1977)	x	x	x	Mondal *et al.*, 2012
Genus *Scapophyllia* Milne Edwards & Haime, 1848					
485.	*Scapophyllia cylindrica* Milne Edwards & Haime, 1849	x	x	x	Tikader *et al.*,1986
Genus *Trachyphyllia* Milne Edwards & Haime, 1849					
486.	*Trachyphyllia geoffroyi* (Audouin, 1826)	x	x	x	Tikader *et al.*,1986
Family MUSSIDAE Ortmann, 1890					
Genus *Isophyllia* Milne Edwards & Haime, 1851					
487.	*Isophyllia sinuosa* (Ellis & Solander, 1786)	x	x	x	Mondal & Raghunathan 2016
488.	*Isophyllia rigida* (Dana, 1846)	x	x	x	Mondal *et al.*, 2012
Family OCULINIDAE Grey, 1847					
Genus *Galaxea* Oken, 1815					
489.	*Galaxea astreata* (Lamarck, 1816)	x	Pillai 1967	Caeiro 1999	Pillai 1967
490.	*Galaxea acrhelia* Veron, 2000	x	x	x	Turner *et al.*, 2009
491.	*Galaxea cryptoramosa* Fenner & Veron, 2000	x	x	x	Ramakrishna *et al.*, 2010; Mondal & Raghunathan 2011; Mondal *et al.*, 2019
492.	*Galaxea fascicularis* (Linnaeus, 1767)	x	Pillai 1967	Pillai 1967	Matthai 1924; Pillai 1972
Genus *Madrepora* Linnaeus, 1758					
493.	*Madrepora oculata* (Linnaeus, 1758)	Alcock (1898) from Kerala coast, Alcock (1893) off Konkan coast (as per Pillai 1967; Venkataraman 2007			
Family POCILLOPORIDAE Grey, 1842					
Genus *Pocillopora* Lamarck, 1816					
494.	*Pocillopora ankeli* Scheer & Pillai, 1974	x	x	x	Tikader *et al.*, 1986
495.	*Pocillopora brevicornis* Lamarck, 1816	x	x	x	Pillai 1967
496.	*Pocillopora damicornis* (Linnaeus, 1758)*	Satyanarayana & Ramakrishna 2009	Krishnan *et al.*, 2018, Gravely 1941 Chennai coast	Pillai 1967	Pillai 1967
497.	*Pocillopora elegans* Dana, 1846	x	Pillai 1967	x	Reddiah 1977
498.	*Pocillopora grandis* Dana, 1846	x	Pillai 1967	Pillai 1971	Mondal & Raghunathan 2017

499.	*Pocillopora kelleheri* Veron, 2002	x	x	x	Ramakrishna *et al.*, 2010; Mondal & Raghunathan. 2017
500.	*Pocillopora ligulata* Dana, 1846	x	x	Pillai 1971	Venkataraman *et al.*, 2003; Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
501.	*Pocillopora meandrina* Dana, 1846	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
502.	*Pocillopora molokensis* Vaughan, 1907	x	x	Suresh 1991	x
503.	*Pocillopora verrucosa* (Ellis & Solander, 1786)	x	Pillai 1967e	Pillai 1967	Pillai 1967e; Mondal & Raghunathan. 2017
Genus *Seriatopora* Lamarck, 1816					
504.	*Seriatopora aculeata* Quelch,1886	x	x	x	Ramakrishna *et al.*, 2010, Mondal *et al.*, 2010a
505.	*Seriatopora caliendrum* Ehrenberg, 1834	x	x	x	Turner *et al.*, 2009
506.	*Seriatopora crassa* Quelch, 1886 (*nomen dubium*)	x	x	x	Tikader *et al.*, 1986
507.	*Seriatopora guttata* Veron, 2000	x	x	x	Mondal *et al.*, 2012
508.	*Seriatopora hystrix* Dana, 1846	x	x	x	Tikader *et al.*, 1986
509.	*Seriatopora stellata* Quelch, 1886	x	x	x	Tikader *et al.*, 1986
Genus *Stylophora* Schweigger, 1820					
510.	*Stylophora danae* Milne Edwards & Haime, 1850	x	x	x	Mondal *et al.*, 2015
511.	*Stylophora pistillata* Esper, 1797	x	x	Pillai 1967e	Reddiah 1977
512.	*Stylophora subseriata* (Ehrenberg, 1834)	x	x	x	Mondal *et al.*, 2012, 2015
513.	*Stylophora wellsi* Scheer, 1964	x	x	x	Mondal *et al.*, 2015
Family PORITIDAE Grey, 1842					
Genus *Bernardpora* Kitano & Fukami, 2014					
514.	*Bernardpora stutchburyi* Wells, 1955	Pillai & Patel 1988	Pillai 1967b	x	x
Genus *Goniopora* de Blainville, 1830					
515.	*Goniopora albiconus* Veron, 2000	x	x	x	Mondal *et al.*, 2012
516.	*Goniopora burgosi Nemenzo, 1955*	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2019
517.	*Goniopora columna* Dana,1846	Singh *et al.*, 2003	x	x	Reddiah 1977
518.	*Goniopora djiboutiensis* Vaughan, 1907	Sreenath 2015	Pillai 1967	x	Raghuraman *et al.*, 2012[#]
519.	*Goniopora eclipsensis* Veron & Pichon, 1982	x	x	x	Mondal *et al.*, 2015
520.	*Goniopora fruticosa* Saville-Kent, 1891	x	x	x	Mondal *et al.*, 2019
521.	*Goniopora lobata* Milne Edwards & Haime,1860	x	x	Caeiro 1999	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
522.	*Goniopora norfolkensis* Veron & Pichon,1982	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012

523.	*Goniopora paliformis* (Veron, 2000)	x	x	x	Raghuraman *et al.*, 2012#
524.	*Goniopora palmensis* Veron & Pichon, 1982	x	x	x	Mondal *et al.*, 2012
525.	*Goniopora pandoraensis* Veron & Pichon, 1982	x	x	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
526.	*Goniopora pearsoni* Veron, 2000		x	x	Raghunathan 2015; Mondal *et al.*, 2015, 2019
527.	*Goniopora pedunculata* Quoy & Gaimard, 1833*	Pillai & Patel 1988	Venkataraman & Rajan 2013	Pillai 1971	Mondal *et al.*, 2013
528.	*Goniopora pendulus* Veron, 1985	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2011
529.	*Goniopora planulata* (Ehrenberg,1834)	Pillai & Patel 1988	Pillai 1967	x	Tikader *et al.*, 1986
530.	*Goniopora somaliensis* Vaughan, 1907	x	x	x	Mondal *et al.*, 2019
531.	*Goniopora stokesi* Milne Edwards & Haime, 1851*	Sreenath 2015	Pillai 1967 Palk bay	Pillai 1971	Reddiah 1977
532.	*Goniopora savignyi* (Dana, 1846)	x	x	x	Mondal *et al.*, 2019
533.	*Goniopora tenella* (Quelch, 1886)	x	x	x	Mondal *et al.*, 2012
534.	*Goniopora tenuidens* (Quelch, 1886)	x	x	x	Reddiah 1977
Genus *Stylaraea* Edwards & Haime, 1851					
535.	*Stylaraea punctata* (Linnaeus, 1758)	x	Raghuram & Venkataraman, 2006	x	x
Genus *Porites* Link, 1807					
536.	*Porites annae* Crossland, 1952	x	Raghuram & Venkataraman, 2005	Sreenath *et al.*, 2015	Turner *et al.*,2009; Rajan *et al.*, 2010
537.	*Porites arnaudi* Reyes-Bonilla & Carricart Ganivet, 2000	x	Geetha & Kumar 2012	x	Ramakrishna *et al.*, 2010; Venkataraman *et al.*, 2012
538.	*Porites australiensis* Vaughan, 1918	x	x	Sreenath *et al.*, 2015	Mondal *et al.*, 2016
539.	*Porites cumulatus* Nemenzo,1955	x	x	x	Raghuraman *et al.*, 2012#
540.	*Porites compressa* Dana, 1846*	Pillai & Patel 1988	Pillai 1967	Caeiro 1999	Venkataraman *et al.*, 2012
541.	*Porites cylindrica* Dana, 1846	x	x	Pillai 1967	Raghuram & Venkataraman, 2005
542.	*Porites densa* Vaughan, 1918	x	x	x	Raghunathan *et al.*, 2013; Mondal *et al.*, 2015
543.	*Porites evermanni* Vaughan, 1907	x	x	x	Turner *et al.*, 2009
544.	*Porites exserta* Pillai, 1967	x	Pillai 1967	x	x
545.	*Porites fragosa* Dana, 1846	x	Pillai 1967	x	x
546.	*Porites harrisoni* Veron, 2000	x	x	x	Ramakrishna *et al.*, 2010; Raghuraman *et al.*, 2013#
547.	*Porites heronensis* Veron, 1985	x	x	x	Ramakrishna *et al.*, 2010; Sarkar & Ghosh 2013

548.	*Porites latistellata* Quelch, 1886	x	x	x	Rajan *et al.*, 2010
549.	*Porites lichen* Dana, 1846*	Pillai & Patel 1988	Pillai 1967	Pillai 1989	Tikader *et al.*, 1986
550.	*Porites lutea* Milne Edwards & Haime, 1851*	Pillai & Patel 1988	Pillai 1967, Venkataraman & Rajan 2013	Pillai 1971	Reddiah 1977
551.	*Porites lobata* Dana, 1846	x	Venkataraman & Rajan, 2013	Pillai 1989	Tikader *et al.*, 1986
552.	*Porites mannarensis* Pillai, 1967	x	Pillai 1967	x	x
553.	*Porites mayeri* Vaughan, 1918	x	x	x	Mondal *et al.*, 2015
554.	*Porites minicoiensis* Pillai, 1967	x	x	Pillai 1967b	x
555.	*Porites monticulosa* Dana,1846	x	x	x	Turner *et al.*, 2009
556.	*Porites murrayensis* Vaughan, 1918	x	Pillai 1967b, Venkataraman & Rajan, 2013	Suresh 1991	Venkataraman *et al.*, 2012; Mondal *et al.*, 2019
557.	*Porites myrmidonensis* Veron, 1985	x	x	x	Ramakrishna *et al.*, 2010
558.	*Porites nigrescens* Dana, 1848	x	x	Caeiro 1999	Reddiah 1977
559.	*Porites nodifera* Klunzinger, 1879	x	Edward *et al.*, 2007	x	Raghunathan *et al.*, 2015; Mondal *et al.*, 2019
560.	*Porites rus* (Forskål, 1775)	x	x	Pillai 1989 as	Venkataraman *et al.*, 2003; Venkataraman *et al.*, 2012
561.	*Porites sillimaniani* Nemenzo, 1976	x	x	x	Mondal *et al.*, 2019#
562.	*Porites solida* (Forskal, 1775)*	Singh *et al.*, 2003	Pillai 1967	Pillai 1971	Pillai 1967
563.	*Porites somaliensis* Gravier, 1911	x	Pillai 1967	Pillai 1971	Raghuraman *et al.*, 2012#
564.	*Porites stephensoni* Crossland, 1952	x	x	x	Rajan *et al.*, 2010
565.	*Porites vaughani* Crossland, 1952	x	x	Jeyabaskaran 2009	Mondal *et al.*, 2013
Family PSAMMOCORIDAE Chevalier & Beauvais, 1987					
Genus *Psammocora* Dana, 1846					
566.	*Psammocora contigua* (Esper, 1794)	x	x	Pillai 1967	Reddiah 1977,
567.	*Psammocora digitata* Milne Edwards & Haime, 1851	Pillai & Patel 1988	x	Pillai 1989	Turner *et al.*, 2009
568.	*Psammocora haimiana* Milne Edwards & Haime, 1851	x	x	Pillai 1971	Turner *et al.*, 2009
569.	*Psammocora nierstraszi* Van der Horst, 1921	x	x	x	Mondal *et al.*, 2012
570.	*Psammocora profundacella* Gardiner, 1898	x	x	Pillai 1989	Matthai 1924a, Turner *et al.*, 2009
Family SCLERACTINIA *incertae sedis*					
Genus *Leptastrea* Milne Edwards & Haime, 1849					
571.	*Leptastrea aequalis* Veron, 2000	x	Venkataraman & Rajan 2013	x	Mondal *et al.*, 2015, 2017, 2019
572.	*Leptastrea bottae* (Milne Edwards & Haime, 1849)	x	x	Pillai 1971	Mondal *et al.*, 2013

573.	*Leptastrea pruinosa* Crossland, 1952	x	x	x	Raghunathan 2015; Mondal *et al.*, 2017
574.	*Leptastrea purpurea* (Dana, 1846)*	Pillai & Patel 1988	Pillai 1967, Venkataraman & Rajan 2013	Pillai 1971	Tikader *et al.*,1986
575.	*Leptastrea transversa* Klunzinger, 1879*	Pillai & Patel 1988	Pillai 1967	Pillai 1971	Turner *et al.*, 2009; Mondal *et al.*, 2011
Genus *Oulastrea* Milne Edwards & Haime, 1848					
576.	*Oulastrea crispata* (Lamarck, 1816)	x	x	x	Pillai 1967
Genus *Pachyseris* Milne Edwards & Haime, 1849					
577.	*Pachyseris foliosa* Veron, 1990	x	x	x	Ramakrishna *et al.*, 2010; Mondal *et al.*, 2011
578.	*Pachyseris gemmae* Nemenzo, 1955	x	x	x	Reddiah 1977 Venkataraman *et al.*, 2003
579.	*Pachyseris rugosa* (Lamarck, 1801)	x	Pillai 1967	Jeyabaskaran 2009	Tikader *et al.*, 1986
580.	*Pachyseris speciosa* (Dana, 1846)	x	x	Caeiro 1999	Tikader *et al.*, 1986
Genus *Physogyra* Quelch, 1884					
581.	*Physogyra lichtensteini* (Milne Edwards & Haime, 1851)	x	x	Jeyabaskaran 2009	Tikader *et al.*,1986
Genus *Plesistrea* Milne Edwards & Haime, 1848					
582.	*Plesiastrea versipora* (Lamarck, 1816)*	Pillai & Patel 1988	Venkataraman & Rajan 2013	Pillai 1971	Tikader *et al.*,1986
Genus *Plerogyra* Milne Edwards & Haime, 1848					
583.	*Plerogyra sinuosa* (Dana, 1846)	x	x	x	Tikader *et al.*, 1986
584.	*Plerogyra simplex* Rehberg, 1892	x	x	x	Mondal *et al.*, 2011
Family SIDERASTREIDAE Vaughan & Wells, 1943					
Genus *Pseudosiderastrea* Yabe & Sugiyama, 1935					
585.	*Pseudosiderastrea tayamai* Yabe & Sugiyama, 1935	Pillai & Patel 1988	x	x	Tikader *et al.*, 1986
Genus: *Siderastrea* Blainville, 1830					
586.	*Siderastrea savignyana* Milne Edwards & Haime, 1850	Pillai & Patel 1988	Pillai 1967	x	Pillai 1967
Family RHIZANGIIDAE d'Orbigny, 1951					
Genus *Astrangia* Milne Edwards & Haime, 1848					
587.	*Astrangia* Sp.	Anil & Wagh, 1984 reported from Goa			
Genus *Cladangia* Milne Edwards & Haime, 1851					
588.	*Cladangia exusta* Lütken, 1873	Pillai 1967d reported from off Cochin water			
Genus *Culicia* Dana, 1846					
589.	*Culicia rubeola* (Quoy & Gaimard, 1833)	x	Pillai 1967	x	Tikader *et al.*, 1986

***** Species are present in all the coral reefs of GoK, GoMBR, LKD, A&N. #species distribution records need validation.

2.4.1. Annotated list of erroneous records of scleractinian species that excluded in the present checklist

ACROPORIDAE

1. ***Acropora akajimensis* Veron, 1990:** Mondal *et al.*, (2014) reported this species from A&N Islands. This is a synonymy of *Acropora donei* (Veron and Wallace, 1984) as of Hoeksema and Cairns (2018).
2. ***Acropora armata* (Brook, 1892):** The inclusion of this species is based on the report of Reddiah (1977) from the A&N Island, which is a synonymy of *Acropora cytherea* (Dana, 1846) as of Hoeksema and Cairns (2018).
3. ***Acropora azurea* Veron and Wallace, 1894:** This species is mentioned by MOEF (2015) from the A&N Island, which is a synonymy of *Acropora nana* (Studer, 1878) as of Hoeksema and Cairns (2018).
4. ***Acropora brueggemanni* (Brook, 1893)**: This species reported by Reddiah (1977) in A&N Islands, is an old combination of *Isopora brueggemanni* (Brook, 1893) as of Hoeksema and Cairns (2018).
5. ***Acropora calamaria* (Brook 1892):** This species is recorded by Reddiah (1977) from the A&N Islands, which is the synonymy of *Acropora valida* (Dana, 1846) as of Hoeksema and Cairns (2018).
6. ***Acropora caroliana* Nemenzo, 1976:** This report is by Mondal *et al.,* (2013) in the A&N Islands, which is the wrong spelling of *Acropora caroliniana* Nemenzo, 1976 as of Hoeksema and Cairns (2018).
7. ***Acropora conferta* (Quelch, 1886):** This species described by Pillai (1967, 1971) from the GoMBR, and the Lakshadweep, is the synonymy of *Acropora hyacinthus* (Dana, 1846) as of Hoeksema and Cairns (2018).
8. ***Acropora conigera* (Dana, 1846):** Reddiah (1977) reported this species in the A&N Islands, which is the synonymy of *Acropora robusta* (Dana,1846) as of Hoeksema and Cairns (2018).
9. ***Acropora copiosa* Nemenzo, 1967:** Raghuraman *et al.,* (2012) and Mondal *et al.,* (2013) recorded this species from the A&N Islands, which is the synonymy of *Acropora muricata* (Linnaeus, 1758) as of Hoeksema and Cairns (2018).

10. ***Acropora corymbosa*** **(Lamarck, 1816):** This record is based on Reddiah (1977), and Pillai (1971) from the A&N Islands is the synonymy of *Acropora cytherea* (Dana, 1846) as of Hoeksema and Cairns (2018).
11. ***Acropora crateriformes*** **(Gardiner, 1898):** Mondal *et al.,* (2014) described this species from the A&N Islands, is an old combination of *Isopora crateriformes* (Gardiner, 1898) as of Hoeksema and Cairns (2018).
12. ***Acropora cuneata*** **(Dana, 1846):** The inclusion of this record is based on Pillai (1967) from the GoMBR and Raghuraman *et al.,* (2012) in the A&N Islands, is an old combination of *Isopora cuneata* (Dana, 1846) as of Hoeksema and Cairns (2018).
13. ***Acropora efflorescens*** **(Dana, 1846):** The inclusion of this record is based on Pillai (1971) and Mondal *et al.,* (2014) from the Lakshadweep and the A&N Islands respectively, is the synonymy of *Acropora cytherea* (Dana, 1846) as of Hoeksema and Cairns (2018).
14. ***Acropora elizabethensis*** **Veron, 2000:** Raghuraman *et al.,* (2012, 2013) reported this species from the A&N Islands, is the synonymy of *Isopora elizabethensis* (Veron, 2000) as of Hoeksema and Cairns (2018).
15. ***Acropora formosa*** **(Dana, 1846):** This record is based on work of Pillai (1967) from the A&N Islands, the Lakshadweep and the GoMBR, also in a subsequent publication of Raghuraman *et al.,* (2012) and Mondal *et al.,* (2013) from A&N, is the original combination, the basionym of *Acropora muricata* (Linnaeus, 1758) as reported by Venkataraman *et al.,* 2003 as of Hoeksema and Cairns (2018).
16. ***Acropora inermis*** **(Brook, 1891):** The inclusion of this species is based on Raghuraman *et al.,* (2012) from the A&N Islands, which is the synonymy of *Acropora microphthalma* (Verrill, 1859) as of Hoeksema and Cairns (2018).
17. ***Acropora irregularis*** **(Brook, 1892):** This species is recorded by Reddiah 1977, Raghuraman *et al.,* (2012), and Mondal *et al.,* (2013) from the A&N Islands, is the synonymy of *Acropora abrotanoides* (Lamarck,1816) as of Hoeksema and Cairns (2018).
18. ***Acropora hebes*** **(Dana, 1846):** This species recorded by Patterson *et al.,* (2007) in the GoMBR, which is the synonymy of *Acropora aspera* (Dana,1846) as of Hoeksema and Cairns (2018).
19. ***Acropora massawensis*** **von Marenzeller, 1907:** The inclusion of this species based on Raghuraman *et al.,* (2012) and Mondal *et al.,* (2013) in the A&N Islands, is the synonymy of *Acropora polystoma* (Brook, 1891) as of Hoeksema and Cairns (2018).

20. ***Acropora meridiana* Nemenzo, 1971:** Mondal *et al.,* (2015) description of this species from the A&N Islands, is an old combination of *Isopora brueggemanni* (Brook, 1893) as of Hoeksema and Cairns (2018).
21. ***Acropora nobilis* (Dana 1846):** This species reported by Pillai (1967) from the GoMBR and the A&N Islands, is the synonymy of *Acropora robusta* (Dana,1846) as of Hoeksema and Cairns (2018).
22. ***Acropora ocellata* (Klunzinger, 1879):** The inclusion of this record by Raghunathan *et al.,* (2013) in the A&N Islands, is the synonymy of *Acropora humilis* (Dana, 1846) as of Hoeksema and Cairns (2018).
23. ***Acropora pacifica* (Brook)**: This species was reported by Reddiah (1977) from the A&N Islands, which is the synonymy of *Acropora robusta* (Dana,1846) as of Hoeksema and Cairns (2018).
24. ***Acropora palifera* (Lamarck, 1816):** Reported by Pillai (1971) from the Lakshadweep, Reddiah (1977), Raghuraman *et al.,* (2012) and Mondal *et al.,* (2013) from the A&N islands, is s of *Isopora palifera* (Lamarck, 1816) as of Hoeksema and Cairns (2018).
25. ***Acropora pillaii* Patterson Edward *et al.*, 2008:** Patterson *et al.*, 2008 mentioned the presence of this species in the GoMBR, but this is a nomen nudum, which is not a valid name. Hence, this species not included in the present checklist Hoeksema and Cairns (2018).
26. ***Acropora pinguis* Wells, 1950:** This species reported by Mondal *et al.,* (2010) from the A&N Islands, is the synonymy of *Acropora robusta* (Dana,1846) as of Hoeksema and Cairns (2018).
27. ***Acropora plana* Nemenzo, 1967**: The inclusion of this record by Ramakrishna *et al.,* (2010) from the A&N Islands, is the synonymy of *Acropora tenuis* (Dana, 1846) as of Hoeksema and Cairns (2018).
28. ***Acropora polymorpha* (Brook):** Reddiah (1977) reported from the A&N Islands, is the synonymy of *Acropora florida* (Dana,1846) as of Hoeksema and Cairns (2018).
29. ***Acropora rambleri* Bassett-Smith, 1890:** This record based by Pillai (1971) from the Lakshadweep, and Tikader *et al.,* (1986) from the A&N Islands, is the synonymy of *Acropora speciosa* (Quelch, 1886) as of Hoeksema and Cairns (2018).
30. ***Acropora reticulata* (Brook, 1892):** Occurrence of this species recorded by Pillai (1971) from the Lakshadweep, is the synonymy of *Acropora cytherea* (Dana, 1846) as of Hoeksema and Cairns (2018).

31. ***Acropora rosaria* (Dana, 1846):** This record described by Mondal *et al.,* (2015) from the A&N Islands, is the synonymy of *Acropora loripes* (Brook, 1892) as of Hoeksema and Cairns (2018).
32. ***Acropora schmitti* Wells, 1950:** The occurrence of this species reported by Mondal *et al.,* (2014) from the A&N Islands, is the synonymy of *Acropora dijitifera* (Dana, 1846) as of Hoeksema and Cairns (2018).
33. ***Acropora sekiseiensis* Veron, 1990:** The inclusion of this record by Raghuraman *et al.*, (2012) and Mondal *et al.,* (2013) from the A&N Islands, is the synonymy of *Acropora horrida* (Dana, 1836) as of Hoeksema and Cairns (2018).
34. ***Acropora surculosa* (Dana, 1846):** This species reported by Reddiah (1977) from the A&N Islands, is the synonymy of *Acropora hyacinthus* (Dana, 1846) as of Hoeksema and Cairns (2018).
35. ***Acropora syringoides* (Brook, 1892):** Pillai (1967) recorded this species from the GoMBR and the A&N Islands, which is the synonymy of *Acropora longicyathus* (Milne Edwards, 1860) as of Hoeksema and Cairns (2018).
36. ***Acropora tenius* (Dana, 1846):** Raghuraman *et al.,* (2012) reported this species from A&N Islands, which is a wrong spelling of *Acropora tenuis* (Dana, 1846) as of Hoeksema and Cairns (2018).
37. ***Acropora torresiana* Veron, 2000**: This species reported by Sukumaran *et al.,* 2007 from GoMBR and Ramakrishna *et al.,* 2010 from A&N Islands is unaccepted. Accepted name is *Acropora samoensis* (Brook, 1891) as of Hoeksema and Cairns (2018).
38. ***Acropora tutuilensis* Hoffmeister, 1925:** This record based on Mondal *et al.,* (2014) from the A&N Islands, is the synonymy of *Acropora abrotanoides* (Lamarck,1816) as of Hoeksema and Cairns (2018).
39. ***Acropora variabilis* (Klunzinger, 1879):** The occurrence of this species reported by Pillai (1972) from the A&N Islands, is the synonymy of *Acropora valida* (Dana, 1846) as of Hoeksema and Cairns (2018).
40. ***Acropora vermiculata* Nemenzo, 1967**: Turner *et al.,* (2009) documented from the A&N Islands, which is the synonymy of *Acropora sarmentosa* (Brook, 1892) as of Hoeksema and Cairns (2018).

41. ***Acropora wallaceae*** **Veron, 1990:** The Occurrence report based on Mondal *et al.,* (2014) from the A&N Islands, is the synonymy of *Acropora samoensis* (Brook,1891) as of Hoeksema and Cairns (2018).
42. ***Anacropora pillai*** **(Patterson, 2006):** The inclusion of this record by Raghuraman *et al.,* (2013) the A&N Islands is a wrong spelling of *Anacropora pillai* Veron, 2000 as of Hoeksema and Cairns (2018).
43. ***Montipora composita*** **Crossland, 1952:** This species reported from the GoMBR (Pillai 1967) and the A&N Islands (Reddiah 1977), is the synonymy of *Montipora aequituberculata* Bernard,1897 as of Hoeksema and Cairns (2018).
44. ***Montipora divericata*** **Brüggemann, 1879:** Reported from the GoMBR (Bernard 1897), is the synonymy of *Montipora digitata* (Dana, 1846) as of Hoeksema and Cairns (2018).
45. ***Montipora foveolate*** **(Dana, 1846)***:* Reported from Lakshadweep Island by Jeyabaskaran 2007 is the wrong spelling of *Montipora feveolata* (Dana, 1846) as of Hoeksema and Cairns (2018).
46. ***Montipora fruiticosa*** **Bernard, 1897:** Reddiah (1977) recorded this species from the A&N Islands, which is the synonymy of *Montipora digitata* (Dana, 1846) as of Hoeksema and Cairns (2018).
47. ***Montipora subtilis*** **Bernard, 1897:** Pillai (1967) reported this species from GoMBR, is the synonymy of *Montipora millepora* Crossland, 1952 as of Hoeksema and Cairns (2018).
48. ***Montipora turgescens*** **(Dana):** Tikader *et al.*, (1986) included this species from the A&N Islands, which is the wrong reference of *Montipora turgescens* Bernard, 1897, as of Hoeksema and Cairns (2018).

AGARICIIDAE

49. ***Agaricia fragilis*** **Dana, 1848**: Ramakrishna *et al.,* 2010 misidentified from the Andaman Islands. This is an Atlantic species (Hoeksema and Cairns, 2018 and IUCN 2008). Hence, excluded from this checklist.
50. ***Leptoseris cucullata*** **(Ellis and Solander, 1786):** Raghuraman *et al.*, (2012), Raghunathan and Venkataraman (2012) and Mondal *et al.,* (2013) reported from the A&N Islands, is the previous combination of *Helioseris cucullata* (Ellis and Solander, 1786) as of Hoeksema, (2015) This is an Atlantic species (Hoeksema and Cairns, 2018), hence, excluded from the present checklist.

51. ***Pavona* (*Polyastra*) *obtusta* (Quelch, 1884)**: Reddiah (1977) recorded from the A&N Islands, is the previous combination of *Pavona venosa* (Ehrenberg, 1834) as of Hoeksema and Cairns (2018).
52. ***Pavona lata* Dana 1846:** Matthai (1924) documented from the A&N Islands, is the synonymy of *Pavona decussata* (Dana, 1846) as of Hoeksema and Cairns (2018).
53. ***Pavona praetorta* (Dana 1846):** Matthai (1924) reported from the A&N Islands, is the synonymy of *Pavona cactus* (Forskål, 1775) as of Hoeksema and Cairns (2018).

CARYOPHYLLIIDAE

54. ***Caryophyllia (Caryophyllia) clavus* (Scacchi, 1835):** This species reported by Alcock (1988) from the Kerala coast and Raghuraman *et al.,* (2012) from the A&N Islands, is synonymy of *Caryophyllia (Caryophyllia) smithii* Stokes and Broderip, 1828 as of Hoeksema and Cairns, (2018).
55. ***Caryophyllia acanthocyathus* Milne Edwards and Haime, 1848:** Venkataraman (2006) erroneously included this record from Andaman Island, is a Subgenus as of Hoeksema and Cairns (2018).
56. ***Paracyathus caeruleus* Duncan, 1889:** Mondal *et al.,* (2014) reported the occurrence of this species from the A&N Islands, is the synonymy of *Paracyathus rotundatus* Semper, 1872 as of Hoeksema and Cairns (2018).
57. ***Paracyathus stokesi* Milne Edwards & Haime, 1848:** Venkatraman (2006), Mondal *et al.*, (2015), Mondal and Raghunathan (2017) repoterted from the Andaman Islands. Which is a spelling mistake, accepted name is *Paracyathus stokesii* Milne Edwards & Haime, 1848.
58. ***Desmophyllum vitreum* Alcock 1898:** Alcock (1898) reported this species from the Kerala coast, which is a synonymy of *Javania cailleti* (Duchassaing and Michelotti, 1864) as of Hoeksema and Cairns (2018).
59. ***Solenosmilia jeffreyi* Alcock, 1898***:* Alcock (1898) reported this species from the Kerala coast. This is a synonym of the *Solenosmilia variabilis* Duncan, 1873 as of Hoeksema and Cairns (2018).

COSCINARAEIDAE

60. ***Coscinaraea wellsi* Veron & Pichon, 1980:** Mondal *et al.,* (2015) reported this species from the Middle and South Andaman Archipelago. However, it is an original combination, basionym of *Cycloseris wellsi* (Veron & Pichon, 1980).

DENDROPHYLLIIDAE

61. ***Dendrophyllia coarctata* Duncan, 1889**: This record included by Pillai (1986) from the GoMBR, which is the synonymy of *Cladopsammia gracilis* (Milne Edwards and Haime, 1848) as of Hoeksema and Cairns (2018).
62. ***Dendrophyllia micrantha* (Ehrenberg, 1834):** Raghuraman *et al.*, (2013) included this species. The original name is *Tubastraea micranthus* (Ehrenberg, 1834) as of Hoeksema and Cairns (2018).
63. ***Dendrophyllia micranthus*:** Tikader *et al.*, (1986) reported in A&N Islands, is the wrong spelling of *Dendrophyllia micrantha*, which is a synonymy of *Tubastraea micranthus* (Ehrenberg, 1834) as of Hoeksema and Cairns (2018).
64. ***Diploria clivosa* (Ellis & Solander, 1786):** Krishnan *et al.*, (2018) reported this species from the GoMBR. However, this is a Caribbean species as per the IUCN red list of threatened species (2008); therefore, we excluded this species from the present checklist.
65. ***Diploria strigosa* (Dana, 1846):** Raghuraman *et al.*, (2012) reported from the Andaman Islands, is a new combination of *Pseudodiploria strigosa* (Dana, 1846) as of Hoeksema and Cairns (2018). However, *P. strigosa* (Dana, 1846) was reported from Andaman by Ramakrishna *et al.*, 2010, which is Atlantic species (IUCN, 2008). Hence, this record is not listed.
66. ***Enallopsammia amphleioides*:** This species was reported by Tikader *et al.*, (1986) from the A&N Islands, which is synonymy of *Enallopsammia rostrata* (Pourtalès, 1878) as of Cairns, (2010).
67. ***Enallopsammia marenzelleri* Zibrowius, 1973:** Tikader *et al.*, (1986) recorded from the A&N Islands, is synonymy of *Enallopsammia pusilla* (Alcock, 1902) as of Hoeksema and Cairns (2018).
68. ***Heteropsammia geminata* Verrill, 1870:** Alcock (1893) and Pillai (1972) recorded from the A&N Islands, is synonymy of *Heteropsammia eupsammides* (Gray, 1849) as of Hoeksema and Cairns (2018).
69. ***Heteropsammia michelinii* Milne Edwards & Haime, 1848**: Tikader *et al.*, (1986) reported from A&N and Pillai and Jasmine (1995) from off Quilon, synonymy of *Heteropsammia cochlea* (Spengler, 1781) as of Hoeksema and Cairns (2018).

70. ***Turbinaria veluta* Bernard, 1896:** This species was included by Tikader *et al.,* (1986) from the A&N Islands, which is synonymy of *Turbinaria reniformis* Bernard, 1896 as of Hoeksema and Cairns (2018).

FLABELLIDAE

71. ***Flabellum crassum* Milne Edwards and Haime 1848**: Bourne (1905) reported this species from GoMBR, this is a synonym of *Truncatoflabellum crassum* (Milne Edwards and Haime 1848) as of Hoeksema and Cairns (2018).

FUNGIIDAE

72. ***Cantharellus noumeae* Hoeksema and Best, 1984**: This species is an endemic of New Caledonia and does not occur elsewhere. Hence, it is excluded from the present checklist.

73. ***Cycloseris colini* Veron, 2000:** Mondal *et al.,* (2015, 2019) and Raghunathan *et al.,* (2015) documented from the A&N Islands, is an original combination, the basionym of *Lithophyllon spinifer* (Claereboudt and Hoeksema, 1987) as of Hoeksema and Cairns (2018).

74. ***Cycloseris erosa* (Döderlein, 1901):** This record described by Ramakrishna *et al.,* (2010) from the A&N Islands, is synonymy of *Cycloseris tenuis* (Dana, 1846) as of Hoeksema and Cairns (2018).

75. ***Cycloseris marginata* (Boschma, 1923):** The inclusion of this species is based on Raghunathan *et al.,* (2013) from the A&N Islands, which is synonymy of *Cycloseris costulata* (Ortmann, 1889) as of Hoeksema and Cairns (2018).

76. ***Cycloseris mycoides* Alcock 1893:** Raghunathan *et al.,* (2013) recorded from the A&N Islands, is synonymy of *Cycloseris sinensis* Milne Edwards and Haime, 1851 as of Hoeksema and Cairns (2018).

77. ***Cycloseris patelliformis* (Boschma, 1923):** Raghuraman *et al.,* (2012) mentioned presence of this species in the A&N Islands, which is synonymy of *Cycloseris fragilis* (Alcock, 1893) as of Hoeksema and Cairns (2018).

78. ***Diaseris distorta* (Michelin, 1842):** Raghuraman *et al.,* (2012) reported from the A&N Islands, is the previous combination of *Cycloseris distorta* (Michelin, 1842) as of Hoeksema and Cairns (2018).

79. ***Diaseris fragilis* Alcock 1893**: This report, based on Raghuraman *et al.,* (2013) in the A&N Islands, is an original combination, basionym of *Cycloseris fragilis* Alcock, 1893 as of Hoeksema and Cairns (2018).
80. ***Fungia (Danafungia) corona* Döderlein, 1901:** Matthai (1924), Raghuraman *et al.*, (2012) recorded from the A&N Islands, is a previous combination of *Danafungia scruposa* (Klunzinger, 1879) as of Hoeksema, and Cairns (2018).
81. ***Fungia (Danafungia) danai* Milne Edwards and Haime, 1851:** Matthai (1924) and Pillai (1971) included from the A&N Islands and Lakshadweep respectively, is the previous combination of *Danafungia horrida* (Dana, 1846) as of Hoeksema and Cairns (2018).
82. ***Fungia scruposa* Klunzinger, 1879**: Caeiro (1999) and Raghuraman *et al.,* (2012) recorded from the Lakshadweep and the A&N Islands, respectively, which is an original combination, basionym of *Danafungia scruposa* (Klunzinger, 1879) as of Hoeksema and Cairns (2018).
83. ***Fungia (Danafungia) subrepanda* Döderlein, 1901:** Matthai (1924) recorded from the A&N Islands, is a junior synonym of *Danafungia scruposa* (Klunzinger, 1879) as of Hoeksema and Cairns (2018).
84. ***Fungia (Verrillofungia) concinna* Verrill, 1864:** Matthai (1924), and Caeiro (1999) reported from Andaman and Lakshadweep, respectively, is the previous combination of *Lithophyllon concinna* (Verrill, 1864) as of Hoeksema and Cairns (2018).
85. ***Fungia echinata* (Pallas, 1766):** Pillai (1967) described this species from the A&N Islands, is the previous combination of *Ctenactis echinata* (Pallas, 1766) as of Hoeksema and Cairns (2018).
86. ***Fungia fralinae* Nemenzo, 1955:** Raghuraman *et al.*, (2012), included the A&N Islands, is an original combination, and basionym of *Heliofungia fralinae* (Nemenzo, 1955) as of Hoeksema and Cairns (2018).
87. ***Fungia granulosa* Klunzinger, 1879:** Jeyabaskaran (2009) reported from Lakshadweep, Turner *et al.,* (2009) and Rajan *et al.,* (2010) reported from the A&N Islands, is an original combination, and the basionym of *Pleuractis granulosa* (Klunzinger, 1879) as of Hoeksema and Cairns (2018).
88. ***Fungia klunzingeri* Döderlein, 1901**: Turner *et al.,* (2009) recorded from the A&N Islands, is junior synonymy of *Danafungia horrid*a (Dana, 1846) as of Hoeksema and Cairns (2018).

89. ***Fungia molluccensis* Horst, 1919:** Turner *et al.,* (2009) and Rajan *et al.,* (2012) reported from the A&N Islands, is an original combination, the basionym of *Pleuractis moluccensis* (Van der Horst, 1919) as of Hoeksema and Cairns (2018).
90. ***Fungia paumotensis* Stuchbury, 1833:** Matthai (1924a) documented from the A&N Islands, is an original combination, the basionym of *Pleuractis paumotensis* (Stutchbury, 1833) as of Hoeksema and Cairns (2018).
91. ***Fungia puishani* Veron and DeVantier, 2000:** Mondal *et al.,* (2012) reported from the A&N Islands, is a junior synonym of *Fungia fungites* (Linnaeus, 1758) as of Hoeksema and Cairns (2018).
92. ***Fungia repanda* Dana, 1846:** Tikader *et al.,*1986, Mondal *et al.,* (2013) reported from the A&N Islands, is an original combination, the basionym of *Lithophyllon repanda* (Dana, 1846) as of Hoeksema and Cairns (2018).
93. ***Fungia scabra* Döderlein, 1901:** Raghuraman *et al.,* (2012) recorded from the A&N Islands, is the original combination, basionym of *Lithophyllon scabra* (Döderlein, 1901) as of Hoeksema and Cairns (2018).
94. ***Fungia scutaria,* Lamarck 1801:** Pillai (1967) reported from Lakshadweep, Tikader *et al.,*1986 and Mondal *et al.,* 2013 from the A&N Islands, is the original combination, basionym of *Lobactis scutaria* (Lamarck, 1801) as of Hoeksema and Cairns (2018).
95. ***Fungia seychellensis* Hoeksema, 1993:** Jeyabaskaran (2009) reported from Lakshadweep, Raghuraman *et al.,* 2012, from the A&N Islands, is an original combination, the basionym of *Pleuractis seychellensis* (Hoeksema, 1993) as of Hoeksema and Cairns (2018).
96. ***Fungia simplex* (Gardiner, 1905):** Raghuraman *et al.,* (2013) recorded from the A&N Islands, is synonymy of *Ctenactis crassa* (Dana, 1846) as of Hoeksema and Cairns (2018).
97. ***Fungia spinifer* Claereboudt and Hoeksema, 1987:** Raghuraman *et al.,* (2012) documented from the A&N Islands, is an original combination, the basionym of *Lithophyllon spinifer* (Claereboudt and Hoeksema, 1987) as of Hoeksema and Cairns (2018).
98. ***Fungia taiwanensis* Hoeksema and Dai, 1991:** Raghuraman *et al.,* (2012) noted from the A&N Islands, is an original combination, basionym of *Pleuractis taiwanensis* (Hoeksema and Dai, 1991) as of Hoeksema and Cairns (2018).
99. ***Halomitra clavator* Höeksema, 1989:** Mondal and Raghunathan (2012) reported from the A&N Islands, but this species is patchily distributed in a small range and is rare (IUCN, 2008). It is

also an endemic species to Indonesia, Papua New Guinea, Philippines, and the Solomon Islands. Hence not included in this checklist.

100. ***Herpolitha weberi* (van der Horst, 1921):** Pillai (1972), Rajan *et al.,* (2010) reported from the A&N Islands, is a junior synonym of *Herpolitha limax* (Esper, 1797) as of Hoeksema and Cairns (2018).

101. ***Herpitoglossa simplex* (Gardiner, 1905):** Tikader *et al.,* (1986) noted this record from the A&N Islands, it is a misspelling of *Herpetoglossa simplex* (Gardiner, 1905), and is synonymy of *Ctenactis crassa* (Dana, 1846) as of Hoeksema and Cairns (2018).

FUNGIACYATHIDAE

102. ***Fungiacyathus symmetrica***: The inclusion of this record is based on the wrong spelling of *Fungiacyathus (Bathyactis) symmetricus* (Pourtalès, 1871) by Pillai (1972) from the A&N Islands as of Hoeksema and Cairns (2018).

LOBOPHYLLIDAE

103. ***Acanthastrea faviaformis* Veron, 2000:** Occurrence of this species mentioned by Mondal *et al.,* (2015, 2019) in the A&N Islands. The record is a synonymized name of *Dipsastraea faviaformis* (Veron, 2000) as of Hoeksema and Cairns (2018).

104. ***Acanthastrea hillae* Wells, 1955:** Venkataraman *et al.,* (2012) and Kumar *et al.,* (2014) reported this species from A&N Islands and GoK respectively. It is a synonym of *Homophyllia bowerbanki* (Milne Edwards, 1857) as of Hoeksema and Cairns (2018).

105. ***Acanthastrea ishigakiensis* Veron, 1990:** This species recorded by Turner *et al.,* (2009) in the A&N Islands, is basionym, and a previous combination of *Lobophyllia ishigakiensis* (Veron, 1990), as of Hoeksema and Cairns (2018).

106. ***Acanthastrea maxima* Sheppard and Salm, 1988:** The inclusion of this species in the A&N Islands by Mondal *et al.,* (2015), is an original combination, the basionym of as *Sclerophyllia maxima* (Sheppard and Salm, 1988) as of Hoeksema and Cairns (2018).

107. ***Acanthastrea regularis* Veron, 2000:** The inclusion of this species is based on Raghuraman *et al.,* (2013) in the A&N Islands, which is basionym, a previous combination of *Micromussa regularis* (Veron, 2000) as of Hoeksema and Cairns (2018).

108. ***Australomussa rowleyensis* Veron, 1985:** Turner *et al.,* (2009) recorded from the A&N Islands, is basionym, and the previous combination of *Lobophyllia rowleyensis* (Veron, 1985) as of Hoeksema and Cairns (2018).

109. ***Lobophyllia dentatus* Veron, 2000:** Mondal et aḷ. (2013) from the A&N Islands, is synonymy of *Lobophyllia dentata* Veron, 2000 as of Hoeksema and Cairns (2018).

110. ***Lobophyllia pachysepta* Chevalier, 1975**: Raghuraman *et al.,* (2012) from the A&N Islands, is synonymy of *Acanthastrea pachysepta* (Chevalier, 1975) as of Hoeksema and Cairns (2018).

111. ***Lobophyllia serratus* Veron, 2000:** Jeyabaskaran (2009) reported this species from Lakshawadeep Island but was wrongly spelled. Correct spelling is *Lobophyllia serrata* Veron, 2000, as of Hoeksema and Cairns (2018).

112. ***Symphyllia agaricia* Milne Edwards and Haime, 1849**: Raghuraman *et al.,* (2012) reported from the A&N Islands, is synonymized name of *Lobophyllia agaricia* (Milne Edwards and Haime, 1849) as of Hoeksema and Cairns (2018).

113. ***Symphyllia erythraea* (Klunzinger, 1879):** Ramakrishna *et al.*, (2010) from the A&N Islands, is the previous combination of *Lobophyllia erythraea* (Klunzinger, 1879) as of Hoeksema and Cairns (2018).

114. ***Symphyllia hassi* Pillai and Scheer, 1976:** Raghuraman *et al.*, (2012) from the A&N Islands, is basionym, a previous combination of *Lobophyllia hassi* (Pillai and Scheer, 1976) as of Hoeksema and Cairns (2018).

115. ***Symphyllia nobilis* (Dana 1846):** Pillai (1989) from Lakshadweep, Tikader *et al.*, (1986) from the A&N Islands, is synonymy of *Lobophyllia recta* (Dana, 1846) as of Hoeksema and Cairns (2018).

116. ***Symphyllia radians* Edwards & Haime, 1849:** This species was recorded by Pillai and Patel 1988 from GoK, Matthai 1924 from GoMBR, Pillai 1971 from LKD, Tikader *et al.*, 1986 from A&N Islands, which is a synonym of *Lobophyllia radians* (Milne Edwards & Haime, 1849) as of Hoeksema and Cairns (2018).

117. ***Symphyllia recta* (Dana, 1846):** Occurrence report of this species based on Singh *et al.*, 2003 from GoK, Matthai 1924 and Venkataraman and Rajan, 2013 from GoMBR, Pillai 1971 from LKD, Matthai 1924, Pillai 1972 from A&N Islands, which is a synonym of *Lobophyllia recta* (Dana, 1846) as of Hoeksema and Cairns (2018).

118. ***Symphyllia valenciennesii* Milne Edwards and Haime,1849:** Ramakrishna *et al.*, (2010) reported from the A&N Islands, is basionym and a previous combination of *Lobophyllia valenciennesii* (Milne Edwards and Haime, 1849) as of Hoeksema and Cairns (2018).

MERULINIDAE

119. ***Barabattoia amicorum* (Milne Edwards and Haime, 1849):** This record based on Satyanarayana and Ramakrishna (2009) at GoK and Mondal *et al.,* (2013), (2014) from the A&N Islands, is previous combination of *Dipsastraea amicorum* (Milne Edwards and Haime, 1849) as of Hoeksema and Cairns (2018).

120. ***Barabattoia laddi* (Wells, 1954):** The occurrence report based on Raghuraman *et al.,* (2012) and Mondal *et al.,* (2013) from the A&N Islands, is the previous combination of *Dipsastraea laddi* (Wells, 1954) as of Hoeksema and Cairns (2018).

121. ***Caulastrea curvata* Wijsman-Best, 1972:** The inclusion of this species by Mondal *et al.,* (2013) from the A&N Islands, is synonymized name of *Caulastraea curvata* Wijsman-Best, 1972 as of Hoeksema and Cairns (2018).

122. ***Favites bestae* Veron, 2000*:*** Mondal *et al.,* (2013) from the A&N Islands, is synonymy of *Favites melicerum* (Ehrenberg, 1834) as of Hoeksema and Cairns (2018).

123. ***Favites russelli* (Wells, 1954):** Caeiro (1999) from Lakshadweep, Raghuraman *et al.,* (2012), Mondal *et al.,* (2013) from the A&N Islands and Venkataraman and Rajan 2013 from Palk Bay, is the previous combination of *Paragoniastrea russelli* (Wells, 1954) as of Hoeksema and Cairns (2018).

124. ***Favites paraflexuosa* Veron, 2002**: This species reported from Andaman and Nicobar Islands by Ramakrishna *et al.,* 2010 and Mondal *et al.,* 2010a, is unaccepted because of wrong spelling (Hoeksema, and Cairns, 2018). The accepted name is *Favites paraflexuosus* Veron, 2000.

125. ***Goniastrea aspera* Verrill, 1866**: Caeiro (1999) reported from Lakshadweep, Mondal *et al.,* (2013, 2015) from the A&N Islands, is an original combination, and the basionym of *Coelastrea aspera* (Verrill, 1866) as of Hoeksema and Cairns (2018).

126. ***Goniastrea australensis* (Milne Edwards, 1857):** Pillai (1989) described from the Lakshadweep, Raghuraman *et al.,* (2012) reported from the A&N Islands, is a previous combination of *Paragoniastrea australensis* (Milne Edwards, 1857) as of Hoeksema and Cairns (2018).

127. ***Goniastrea benhami* Vaughan, 1917**: Reddiah (1977) recorded from the A&N Islands, is a synonym of *Paragoniastrea australensis* (Milne Edwards, 1857) as of Hoeksema and Cairns (2018).

128. ***Goniastrea hombroni* (Rosseau, 1854):** Pillai (1971) noted from the Lakshadweep, is a synonym of *Goniastrea stelligera* (Dana, 1846) as of Hoeksema and Cairns (2018).

129. ***Goniastrea incrustans* Duncan, 1889**: Pillai (1967) reported from the GoMBR, is a synonym of *Coelastrea aspera* (Verrill, 1866) as of Veron *et al.,* (1977) and Hoeksema and Cairns (2018)

130. ***Goniastrea palauensis* (Yabe and Sugiyama, 1936)**: Mondal *et al.,* (2012) recorded from the A&N Islands, is the previous combination of *Coelastrea palauensis* (Yabe and Sugiyama, 1936) as of Hoeksema and Cairns (2018).

131. ***Goniastrea peresi* (Faure and Pichon, 1978)**: Venkataraman *et al.,* (2012), Mondal and Raghunathan (2017) noted from the A&N Islands and Krishnan *et al.,* 2018 from the GoMBR, which is the previous combination of *Paramontastraea peresi* (Faure and Pichon, 1978) as of Hoeksema and Cairns (2018).

132. ***Goniastrea planulata* Milne Edwards and Haime, 1849:** Reddiah (1977) reported from the A&N Islands, is a synonym of *Goniastrea pectinata* (Ehrenberg, 1834) as of Hoeksema and Cairns (2018).

133. ***Merulina laxa* Dana 1846**: Reddiah (1977) recorded from the A&N Islands, is a synonym of *Hydnophora rigida* (Dana, 1846) as of Hoeksema and Cairns (2018).

134. ***Mycedium tubifex* (Dana, 1846):** Pillai (1967) reported from the GoMBR, is a synonym of *Mycedium elephantotus* (Pallas, 1766) as of Hoeksema and Cairns (2018).

MEANDRINIDAE

135. ***Dichocoenia stokesii* Milne Edwards and Haime, 1848**: Mondal *et al.,* (2011) recorded the occurrence of this species in the Andaman Sea. However, this is an Atlantic species (Hoeksema and Cairns, 2018, and IUCN 2008). Hence, we excluded this record in the present checklist.

MONTASTRAEIDAE

136. ***Montastraea salebrosa* (Nemenzo, 1959)**: Raghuraman *et al.,* (2012) and Mondal *et al.,* (2013) reported from the A&N Islands, is previous combination and wrong genus spelling of *Paramontastraea salebrosa* (Nemenzo, 1959) as of Hoeksema and Cairns (2018).

137. ***Montastrea annularis* (Ellis and Solander, 1786):** Mondal *et al.,* (2013) recorded from the A&N Islands and Krishnan *et al.,* (2018) from GoMBR, is s previous combination, wrong genus spelling of *Orbicella annularis* (Ellis and Solander, 1786). This species is native Atlantic

species (Hoeksema and Cairns 2018, and IUCN 2008) and erroneously reported from the Andaman Sea and GoMBR. Hence, it is excluded from this checklist.

138. ***Montastrea annuligera* (Milne Edwards and Haime, 1849):** Mondal *et al.,* (2013) from the A&N Islands, is a previous combination and wrong genus spelling of *Astrea annuligera* Milne Edwards and Haime, 1849 as of Hoeksema and Cairns (2018).

139. ***Montastraea cavernosa* (Linnaeus, 1767)**: Mondal *et al.,* (2011) recorded from the Andaman Islands. However, this species is native to the Atlantic Ocean (Hoeksema, and Cairns, 2018, and IUCN 2008). Therefore, excluded in the present checklist.

140. ***Montastrea colemani* Veron, 2000**: This species reported by Raghuraman *et al.,* (2012) from the A&N Islands and Krishnan *et al.,* (2018) from GoMBR, which is original combination, basionym and wrong genus spelling of *Favites colemani* (Veron, 2000) as of Hoeksema and Cairns (2018).

141. ***Montastrea curta* (Dana, 1846):** Caeiro (1999) reported from the Lakshadweep, Turner *et al.,* (2009) reported from the A&N Islands, which is a previous combination and wrong genus spelling of *Astrea curta* Dana, 1846 as of Hoeksema, B., and Cairns, S. (2018).

142. ***Montastrea magnistellata* Chevalier, 1971:** Caeiro (1999) from the Lakshadweep, Mondal *et al.,* (2013) from the A&N Islands, is original combination, basionym and wrong genus spelling of *Favites magnistellata* (Chevalier, 1971) as of Hoeksema and Cairns (2018).

143. ***Montastrea valenciennesi* (Milne Edwards and Haime, 1849):** Raghuraman *et al.,* (2012) and Mondal *et al.,* (2013) documented from the A&N Islands, Venkataraman and Rajan (2013) and Krishnan *et al.,* (2018) from Palk bay and Sreenath *et al.,* (2015) from the Lakshadweep, is previous combination, wrong genus spelling of *Favites valenciennesi* (Milne Edwards and Haime, 1849) as of Hoeksema and Cairns (2018).

MUSSIDAE

144. ***Favia albidus* Veron, 2000:** Raghuraman *et al.*, (2012) and Mondal *et al.,* (2013) noted from the A&N Islands and Venkataraman and Rajan 2013 reported from Palk Bay, is an original combination, basionym, wrong spelling of *Dipsastraea albida* (Veron 2000) as of Hoeksema and Cairns (2018).

145. ***Favia danae* Verrill, 1872:** Mondal *et al.,* (2015, 2019) noted in the A&N Islands, which is synonym of *Dipsastraea danai* (Milne Edwards, 1857) as of Hoeksema and Cairns (2018).

146. ***Favia danai* (Milne Edwards, 1857):** Mondal *et al.,* (2013) reported from the A&N Islands, is an original combination, and the basionym of *Dipsastraea danai* (Milne Edwards, 1857) as of Hoeksema and Cairns (2018).

147. ***Favia fragum* (Esper, 1795):** Mondal (2015) included this species from Andaman water. Nevertheless, this is an Atlantic species (IUCN, 2008). Hence, not included in the present checklist.

148. ***Favia favus* (Forskål, 1775):** Pillai (1967) from GoMBR, Pillai (1971) from Lakshadweep, Pillai (1972) from the A&N Islands, is the previous combination of *Dipsastraea favus* (Forskål, 1775) as of Hoeksema and Cairns (2018).

149. ***Favia helianthoides* Wells, 1954:** Mondal *et al.,* (2011) reported from the A&N Islands, is an original combination, and the basionym of *Dipsastraea helianthoides* (Wells, 1954) as of Hoeksema and Cairns (2018).

150. ***Favia lacuna* Veron, Turak and DeVantier, 2000*:*** Satyanarayana and Ramakrishna (2009) reported from GoK, and Ramakrishna *et al.,* 2010; Venkataraman *et al.,* (2012) reported from the A&N Islands, is original combination, basionym of *Dipsastraea lacuna* (Veron, Turak and DeVantier, 2000) as of Hoeksema and Cairns (2018).

151. ***Favia laxa* (Klunzinger, 1879)**: Raghuraman *et al.*, (2012) from the A&N Islands, is the previous combination of *Dipsastraea laxa* (Klunzinger, 1879) as of Hoeksema and Cairns (2018).

152. ***Favia lizardensis* Veron, Turak and DeVantier, 2000:** Raghuraman *et al.*, (2012) and Mondal *et al.,* (2013) documented from the A&N Islands, is original combination, the basionym of *Dipsastraea lizardensis* (Veron, Pichon, and Wijsman-Best, 1977) as of Hoeksema and Cairns (2018).

153. ***Favia marshae* Veron, 2000:** Mondal *et al.,* (2013) recorded from the A&N Islands, is an original combination, the basionym of *Dipsastraea marshae* (Veron, 2000) as of Hoeksema and Cairns (2018).

154. ***Favia matthaii* Vaughan, 1918:** Raghuraman *et al.,* (2012) and Mondal *et al.,* (2013) reported from the A&N Islands, is an original combination, the basionym of *Dipsastraea matthaii* (Vaughan, 1918) as of Budd *et al.,* (2012) and Hoeksema and Cains (2018).

155. ***Favia maxima* Veron, Pichon and Wijsman-Best, 1977:** Raghuraman *et al.,* (2012) and Mondal *et al.*, (2013) reported from the A&N Islands, is the original combination, basionym

of *Dipsastraea maxima* (Veron, Pichon, and Wijsman-Best, 1977) as of Hoeksema and Cairns (2018).

156. ***Favia pallida* (Dana, 1846):** Matthai (1924) from A&N, Pillai (1967) from GoMBR, Pillai 1971 from the Lakshadweep, is an original combination, the basionym of *Dipsastraea pallida* (Dana, 1846) as of Hoeksema and Cairns (2018).

157. ***Favia rotundata* (Veron, Pichon and Wijsman Best, 1977):** Turner *et al.*, (2009) recorded from the A&N Islands, is the previous combination of *Favites rotundata* Veron, Pichon and Wijsman-Best, 1977 as of Hoeksema and Cairns (2018).

158. ***Favia rotumana* (Gardiner, 1899)**: Tikader *et al.,* (1986), Venkataraman *et al.*, 2012 and Mondal *et al.,* (2013) reported from the A&N Islands, is the previous combination of *Dipsastraea rotumana* (Gardiner, 1899) as of Hoeksema and Cairns (2018).

159. ***Favia speciosa* (Dana, 1846):** This record included by Pillai (1967) from GoM, Pillai (1971) from Lakshadweep, Reddiah (1977), Mondal *et al.,* (2013) from the A&N Islands. However, this is the previous combination of *Dipsastraea speciosa* (Dana, 1846) as of Hoeksema and Cairns (2018).

160. ***Favia stelligera* (Dana 1846)**: This species reported by Pillai (1967) from GoMBR, Tikader *et al.,* (1986), as Mondal *et al.,* (2013) from the A&N Islands, is the previous combination of *Goniastrea stelligera* (Dana, 1846) as of Huang *et al.*, (2014) and Hoeksema and Cairns (2018).

161. ***Favia trancatus* Veron, 2000**: Turner *et al.*, (2009) noted the occurrence of this species from the A&N Islands, is an original combination, basionym, wrong spelling of *Dipsastraea truncata* (Veron, 2000) as of Budd *et al.,* (2012) and Hoeksema and Cairns (2018).

162. ***Favia valenciennesi* (Milne Edwards and Haime, 1849):** Pillai (1972) reported from the A&N Islands and GoMBR; also, Pillai (1989) from Lakshadweep, is the previous combination of *Favites valenciennesi* (Milne Edwards and Haime, 1849) as of Hoeksema (2014).

163. ***Favia veroni* Moll and Best, 1984:** Mondal *et al.,* (2013) recorded from the A&N Islands, is an original combination, the basionym of *Dipsastraea veroni* (Moll and Best, 1984) as of Budd *et al.,* (2012) and Hoeksema and Cairns (2018).

164. ***Mussa angulosa* (Pallas 1766):** Reddiah 1977 reported this species from the Andaman. Nevertheless, distribution of this species restricted to the Atlantic (IUCN, 2008). So, we excluded this record from this checklist.

165. ***Mussismilia braziliensis* (Verrill, 1868):** Mondal *et al.*, (2015) recorded from the A&N Islands. However, this species is native to Brazilian water (IUCN, 2008). Hence, we excluded this species from the present checklist.

166. ***Mycetophyllia danaana* Milne Edwards and Haime, 1849:** Ramakrishna *et al.*, (2010) misidentified this species from the Andaman and Nicobar Islands. However, this species is native to Western central Atlantic (IUCN, 2008). Therefore, not listed in this checklist.

167. ***Mycetophyllia lamarckiana* Milne Edwards and Haime, 1848:** Mondal and Raghunathan (2016) reported from Andaman water but, distribution of this species restricted to Western central Atlantic region (IUCN, 2008). Therefore, not included in this checklist.

168. ***Scolymia australis* (Milne Edwards and Haime, 1849):** Mondal *et al.*, (2011) recorded this species from the A&N Islands, which is the previous combination *Homophyllia australis* (Milne Edwards and Haime, 1849) as of Hoeksema and Cairns (2018).

169. ***Scolymia cubensis* (Milne Edwards and Haime, 1849):** Ramakrishna *et al.*, (2010) erroneously included this species from the Andaman Islands. However, this is an Atlantic species (Hoeksema and Cairns, 2018).

170. ***Scolymia vitiensis* Brüggemann, 1877:** This species documented from the A&N Islands by Venkataraman *et al.*, (2012) and Mondal *et al.*, (2013) respectively, but, is original combination, basionym a synonymy of *Lobophyllia vitiensis* (Brüggemann, 1877) as of Huang *et al.*, (2016) and Hoeksema and Cairns (2018).

171. ***Colpophyllia natans* (Houttuyn, 1772):** Mondal *et al.*, (2011) erroneously reported from Andaman. It is native Atlantic species (IUCN, 2008). Therefore, not included in this checklist.

OCULINIDAE

172. ***Galaxea hexagonalis* (Milne Edwards and Haime, 1848):** Pillai (1967) reported from the Lakshadweep, is a synonym of *Galaxea fascicularis* (Linnaeus, 1767) as of Hoeksema and Cairns (2018).

173. ***Lophelia investigatoris* Alcock 1898: Alcock (1893, 1898) reported from** off Konkan coast and Kerala coast, is a synonym of *Madrepora oculata* (Linnaeus, 1758) as of Hoeksema and Cairns (2018).

POCILLOPORIDAE

174. ***Pocillopora danae* Verrill, 1864:** Raghuraman *et al.*, (2012) recorded from the A&N Islands, is a synonym of *Pocillopora verrucosa* (Ellis and Solander, 1786) as of Hoeksema and Cairns (2018).

175. ***Pocillopora eydouxi* Milne Edwards, 1860:** Pillai (1967 and 1971) reported from the GoMBR, Lakshadweep, Raghuraman *et al.,* (2012) from the A&N Islands, is synonymy of *Pocillopora grandis* Dana, 1846 as of Hoeksema and Cairns (2018).

176. ***Polyastra venosa* Ehrenberg, 1834:** Pillai (1967) reported from the GoMBR, is an original combination, the basionym of *Pavona venosa* (Ehrenberg, 1834) as of Hoeksema and Cairns (2018).

177. ***Seriatopora aculeate* Quelch,1886:** Mondal *et al.,* (2010) reported from the A&N Islands. It is the wrong spelling. Accepted name is *Seriatopora aculeate* Quelch,1886 as of Hoeksema and Cairns (2018).

178. ***Seriatopora guttatus* Veron, 2000**: This record was wrongly spelled by Mondal *et al.,* (2012). Accepted name is *Seriatopora guttata* Veron, 2000, as of Hoeksema and Cairn (2018).

179. ***Stylopora mordax* (Dana, 1846):** Pillai (1967) recorded from Lakshadweep, Reddiah (1977) reported from the A&N Islands, is a synonym of *Stylophora pistillata* Esper, 1797 as of Hoeksema and Cairn (2018).

PORITIDAE

180. ***Goniopora duofaciata* Thiel, 1932**: Pillai (1967) reported from the GoMBR, is synonymized name of *Goniopora planulata* (Ehrenberg,1834) as of Hoeksema and Cairns (2018).

181. ***Goniopora minor* Crossland, 1952*:*** Pillai (1971) described from the Lakshadweep, Mondal *et al.,* (2013) reported from the A&N Islands, is synonymy of *Goniopora pedunculata* Quoy and Gaimard, 1833 as of Veron and Pichon (1982) and Hoeksema and Cairns (2018).

182. ***Goniopora nigra* Pillai, 1967:** Pillai (1967) reported from the GoMBR, which is synonymy of *Bernardpora stutchburyi* Wells, 1955 as of Veron and Pichon (1982) and Hoeksema and Cairns (2018).

183. ***Porites (Synaraea) convexa* (Verrill, 1864):** Pillai (1989) recorded this species from the Lakshadweep, which is synonymy of *Porites rus* (Forskål, 1775) as of Veron and Pichon (1982) and Hoeksema and Cairns (2018).

184. ***Porites andrewsi* Vaughan, 1918:** Pillai (1967) recorded from the Lakshadweep, is synonymy of *Porites cylindrica* Dana, 1846 as of Veron and Pichon (1982) and Hoeksema and Cairns (2018).

185. ***Porites cumul*ates Nemenzo, 1955:** Mondal *et al.*, (2016) reported from the A&N Islands, is the wrong spelling of *Porites cumulatus* (Nemenzo, 1955) as of Hoeksema and Cairns (2018).

186. ***Porites eridani* Umbgrove, 1940***:* Tikader *et al.*, (1986) reported from the A&N Islands, is synonymy of *Porites lichen* Dana, 1846 as of Veron and Pichon (1982) and Hoeksema and Cairns (2018).

187. ***Porites tenuis* Verrill, 1866:** Reddiah (1977) reported from the A&N Islands, is synonymy of *Porites lutea* (Quoy and Gaimard, 1833) as of Scheer and Pillai (1983) and Hoeksema, B., and Cairns, D. (2018).

188. ***Porites lutea* (Quoy and Gaimard, 1833):** This species reported by Pillai and Patel, 1988 from GoK, Pillai, 1967 and 1971 from GoMBR, Reddiah, 1977 from Andaman, has been now changed to *Porites lutea* Milne Edwards and Haime, 1851 by Hoeksema and Cairns (2018).

189. ***Porites porites* (Pallas, 1766)**: We excluded this species from the checklist. Reddiah (1977) and Mondal *et al.*, (2010) reported this species from the Andaman Islands, but this is a native Atlantic species distributed in Mozambique, Nicaragua, Jamaica species (Hoeksema and Cairns).

PSAMMOCORIDAE

190. ***Psammocora exesa* Dana, 1846:** This species was reported by Pillai (1971) from the Lakshadweep, which is synonymy of *Coscinaraea exesa* (Dana, 1846) as of Hoeksema (2014) and Hoeksema and Cairns (2018).

191. ***Psammocora explanulata* Van der Horst, 1922:** Turner *et al.,* (2009) recorded from the A&N Islands, is synonymy of *Cycloseris explanulata* (Van der Horst, 1922) as of Benzoni *et al.,* (2012) and Hoeksema and Cairns (2018).

192. ***Psammocora haimeana* Milne Edwards and Haime, 1851**: Pillai (1971) reported from the Lakshadweep and Turner *et al.,* (2009) from the A&N Islands was wrong spelling. The new

spelling is known as *Psammocora haimiana* Milne Edwards and Haime, 1851, as of Hoeksema and Cairns (2018).

193. ***Psammocora obtusangula* (Lamarck, 1816):** This species recorded by Raghuraman *et al.,* (2012) from the A&N Islands, is synonymy of *Psammocora contigua* (Esper, 1794) as of Hoeksema and Cairns (2018)

194. ***Psammocora superficialis* Gardiner, 1898*:*** Turner *et al.*, (2009) listed from the A&N Islands, is synonymy of *Psammocora profundacella* Gardiner, 1898 as of Benzoni *et al.,* (2010) and as reported in Hoeksema and Cairns (2018)

SCLERACTINIA INCERTAE SEDIS

195. ***Solenastrea bournoni* Milne Edwards and Haime, 1849:** This species was reported by Ramakrishna *et al.,* (2010) and Mondal *et al.,* (2010a) from the Andaman Islands. However, this species occurs in the Atlantic Ocean (IUCN, 2008). Hence, it is not included in this checklist.

SIDERASTREIDAE

196. ***Siderastrea liliacea* Klunzinger, 1879:** This species was reported by Pillai (1972) from GoMBR, which is synonymy of *Pavona clavus* (Dana, 1846) as of Hoeksema and Cairns (2018).

197. ***Siderastrea radians* (Pallas, 1766):** This species reported in the Indian water by Pillai (1967) and Raghuraman *et al.,* (2012) from GoMBR and Andaman, respectively. However, distribution of this species restricted to the Atlantic (IUCN, 2008). Therefore, it is not included in this checklist.

198. ***Siderastrea siderea* (Ellis and Solander, 1786):** Mondal *et al.,* (2011) reported this species from the Andaman Islands. Moreover, this species native to the Western Central Atlantic Ocean (IUCN, 2008). Hence, not listed in the present checklist.

Chapter 3

3. Study Area: The Malvan Marine Sanctuary

3.1. Geographic features

The coastline of Maharashtra stretches ~720 km along the eastern Arabian Sea (Bhatt & Bhargava, 2006). Towards the southern end of Maharashtra's coastline lies the Sindhudurg district, and the Malvan coast (35km) lies within the Sindhudurg district, at latitude 15°58′ N and longitude 73°30′ E and forms part of the Western Ghats where the Sahyadri ranges gradually meet the Arabian Sea with the Sindhudurg plateau in between (Source: district socio-economic report of Ratnagiri and Sindhudurg district, 2012-13). Malvan or Malwan is an urban town, one of the eight talukas of the Sindhudurg district, located approximately 35km distance from the Mumbai-Goa National Highway connected by a coastal highway that joins Mumbai with Panaji. This town is bound by three small creeks viz., Karli, Kolamb, and Kalavli. Karli River flows in the south Malvan coast, and Kalavli and Kolamb rivers flow in the north.

Sahyadri hills have a mean sea level of over 200m at the bottom, while at the upper riches, they attain a height of about 700m (Munjal, 2019). The region is hilly dissected with transverse ridges of Western Ghats and at many places extending as promontories into the Arabian Sea (Ahmed, 1972). The shoreline in the area is very irregular and narrow towards the south and progressively widens towards the north and is very erratic and associated with erosional and depositional features (Chandramohan *et al.*, 1993). This region is drained mainly by parallel westward flowing streams. Extending in a north-south direction through the central portion of the district, it has a transverse chain of small hills, projecting from the Sahyadri hills, and developing a higher elevation in the middle parts. The continental shelf in this region is relatively wide and seabed sediment up to the 100m generally consist of silty clay (Chandramohan *et al.*, 1993).

The Malvan coast is developed on a basement plate of basalt flows of the Deccan volcanic province (Kumaran *et al.*, 2004). The rocky coast of Malvan mainly features granites and biotite gneisses, and dissected mainland with lava promontories, are covered by 8-12m thick lateritic beds (Deshpande & Pitale, 2014; Kumaran *et al.*, 2004). There are the occasional presence of overhanging cliffs, projecting headlands, stacks and erosion platforms, rocky shoals, several submerged reefs, and boulders (Venkataraman *et al.*, 2004).

On the north of Malvan, the littoral concrete or ‘beach rock’ occurs as rocky beach either directly attached to the mainland or separated from the latter by a zone of sandy beach or muddy and marshy area (Deshmukhe *et al.,* 2013). The rocky outcrop reduces the wave energy and later directional distribution, which helped the formation of sandy beach or muddy swamps between the rocky shore and the mainland (Chandramohan *et al.,* 1993). The coastline in Malvan is marked by a chain of islands like Sindhudurg at the northern tip, Vengurla Rocks at the Southern tip, apart from several small islands like Kawaddya Dongar, Padmaged, Mandel Rock, Malvan Rock, and Kedebakal present. This coastline also features rocky shore like Rajkot and Sargikot, sandy beaches viz. Chiwalaychi Vel, Kandvel, Malvan Chowpatty, and Dandi and mudflat and mangroves habitat at Kalavali and Kolamb creeks. There are numerous exposed rocky outcrops in this area (ICMAM, 2001).

The seasonal variation in wave characteristics and tidal waves, their intensity and frequency, approach, height, and persistence were observers along the coast (Karlekar & Thakurdesai, 2017). Malvan is located along the Konkan coast experiences mixed semidiurnal tides, ranges from 2-3.5m (Karlekar & Thakurdesai, 2017). Earlier, Reddy (1976) recorded the tidal ranges of 1-2m, and the predominant wave directions are from southwest, west southwest, and northwest (period ranging from 5-14 seconds). Tidal currents are weak along this coastline, and the velocity rarely exceeds 10 cm/s (Karlekar & Thakurdesai, 2017). The average breaking height of the wave changes from season to season (0.3m in winter and 2.2m in monsoon) wave breaks parallel to the coast, and the wave height and energy is high during monsoon (Chandramohan *et al.,* 1993). This region experiences winds of 5-11 knots in the pre-monsoon, 10-14 knots during monsoon, and 5-8 knots during the post-monsoon season (Karlekar & Thakurdesai, 2017). The currents are south-eastward during monsoon with an average speed of 30 to 40 cm/s, and the rest of the time, these usually are north- north-westward, with a velocity of 8 to 20 cm/s (Karlekar & Thakurdesai, 2017). The longshore current directions are variable in the area, but it is predominantly more towards the south (Chandramohan *et al.,* 1993). The sea condition becomes rough to very rough during the monsoon season. Moderate to massive swell waves also persist along the coast, the waves are steeper near the shore, and the breakers have short wave periods during monsoon. The width of the surf zone and the breaker zone decreases considerably during fair weather season, when the height of the breakers and the number of waves in a breaker decrease significantly. Additionally, during monsoon period, the quantum of sediment input to this coastal water through the streams is very high (Karlekar & Thakurdesai, 2017).

3.2. Climatological features

The tropic climate in the study area is generally moist and humid, and the temperature variations during the day and throughout the seasons are not significant (Munjal 2019). Relative humidity ranges from 72 % in monsoon, 86 % in monsoon, and 82 % in the post-monsoon season (Karlekar & Thakurdesai, 2017), indicating its maxima during monsoon months. There are three seasons viz. summer season from March to May, June to October is the season of southwest monsoon with high rainfall and winter season from November to February marked by light winds and occasional tropical cyclones during this period. In this region, the mean temperature ranging between 16 ^{0} to 36^{0} C. December to March is relatively cold with northeast winds. The weather is dry, and the cloud cover is minimal. The temperature rises slowly from March to May, and April and May are the hottest months. In monsoon months, the temperature drops by 3-4^{o}C, and day temperature increases in October and November (Fig.3.1). Temperatures are the lowest in January-February.

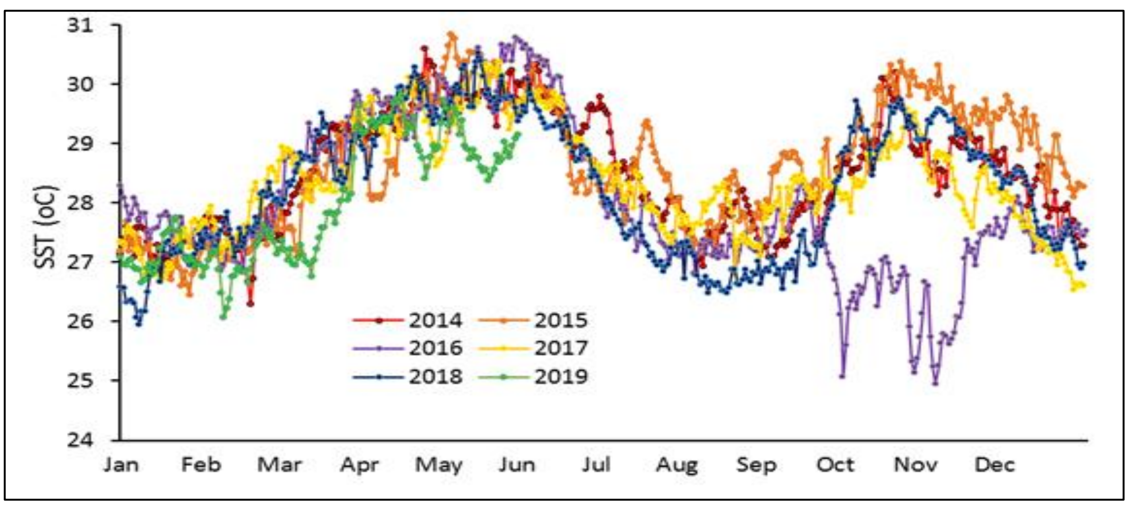

Fig. 3.1: Annual variation of Sea Surface Temperature (SST) during the study period from 2014 to May 2019 (data courtesy NOAA OISST dataset)

From June to October, the season of the southwest monsoon, this region receives the heavy rainfall because of the strong influence of the Arabian Sea and the Sahyadri hilly ranges (Fig.3.2). The average decadal annual rainfall in the district was about 3151mm (IMD, 2019). Due to intensive precipitation events, coastal water of Malvan receives high runoff from land, usually from river discharges, and resuspension of sediment. Though the average rainfall is rather high, this region faces water scarcity in the summer season, as most of the rainwater drained off to the adjacent Arabian Sea due to undulating topography and porous nature of geomorphology, lack of water holding capacity of the soil and lack of any water impounding and storage structures.

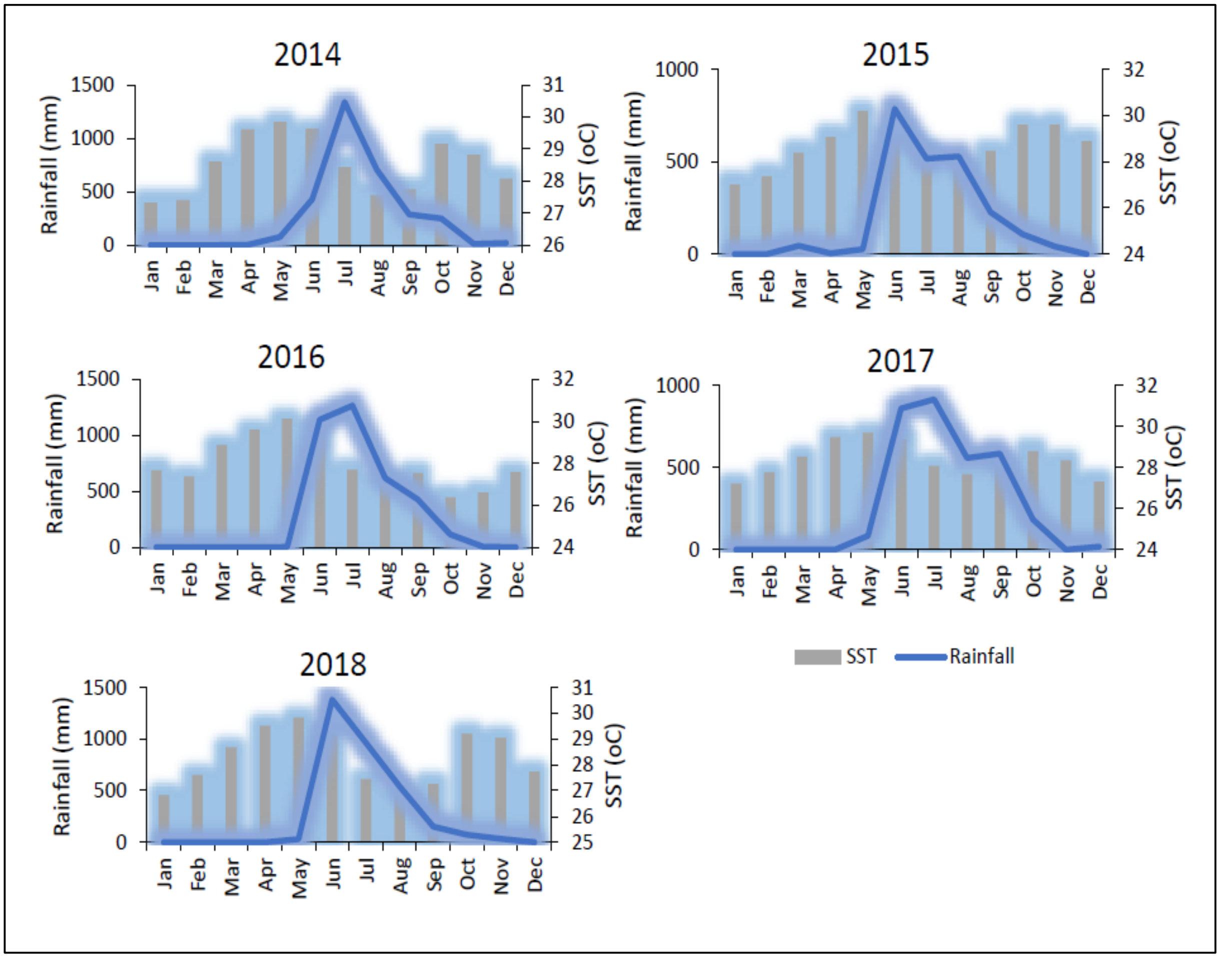

Fig. 3.2: Trends in annual average rainfall and SST of Sindhudurg in millimeters during the study period from 2014 to 2018 (data courtesy Hydromet Division, IMD for rainfall data and NOAA for SST data)

3.3. Ecological features

The Sindhudurg coast is identified as one of the richest in biodiverse and habitat types along the coast of Maharashtra. Amongst that, the Malvan Marine Sanctuary is notable for its unique coastal and marine biodiversity. The ecological significance of the coastal and marine resources of the Malvan coast has been documented dating as far back as 1947 (MacDonald, 1947). The importance of biological diversity of the MMS was recorded in 1980 by the National Institute of Oceanography (NIO), this was further confirmed by a detailed study by the ICMAM and declared Malvan as a 'Biodiversity Hotspot" (ICMAM, 2001), and was designated as one of the most biologically diverse areas of Maharashtra (NIO, 1980). This region included critical habitats like the rocky and sandy shore, rocky islands, estuaries, mudflats, marshy land, mangrove habitats, coral reefs, macroalgal bed, as well as congregation sites for various economically important fish and higher predators like sharks.

Previous studies highlighted relatively rich biodiversity in the MMS (Parulekar, 1981; ICMAM, 2001), and it is also considered as the biodiversity hotspot of the Central West coast of India. While preparing a detailed inventory Parulekar (1981) reported that the coral reef located in the core area of MMS harbours 74 species of fishes, 73 species of seaweed, and nine coral species, including 181 other associated flora and fauna. Other than these few studies, there are very few reports on the presence of hard corals in MMS. Qasim and Wafar (1979) reported living corals from the intertidal region of the Malvan along with two other locations viz. Ratnagiri and Redi. Furthermore, Qasim and Wafer (1979) enlisted nine coral species; Parulekar (1981) also noted the occurrence of nine species of coral, and in ICMAM report, 2001 documented 11 genera of coral. Complex and diverse coral reef systems are thought to be absent from this region because of dramatic changes in the salinity and turbidity of these waters especially during the monsoon season. Absence of branching corals from this region is thought to be because of strong wave action in this region. Nevertheless, the discovery of other genera of corals (e.g. *Porites)* indicate their adaptability to a wide range of environmental fluctuations.

The soft sedimentary rocks of the Malvan coast are soft and easily eroded by the strong wave and wind energy, and forms many crevices and cracks. These crevices and cracks are ideal for sheltering, feeding, and breeding grounds for many burrowing invertebrates and also serve as settlement substratum for marine algae (ICMAM, 2001). The MMS is situated in the Malvan bay, a protected coastal ecosystem by series of consolidated rocky outcrops. Submerged and exposed rocks protect reef corals from intense wave action and endow with ideal substratum for settlement. Most of the coral species have grown in patches on rocky foreshore and subtidal reef flat. It is a shallow subtidal reef, corals in upper reef margin get exposed during the low neap tide, and average depth during high tide is 5-7m. Most notably, elevated SST during summer months, low light penetration due to high sedimentation rate, and intense wave action during monsoon season resulted in surviving only stress-resistant coral species in this area.

Due to its high biodiversity and ecological importance, an area of 29.12 square kilometers of Malvan coastal waters (the core zone of 3.182km^2 and buffer zone is comprising with an area of 25.94 km^2 between 16^0 02'00 N-$16^0$03'90 N and $73^0$25'00 E-$73^0$29'25 E) was designated as the Malvan Marine Sanctuary (MMS) in 1987, under the National Wildlife (Protection) Act, 1972. Among the thirty-one marine and protected coastal areas in India, only seven protected areas can be considered as true representatives of Marine Protected Areas. The Gulf of Kutch Marine National Park and the Gulf of Kutch Marine Sanctuary (Gujarat), Malvan Marine Sanctuary in Maharashtra situated on the west coast of India. On the other hand Gulf of Mannar National Park in Tamil Nadu, Gahirmatha National Park in Orissa, and the Mahatma Gandhi Marine National Park on the east coast and the Rani Jhansi Marine National Park in Andaman & Nicobar Islands (http://pib.nic.in/release/release.asp?relid=32348).

Further, under the Integrated Coastal and Marine Area Management (ICMAM), Govt. of India identified Malvan among the 11 ecologically and economically critical habitats along the Indian coast (ICMAM, 2001). In their 2001 report mentioned the presence of 367 species of marine flora and fauna in the Malvan coast belonging to 173 genera, 97 families, 16 classes, and nine phyla based on the available literature, among the biota the report noted the presence of 73 species of marine algae, 18 species of mangrove trees, nine species of coral, 70 species of mollusks, 47 species each of polychaetes and arthropods, 18 species of sea anemones and 74 species of fish based on earlier report and during the study. However, the ICMAM study found 297 species in the MMS, included 58 species of phytoplankton, nine group of zooplankton, 33 species of foraminiferans, 11 species of Corals, 39 species of benthic organisms include 15 species of polychaetes, 12 species of Gastropods, seven species of Crustaceans, four species of Bivalves, 15 metazoan groups dominated by the nematodes and 32 species of Seaweeds

(ICMAM, 2001). Sittings of Whale shark, a species listed under Schedule I of India's Wildlife (Protection) Act, 1972, were also reported in the MMS. Other important species including Rays, Seahorses, three species of Sea snakes, cetacean like the Blue whale, Bryde's whale, Indo-pacific humpback dolphins, and Indo Pacific finless porpoise have been sighted along the Malvan coast (UNDP, 2011; Jog *et al.*, 2015; Patil *et al.,* 2016; Rao & Muralidharan, 2019). Further, three globally significant species of turtles, namely, Olive Ridley (*Lepidochelys olivacea*), Green (*Chelonia mydas*) and Leatherback (*Dermochelys coriacea*), have been reported from the district. Besides, the avifauna of the area is also abundant, with 121 species, including 66 residents, 24 migrants, and 28 residents population (UNDP, 2011).

Earlier, Barman *et al.,* (2007) surveyed the fish diversity in the MMS and documented 108 fish species belonging to 48 families and 13 orders, included four vulnerable and two near-threatened species. Tike *et al.,* (2009) reported the occurrence of 29 fish species belonging to 28 genera of 22 families, 19 species of Crustaceans belonging to 14 genera and eight families, and 77 species of Molluscan fauna representing 56 genera and 37 families. Further supported by the study on the zooplankton community in Malvan revealed higher zooplankton abundance and biomass, which might support high fish production (D'Costa & Pai, 2019).

However, detailed assessment on the health status of reef-forming corals in the MMS in the era of rapidly growing coastal pressure and changing climate is need of the hour as due to limited economic opportunities, local people in and around Malvan are forced to depend on the coral reef. The MMS is relatively more accessible for recreational activates than the other coral reefs of India, mainly due to the vicinity of mainland India. Hence, tourism activity related to the beautiful coral reef is booming in MMS, and there is a significant unplanned urban development to meet the needs of the booming tourism in MMS. Therefore, the presence of higher species diversity demands urgent ecological studies in this region, which will aid in developing an action plan for proper conservation and sustainability of this fragile ecosystem.

3.4.Economical features

The marine and coastal regions of India play a vital role in the nation's economy by the vast resources, habitats, and rich biodiversity. Marine fisheries (India is the 3rd largest producer of fish in the world), harbors, aquaculture, agriculture, tourism, oil, and mineral exploitation-contribute about 10% of the national Gross Domestic Products (GDP) (Source: Planning Commission, Government of India).

The coastal area of Malvan is thickly populated with a large population living along the coast, and depend on the sea for their livelihood. In Malvan, population density is 190, and 35.49% of the population were below the poverty line, whereas, during 2005-06 per capita income of the Sindhudurg district was INR 32,862 against the state average of INR 42,056 (UNDP 2011). Fishing is one of the primary occupations, and the principal economic activity of the population of the Malvan region and Malvan is a major fish landing center in Sindhudurg (Munjal, 2019) and of economic importance on the west coast of Maharashtra (Tike *et al.*, 2009). The number of marine fishing villages in the Malvan region during the period of 1997-2003 was 29 (Source: census of marine fishermen of Maharashtra state, 1997 and 2003). Fishery plays a vital role in the local minor fishery. Minor fisheries play an important for the local poor for livelihood and local economy, wherein women performed a significant role in this activity (Tike *et al.*, 2009).

Fishing practice within the territorial waters has been conducted by using trawlers, fiber-glass boats, and the fishing gears are mainly composed of trawl nets, gillnets, hooks, and line, and Rampans or the traditional fishing gears (UNDP, 2011). There were 1,068 fishing vessels, which include 186 mechanized vessels, 390 motorized vessels, and 492 nonmotorized vessels during 2005-06 (UNDP, 2011; CMFRI, 2006). The. Mechanized fishing vessels undertake one-day fishing operations and are anchored in the Sindhudurg Fort area. As per the census of marine fishers' population 1997 and 2003, the active fishermen population living in Malvan was 2271 and 2660, respectively, and a population of 7951 and 9429 dependent on fisheries and allied sector. However, in 2010, the survey revealed the presence of only 2897 fishers in the Malvan region, and 5739 people are dependent on fishing and allied work (UNDP, 2011). Fish processing is one of the related activities in marine fishery (UNDP, 2011). Many fishing communities are low-income groups, and they do not have access to formal education. Essential services like water, electricity roads, medical facilities, housing, banking, etc., are not available to fishing communities (Munjal, 2019).

Malvan attracts the highest inflow of tourists in the district, indicating the increasing attraction of people towards the Sindhudurg Fort and beautiful sandy beaches around the Malvan coast. The iconic Sindhudurg Island Fort built by the Maratha King Shivaji during in the year 1656 by

Chhatrapati Shivaji Maharaj (The Maratha King) during the peak of his Maratha Empire for protection of the fleets and as a stronghold in adversity. The fort is constructed on the island called Kurte, about 1.6km offshore from the mainland Malvan. Currently, the fort is partly dilapidated as the western outer wall of the fort has been damaged as a result of wave action and weather events.

The Konkan region on the west coast is internationally acclaimed for its sun and sand (Sathe & Chauhan, 2003). Apart from the beautiful beaches and natural beauty, ancient forts, old temples, unique culture, distinctive cuisine, the coast is also well known for its mangoes, cashew nuts, and Kokam, which in turn attracts a lot of tourists and traders. Sindhudurg was declared the country's first ecotourism district in 1997 (Munjal, 2019). According to month-wise tourist arrivals in Sindhudurg district, December and January are the peak months for foreign and domestic tourists. However, domestic tourists flow is higher during the first half of the year, and the small foreign tourists flow reported only during the winter months (Munjal 2019). Tourist footfall in Sindhudurg district estimated 200,000 in 2001-2002 and projected to be 641,427 in 2021-22 with an overall increase of 6% (Source: District tourism master plan Sindhudurg, Maharashtra year 2013-2033; Munjal, 2019).

Apart from the marine fishery, eco-tourism is the fastest growing economic sector associated with coral reefs in this region. Malvan is renowned for coastal cuisine, local handicraft, old temples, water sports, and scuba diving. However, due to the lack of a management plan and opposition of the local community, the MMS is not operational MPA in the true sense (Rajagopalan, 2008). I recent times, tourism has helped with new employment and income generation for local coastal communities in Malvan, and most of the benefits of tourism are percolating to local communities due to the absence of the big industrial houses. Around 3000 people are involved in the tourism business, the estimated annual revenue from tourism is about $2.5 million, and the primary beneficiaries are the local coastal communities that were earlier involved in fishing (UNDP, 2011).

Chapter 4

4. Scleractinians diversity in the Malvan Marine Sanctuary

4.1. Introduction

Coral reefs are one of the most important ecosystems in the tropical seas. Coral reefs provide valuable ecosystem services and goods, such as habitat formation (Messmer *et al.,* 2011), secondary production (Appeldoorn *et al.,* 2009), carbon, and nitrogen-fixing (Casareto *et al.*, 2008), shoreline protection (Maragos *et al.*, 1996). Scleractinians are one of the five orders under the subclass Hexacorallia (Phylum Cnidaria) and generate a rigid skeleton made of calcium carbonate ($CaCO_3$). They depend on symbiotic single-celled dinoflagellate zooxanthellae for nutrition. They form coral reef in shallow tropical seas and cater great diversity of life.

India is blessed with four major coral reefs in the Gulf of Kachchh, Lakshadweep archipelago, Gulf of Mannar, and the Andaman and Nicobar Islands. Apart from these reefs, there are several newly reported small reefs surrounding islands like Netrani Island in Karnataka coast (Zacharia *et al.*, 2008), Grande islands in Goa coast (Rodriques,1998), Gavesani bank in off Malpe coast (Nair, 1978), Angria bank in off Vijaydurg coast (Ghose & Fernandes, 2014, Ingole *et al.*, 2015) and the Malvan Marine Sanctuary (MMS). Despite the occurrence of coral reef, little attention has been received to evaluate the biodiversity and ecology of these reefs.

The MMS is the only Marine Protected Area (MPA) in the Central West Coast of India of the Eastern Arabian Sea. The MMS significantly contributes to the livelihood of the local fishermen population. Apart from the marine fishery, eco-tourism is the fastest growing economic sector associated with coral reefs in this region. However, due to the lack of a management plan and opposition of the local community, the MMS is not operational MPA in a real sense (Rajagopalan, 2008).

Although MMS is a known coral reef habitat, however, studies on the coral reef biodiversity and ecology in the MMS are sparse. Most of the literature is decade-old viz. Qasim & wafer

(1979), Parulekar (1981), and NIO (2000) and mostly reported coral reef biodiversity. Studies on historic coral cover and other benthic community covers, coral reef health status, occurrence, and extent of bleaching events are utterly missing from this region. Therefore, the coral reef habitat in MMS needs urgent attention on the health status of reef-forming corals during the rapidly growing coastal pressure and changing climatic conditions. Previous studies were highlighted relatively rich biodiversity in this reef (Parulekar, 1981; NIO, 2000), and reported as a biodiversity hotspot of the Central West Coast of India. The patch coral reefs are located in the core area of MPA, which harbours 74 species of fishes, 73 species of seaweed, and nine coral species, including 181 other associated flora and fauna (Parulekar, 1981). A list of species reported from the MMS region provided in table 4.4. There are very few reports on the presence of hard corals in MMS. Qasim & wafer (1979) and Parulekar (1981) reported the presence of nine species of corals. In contrast, a report by NIO (2000) documented 11 genera of corals. However, these reports are lack details taxonomical observation and distribution of coral species in the MMS. Moreover, taxonomical information is an essential requirement for understanding the diversity and functional role of an organism within the ecosystem. Hence, the present study aims to provide a comprehensive taxonomic description and diversity of scleractinians in this poorly explored MPA.

4.2. Material and Methods

4.2.1. Oceanographic settings of the study area

The MMS is a known coral reef ecosystem and is the only Marine Protected Area (MPA) in the Central West Coast of India of the Eastern Arabian Sea. The MMS is spread over 29.12square km; the core zone of 3.182km^2 and buffer zone is comprising of an area of 25.94 km^2 between 16^0 02ˀ00 N-16^{0}03ˀ90 N and 73^{0}25ˀ00 E-73^{0}29ˀ25 E. The MMS is situated in the Malvan bay; a protected coastal ecosystem by series of consolidated rocky outcrops. Submerged and exposed rocks protect reef corals from strong wave action and endow with ideal substratum for settlement. Most of the coral species have grown in patches on rocky foreshore and subtidal reef flat. It is a shallow subtidal reef, corals in upper reef margin get exposed during the low neap tide, and average depth during high tide is 5-7 meters. Most notably, elevated SST during summer months (De *et al.,* 2015), low light penetration due to high sedimentation rate, and strong wave action have been led to growing stress-resistant coral species in this area.

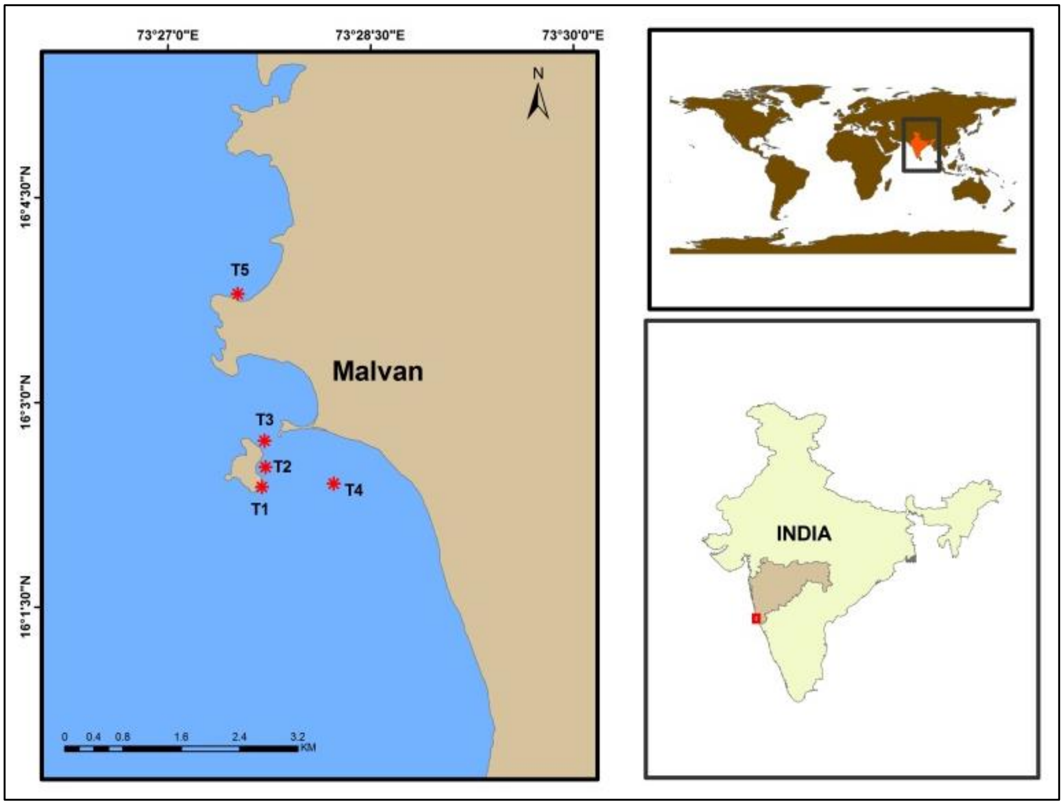

Fig. 4.1. Map showing the location of Malvan Marine Sanctuary and study sites (T1 to T5) in MMS

4.2.2. Survey methods

Annual underwater surveys were carried out using a fishing trawler from 2014 to 2019. Five sites were chosen in the reef along a distance gradient of 500m-1000m (Fig. 4.1 and Table 4.1). During each survey, 20m line transect (English *et al.,* 1997), in triplicate, were placed randomly on each location of the sub-tidal reef flat by employing Self Contained Underwater Breathing Apparatus (SCUBA) diving to assess benthic community composition. Measurements of all coral colonies were done in-situ by measuring tape. Coral cover (live and dead were recorded with digital photographs and videos, taken directly above the contact area using NikonAW120 (14 Mega Pixels) and GoPro Hero4 (12 Mega Pixels) underwater Camera. During the survey, the position fixing was carried out by using a handheld Garmin GPS MAP 78S.

Live corals were not collected for identification, because of corals are Schedule I protected species in India by Wildlife Protection Act, 1972. Hence, identification of the coral species was carried out based on morphological and macrostructural features by in situ observation and from high-resolution digital photographs using the keys described in Veron (1986, 2000) and

Venkataraman *et al.*, (2003). Photo documentation was used to verify taxonomic identification and to avoid any observer bias.

Table 4.1. Location and characteristics of underwater survey sites in the MMS.

Site	Category	Type of substratum	Geographical positions		Depths (m)	Specific Characteristic of the site
			Latitude (^{0}N)	Longitude (^{0}E)		
T1	Inshore	Rocky	$16^0$02'22.8"	$73^0$27'41.9"	2-4	Recreational diving site
T2	Inshore	Rocky	$16^0$02'31.5"	$73^0$27'43.4"	2-4	Recreational diving site
T3	Inshore	Rocky	$16^0$03'786.4"	$73^0$27'725"	2-3	Recreational diving site
T4	Inshore	Rocky & sandy	$16^0$02'24.3"	$73^0$28'13.8"	8-10	Fishing site
T5	Inshore	Rocky & sandy	$16^0$03'84"	$73^0$27'43.3"	7-9	Fishing site

4.3. Results

During the present study, a total of nineteen species of scleractinian coral belonging to fourteen genera and eight families recorded from the MMS (Fig.4.2; Table 4.2). The occurrence of *Favites melicerum, Cyphastrea serailia, Plesiastrea versipora*, *Turbinaria mesenterina, Turbinaria frondens, Porites compressa, Porites sp., Goniopora stokesi, Goniopora pedunculata, Bernardpora stutchburyi* were new record from this reef.

Phylum CNIDARIA Hatschek, 18888

Class **ANTHROZOA** Ehrenberg, 1834

Subclass **HEXACORALLIA** Haeckel, 1896

Order **SCLERACTINIA** Bourne, 1900

Family **Merulinidae** Verrill, 1865

Genus ***Favites*** Link, 1807

Species ***Favites melicerum*** Ehrenberg, 1834

Synonymy: *Astraea melicerum* Ehrenberg, 1834; *Favites melicerum* Veron, 2000

Common name: None

Description:

Colonies are sub-massive to encrusting and surface rising to hillocks; Polyps, corallites, and calices are polygonal; Corallites are polygonal and about 5-6 mm long, 4-5 mm broad and 2-3 mm deep. Total no. of septa is 24. The combined wall is thin. The Paliform lobe is present. Colonies are thick-walled and rounded, becoming subplacoid. Septa are few, which is uniform in height. Septa are in alternating order.

Colour: colonies are mostly brownish and greens.

Occurrence and distribution: This species commonly found in the Indo-West Pacific, this species is found in the central Indo-Pacific, north, west, and South Australia, Japan and the East China Sea, and the oceanic West Pacific. Madagascar (IUCN, 2015). This species described from the Gulf of Kachchh (Pillai & Patel, 1988), Gulf of Mannar (Pillai, 1986), Lakshadweep Islands (Pillai, 1971) and as *Favites bestae* from Andaman and Nicobar Islands (Mondal *et al.*, 2015).

Remarks: It generally grows in all areas of MMS. Colonies were often covered with silt, debris, and filaments on lower side boulders. The observed colony was partially bleached**.**

Genus Cyphastrea Milne Edwards and Haime, 1848

Species ***Cyphastrea serailia*** Forskal, 1775

Synonymy: *Madrepora serailia* Forskằl, 1775 (Original combination, basionyms); *Cyphastrea danai* Milne Edwards, 1857; *Cyphastrea brueggemani* Quelch, 1886; *Cyphastrea suvadivae* Gardiner, 1904; *Cyphastrea conferta* Nemenzo, 1959

Common name: Lesser knob coral

Description:

Colony massive, hillocky, and encrusted with a smooth surface. Polyps are not compacted and round-shaped with 2-3 mm in dia. Columella inconspicuous and trabecular. Crowded flowering polyps. Corallites are rounded and equal in size. Costae do not alternate. Presence of 12 primary septa.

Colour: Colour is usually uniform or mottled grey, brown, or cream.

Occurrence and distribution: This species has been reported from the Indo-West Pacific, this species is found in the Red Sea and the Gulf of Aden, the southwest and the northwest Indian Ocean, the Arabian/Iranian Gulf, the northern Indian Ocean, the central Indo-Pacific, Australia, Southeast Asia, Japan, and the East China Sea, the oceanic West Pacific, and the Central Pacific (IUCN, 2015). Presence of this species reported from the Gulf of Kachchh (Pillai & Patel, 1988), Palk Bay (Pillai, 1969), Gulf of Mannar (Pillai, 1986), Lakshadweep Islands (Pillai,1989), Andaman and Nicobar Islands (Mondal *et al.*, 2015).

Remarks: This species is present in all the stations of the MMS. This species is known to resilient to high salinity and temperature stress.

Genus **Goniastrea** Milne Edwards & Haime, 1848

Species ***Goniastrea retiformis*** (Lamarck, 1816)

Synonymy: *Astraea* (*Fissicella*) *eximia* Dana, 1846, *Astraea eximia* Dana, 1846, *Astrea retiformis* Lamarck, 1816, *Goniastrea eximia* (Dana, 1846), *Goniastrea bournoni* Milne Edwards & Haime, 1849

Description:

This species is massive or columnar. Calices are small, deep, and have thin, sharp walls, generally 3 to 4 mm diameter and four to six-sided. Septa alternate clearly, and all the first order septa bears a tall. Lobes are straight, thin, piliform, and well-developed.

Colour: Colonies are cream or pale brown, pink, or greenish.

Occurrence and distribution: This species is distributed in the Indo-Pacific in Chagos, Seychelles, Mauritius, Reunion, Kenya, Madagascar (Hoeksema, and Cairns 2020)**.** In Indian seas, this species is reported from Lakshadweep Islands (Pillai, 1971), Gulf of Mannar (Pillai, 1967), Andaman and Nicobar islands (Reddiah 1977; Venkataraman, 2012).

Remarks: The species occurs on the shallow-water reef flat and reef slopes. This species is common and usually dominant in the MMS. Colonies frequently exceed 1m in diameter. This species is known to tolerate elevated water temperatures and salinity.

Family **SCLERACTINIA INCERTAE SEDIS**

Genus ***Plesiastrea*** Milne Edwards and Haime, 1848

Species ***Plesiastrea versipora*** Lamarck, 1816

Synonymy: *Astrea versipora* Lamarck, 1816; *Favia versipora* Lamarck, 1816; *Favia ingolfi* Crossland, 1931, *Plesiastrea quatrefagiana* Milne Edwards & Haime, 1849; *Plesiastrea urvelli* Edwards and Haime, 1849.

Common name: Small knob coral

Description:

Colonies are massive, rounded, or flattened, encrusting. Corallites are monocentric and placoid, not dense. Paliform lobes form a circle around columella and are not seen clearly because of flowering tentacles. Daughter corallites are seen. Septa are short of two alternating sizes. Columella is small. Corallum infested with algae and a transparent mucous film. Small rounded shaped placoid corallites.

Colour: Colonies are in yellow, cream, or brown, usually pale colour.

Occurrence and distribution: This species widely distributed within the Indo-Pacific, occurring in the western and northern Indian Ocean, the Red Sea, Gulf of Aden, and Arabian Gulf, around Australia and South East Asia, and in the western and central Pacific Ocean (IUCN, 2015).This species is recorded from Gulf of Kachchh (Pillai & Patel, 1988), Netrani Island (Zacharia *et al.,* 2008), Gulf of Mannar (Pillai, 1986), Lakshadweep Islands (Pillai, 1971), Andaman and Nicobar Islands (Mondal *et al.*, 2015).

Remarks: Mostly grown in the protected area from wave exposure and dominant in station 3. Colonies were relatively big, sometimes more than a meter across. Some colonies were bleached.

Family **DENDROPHYLLIIDAE** Gray, 1847

Genus **Turbinaria** Oken, 1815

Species ***Turbinaria mesenterina*** Lamarck, 1816

Synonymy: *Explanaria mesenterina* Lamarck, 1816 (original combination, basionym); *Gemmipora mesenterina* Lamarck, 1816; *Turbinaria speciosa* Bernard, 1896; *Turbinaria tubifera* Bernard, 1896; *Turbinaria venusta* Bernard, 1896.

Common name: Disc coral

Description:

Colony flat and composed of thin laminae, arranged in convoluted whorls; Ridges are not conspicuous; Corallites are with white margin, crowded, exerted, and with trabeculae and papillae are seen on corallum. The size of polyp here is about 1.25-2 mm. Generally, colonies are less than one meter across but may form the large foliaceous structure on fringing reefs. With laminar vertical growth form, without ridges and with trabeculae and papillae are seen. Corallite margin is seen with budding polyps. Colony is with unifacial laminae. Upright and irregularly contorted fronds are with small, tubular flowering corallites, about1.25-2 mm, slightly exsert. The above size of corallites is large. Colony is encrusting with fronds that are contorted. Fronds are fused when growing in subtidal habitats.

Colour: Colour is usually grey-green or greyish brown. Polyps are usually white.

Occurrence and distribution: This species has recorded in the Indo-West Pacific, this species is found in the Red Sea and the Gulf of Aden, the southwest and the northwest Indian Ocean, the Arabian or Iranian Gulf, central Indian Ocean, the central Indo-Pacific, Australia, southern Japan, and the South China Sea, the oceanic West Pacific, and the Central Pacific (IUCN, 2015). This species is reported from all the major reefs of India, i.e., Gulf of Kachchh, Palk Bay (Rajasuria, 2007), Gulf of Mannar (Pillai, 1986) reported as *Turbinaria undata*, Lakshadweep

Islands (Suresh, 1991), Andaman and Nicobar Islands (Mondal *et al.,* 2015). This is the first report of occurrence from the MMS.

Remarks: This is the most dominant species in sites 1 and 2. Most of the colonies are larger and formed folios to plate structure. Though it can grow in the shallow turbid reef, sediment accumulation was observed and followed by turf algal growth on some colonies.

Genus **Turbinaria** Oken, 1815

Species ***Turbinaria frondens*** Dana, 1946

Synonymy: *Gemmipora frondens* Dana, 1846 (original combination, basionym), *Turbinaria abnormalis* Bernard, 1896; *Turbinaria aurantiaca* Bernard, 1896; *Turbinaria contorta* Bernard, 1896; *Turbinaria danae* Bernard, 1896; *Turbinaria edwardsi* Bernard, 1896; *Turbinaria foliosa* Bernard, 1896; *Turbinaria frondescens* Milne Edwards, 1860; *Turbinaria magna* Bernard, 1896; *Turbinaria pustulosa* Bernard, 1896; *Turbinaria ramosa* Yabe & Sugiyama, 1941; *Turbinaria rugosa* Bernard, 1896

Common name: Cup coral

Description:

Small cup-shaped colonies. They can develop to broad, unifacial, upright to horizontal fronds. Corallites can be long and tubular and usually 3.5 mm in diameter.

Colour: Colour is usually greenish-brown.

Occurrence and distribution: This is a widely distributed species, in the Indo-West Pacific, this species is reported in the Red Sea and the Gulf of Aden, the southwest and the central Indian Ocean, the central Indo-Pacific, Australia, southern Japan, and the South China Sea, the oceanic West Pacific, and the central Pacific (IUCN, 2015). This species is recorded from the Gulf of Kachchh (CMFRI, 2015), Lakshadweep Islands (Pillai, 1982) in Indian water.

Remarks: This species was present in the rocky reef of sites 4 and 5. There it has formed more than a meter across big foliaceous colonies. Some colonies show irregular growth formation.

Family **PORITIDAE** Gray, 1842

Genus Porites Link, 1807

Species ***Porites compressa*** Dana, 1846

Synonymy: None

Common name: Branching pore coral

Description:

Colony arising on a solid base with uneven surface and grazing at places; Polyps are polygonal. Branches are cylindrical and are commonly fused. Corallites are 0.8 to 1.4 mm in diameter. Colonies may form large patches of reef.

Colour: Colour is mostly dull grey, and brown.

Occurrence and distribution: Distributed in Pacific–eastern central; Pacific–North West and recorded as the dominant species in the Hawaiian Islands (IUCN, 2015). This species was reported from the Gulf of Kachchh (Pillai and Patel 1988), Palk Bay (Pillai, 1969), and Gulf of Mannar (Pillai, 1986).

Remarks: Generally present in shallower water. Colonies are massive, showed continuous growth up to 2m. Colony with nodular growth and top portion of some nodes are grazed down by fishes; often, it is covered with sediment. It is also infested with burrowing crypto fauna. Bleached colonies were found around sites 1, 2, and 3.

Genus **Goniopora** de Blainville, 1830

Species ***Goniopora stokesi*** Milne Edwards and Haime, 1851

Synonymy: Alveopora irregularis Crossland, 1952

Common name: Anemone coral

Description:

Colony is hemispherical in growth with fleshy tentacular growth; Corallum with large polyps (>5 mm) having long and fleshy tentacles. Colonies can be free-living or attached. Calices with high walls which have a ragged appearance. Small daughter colonies are found embedded in the living tissue of parent colonies.

Colour: Colonies are uniformly brown or greenish.

Occurrence and distribution: Occurrence of *Goniopora stokesi* is recorded in the Red Sea and the Gulf of Aden, southwest Indian Ocean, northwest Indian Ocean, northern Indian Ocean, central Indo-Pacific, north and west Australia, South-east Asia, Japan, and the South China Sea, eastern Australia, oceanic West Pacific (IUCN, 2015). This species was reported from the Palk Bay (Pillai, 1969), Gulf of Mannar (Pillai,1986), Lakshadweep Islands (Pillai, 1971), Andaman and Nicobar Islands in India (Mondal *et al.,* 2014).

Remarks: Solitary colony on rocky substratum was found at the site 4.

Genus **Goniopora** de Blainville, 1830

Species ***Goniopora pedunculata*** Quoy and Gaimard, 1833

Synonymy: *Goniopora minor* Crossland, 1952

Common name: Anemone coral

Description:

Colony is hemispherical with flowering polyps; Polyps have a white oral disc and tapering tentacles having pale white tips. Corallum encrusting and submassive; Corallite wall indistinct; Axial fossa circular. Septal structures are densely granulated. There are usually six thick palli forming a crown. Distinctively coloured oral disc and tentacles with pale tips.

Colour: Colour is brown or green.

Occurrence and distribution: Occurrence of *Goniopora pedunculata* reported in the Red Sea and the Gulf of Aden, southwest Indian Ocean, northern Indian Ocean, central Indo-Pacific, north and west Australia, eastern Australia, South-east Asia, Japan and the East China Sea, oceanic West Pacific, and the central Pacific (IUCN, 2015). This species is reported as *Goniopora minor* from Gulf of Kachchh (Pillai & Patel, 1988), Lakshadweep (Pillai, 1971 as

Goniopora minor), Gulf of Mannar (Patterson *et al.,* 2008), Andaman and Nicobar Islands (Mondal *et al.,* 2014).

Remarks: Only present in deeper regions (8-9m depth) of the reef at station 4. Polyps were extended during the daylight. In some colonies, the upper part of corallum was found to be bleached. Colony size varies between 40-60cm.

Genus **Bernardpora** Kitano and Fukami, 2014

Species ***Bernardpora stutchburyi*** Wells, 1955

Synonymy: *Goniopora stutchburyi* Wells, 1955 (original combination, basionym); *Goniopora nigra* Pillai, 1967; *Goniopora wotouensis* Zou and Song, 1975

Common name: Anemone coral

Description:

Colonies are sub-massive to encrusting, are smooth with small and shallow calices. Small jointed lobed development with flowering polyps on some lobes. Covered with some film with trapping of flowering polyps. The corallites are <3 mm in diameter and shallow. Polyps have short and tapering tentacles, which may not be extended during the day.

Colour: Colonies are pale brown or cream-coloured.

Occurrence and distribution: This species is recorded in the northern Indian Ocean (Sri Lanka and Gulf of Mannar), central Indo-Pacific, north and west Australia, South-east Asia, Japan, and the South China Sea, eastern Australia, oceanic West Pacific, and Central Pacific to French Polynesia (IUCN, 2015).This species is reported from the Gulf of Kachchh (Ramamoorthy, 2012 as *Goniopora nigra*, CMFRI, 2015 as *Goniopora stutchburyi*), Palk Bay (Pillai, 1969 as *Goniopora nigra*).

Remarks: Though this species generally found in a shallow reef environment, at the MMS, it was found in a deeper region (8-9m) at site 4. Some polyps were damaged by fish bite and showed partial degradation.

Table 4.2. Details of recorded scleractinian species in MMS reefs

Family	Species	Observed site	Relative abundance (Veron, 2000)	IUCN red list status
Poritidae	*Porites lichen*	T1, T2, T3, T5	Common	LC
Poritidae	*Porites lutea*	All sites	Common	LC
Poritidae	*Porites compressa*	T1, T2, T3	Common	LC
Poritidae	*Goniopora stokesi*	T4	Uncommon	NT
Poritidae	*Goniopora pedunculata*	T4	Common	NT
Poritidae	*Bernardpora stutchburyi*	T4	Uncommon	LC
Merulinidae	*Favites melicerum*	T1, T2, T4, T5	Rare	NT
Merulinidae	*Favites halicora*	T1, T2, T4	Uncommon	NT
Merulinidae	*Goniastrea retiformis*	T2, T3	Common	LC
Merulinidae	*Cyphastrea serailia*	All sites	Common	LC
Dendrophylliidae	*Turbinaria mesenterina*	All sites	Common	VU
Dendrophylliidae	*Turbinaria frondens*	T4, T5	Common	LC
Dendrophylliidae	*Tubastrea coccinea*	T2, T4, T5	Common	Not known
Siderastreidae	*Siderastrea savignyana*	T2, T3	Rare	LC
Siderastreidae	*Pseudosiderastrea tayami*	Intertidal rockpools, and T2, T3	Uncommon	NT
Plesiastreidae	*Plesiastrea versipora*	T3, T4, T5	Common	LC
Coscinaraeidae	*Coscinaraea monile*	T2, T3, T4	Common	LC
Scleractinia incertae sedis	*Leptastrea* sp.	T2	Common	-
Agariciidae	*Pavona* sp.	T4	Uncommon	-

LC: Least Concern; NT: Near Threatened; VU: Vulnerable

4.4. Discussion

The overall generic diversity of scleractinian corals in the MMS found to be relatively low compared to the other major Indian coral reefs. Branching corals like *Acropora* and *Pocillopora* are absent from this region. This might be because of high wave surge and high turbidity due to sedimentation and monsoonal dilution of seawater (Nair & Qasim, 1977), limiting the recruitment and survival of stress-sensitive branching species. Qasim & Wafar (1979), in the very first report on the occurrence of live coral in the MMS, reported that overall colonies size ranging from 15-20 cm across and 5cm height and sparse distribution of 1-2 colonies per square meter. However, during the present observation, higher abundance, extended coral cover, and larger colonies were found. Most of the colonies were medium to large in size and can be assumed that adult and old life stage (Gilbert, 2015). Therefore, the coral colonies indicate an increase in growth after the studies of Qasim and Wafar (1979) in the late seventies. However, due to lack of baseline quantitative information on coral and environmental assemblage of the MMS is limiting the ability to discuss on ages, growth of the reef, and to compare changes in live coral coverage along with other benthic life forms over a time scale. This study found that several massive *Porites, Plesiastrea, Goniastrea,* and *Turbinaria* are larger than ≥1m in diameter, suggesting the stress resilience and adaptation of these reef-forming corals species in the MMS. Encrusting corals like *Favites* and *Plesiastrea* showed continuous growth formation, which are encrusted on the hard-bottom rocky substratum and dead coral framework. Colonies of *Turbinaria mesenterina* are well developed and often formed prominent foliaceous structure.

The present study did not find previously reported *Cynaria lacrymalis, Turbinaria crater by* Parulekar, (1981), and *Synerea* sp. (NIO, 2000). Notably, *P. tayami* and *P. lichen* were encountered as dominant in the intertidal to shallow zone, whereas foliaceous *T. mesenterina* was conspicuous in the mid-region. *G. stutchburyi* and *P. versipora* were dominant in the deeper water. In nutshell, the present observations suggest that the reef formation in the MMS is dominated by the coral species known as stress-tolerant species foliaceous coral *Turbinaria, Porites, Goniastrea. Porites* spp. are the most dominant in overall distribution and abundance in the MMS reef environment. However, low numbers of small or juvenile corals (≤50 mm diameter) indicate a low recruitment rate, which might be linked with the rapid deterioration of reef environmental conditions by intensive human activities and recurrent thermal stress events, limiting recruitment and survivability of juvenile corals.

An account of biodiversity is necessary for the management and proper function of MPA in a region. In the present study, thirteen coral species are reported for the first time from this region, of which *Siderastrea savignyana* and *Favites melicerum* are considered to be rare globally (Veron, 2000). *Pseudosiderastrea tayami, Favites halicora, Goniopora stokesi, Goniopora stutchburyi are* uncommon species (Veron, 2000). According to the IUCN red list of threatened species, five species are under ‘near threatened,’ one is ‘vulnerable.’ Moreover, De *et al.,* (2015) recently reported coral bleaching from the MMS. Hence, the occurrence of such rare and threatened species and coral bleaching evidence demand urgent need for conservation of coral population in this region and protection from further deterioration. It has been estimated that 30% of global coral reefs are severely damaged, and predicted close to 60% may be lost by 2030 (Jackson *et al.*, 2001). Generally, damage and loss of coral reefs happen due to an increase in Sea Surface Temperature (SST) related to climate change and anthropogenic activities (Hughes *et al.,* 2003). During the present study, we also observed that the combined effect of reef tourism and coastal development along with climate change imposing adverse threat on reef corals in the form of coral bleaching (De *et al.,* 2015) and disease, leading towards reef degradation in the MMS.

Table 4.3. Comparison of coral species reported by earlier studies and the present study

Quasim & wafer (1979)	Parulekar (1981)	NIO report (2000)	Bhosale (2003)	Present study
Porites lichen	*Porites lichen*	*Porites lichen*	*Porites lichen*	*Porites Lichen*
Porites lutea	*Porites lutea*	*Porites lutea*	*Porites lutea*	*Porites lutea*
X	X	X	X	*Porites compressa*
Coscinaraea sp.	*Coscinaraea monile*	*Coscinaraea* sp.	*Coscinaraea* sp.	*Coscinaraea monile*
Pseodosiderastrea sp.	X	*Pseodosiderastrea* sp.	*Pseudosiderastrea tayami*	*Pseudosiderastrea tayami*
X	*Siderastrea savignyana*	X	X	*Siderastrea savignyana*
Cyphastrea sp.	*Cyphastrea* sp.	*Cyphastrea* sp.	*Cyphastrea* sp.	*Cyphastrea serailia*
Turbinaria sp.	*Turbinaria crater*	*Turbinaria* sp.	X	*Turbinaria mesenterina*
X	X	X	X	*Turbinaria frondens*
Synarea sp.	*Cynaria lacrymalis*	*Synerea* sp.	*Porites (Synarea) sp.*	*X*
X	X	*Goniopora* sp.	X	*Goniopora stokesi*
X	X	X	X	*Goniopora pedunculata*
X	X	X	X	*Bernardpora stutchburyi*
Goniastrea sp.	*Goniostrea retiformis*	*Goniostrea* sp.	X	*Goniostrea retiformis.*
X	X	X	X	*Leptastrea* sp.
X	X	X	X	*Pavona* sp.
Favites sp.	X	*Favites* sp.	X	*Favites melicerum*
X	*Favites halicora*	X	X	*Favites halicora*
X	X	X	X	*Plesiastrea versipora*
X	X	*Tubastrea* sp.	X	*Tubastrea coccinea*

4.5. Conclusion

Being a nearshore coral reef, the MMS is relatively more accessible for recreational activates than the other coral reefs of India. Hence tourism activity related to the coral reef is booming in MMS. Also, it appears that there is a significant urban development in progress to meet the increasing demand of increasing tourism. The occurrence of the stress-tolerant coral species in the marginal coral habitats have high recovery potential due to localized adaptations to multiple chronic stresses and thus can serve as a vital gene bank in the future. Therefore, the reef habitat in MMS demands urgent ecological interventions and commencement of continual monitoring of the effects of local stressors on the reef community structure of the reefs and its health, which will aid in developing an action plan for proper conservation and sustainability of this fragile ecosystem.

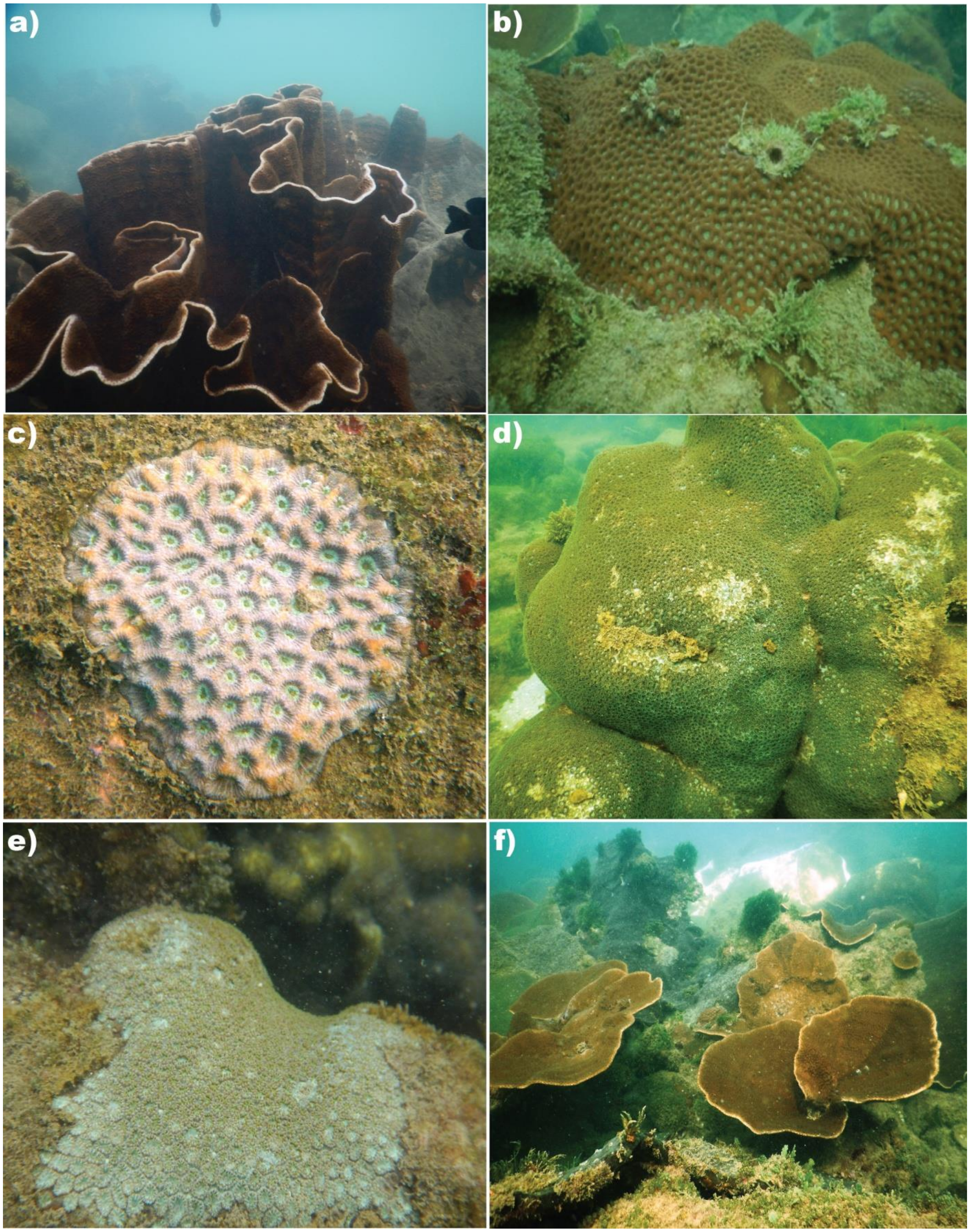

Fig. 4.2: Some of the coral species recorded during the present study a. Turbinaria mesenterina; b. Siderastrea savignyana; c. Favites melicerum; d. Cyphastrea serailia; e. Plesiastrea versipora; f. Turbinaria mesenterina

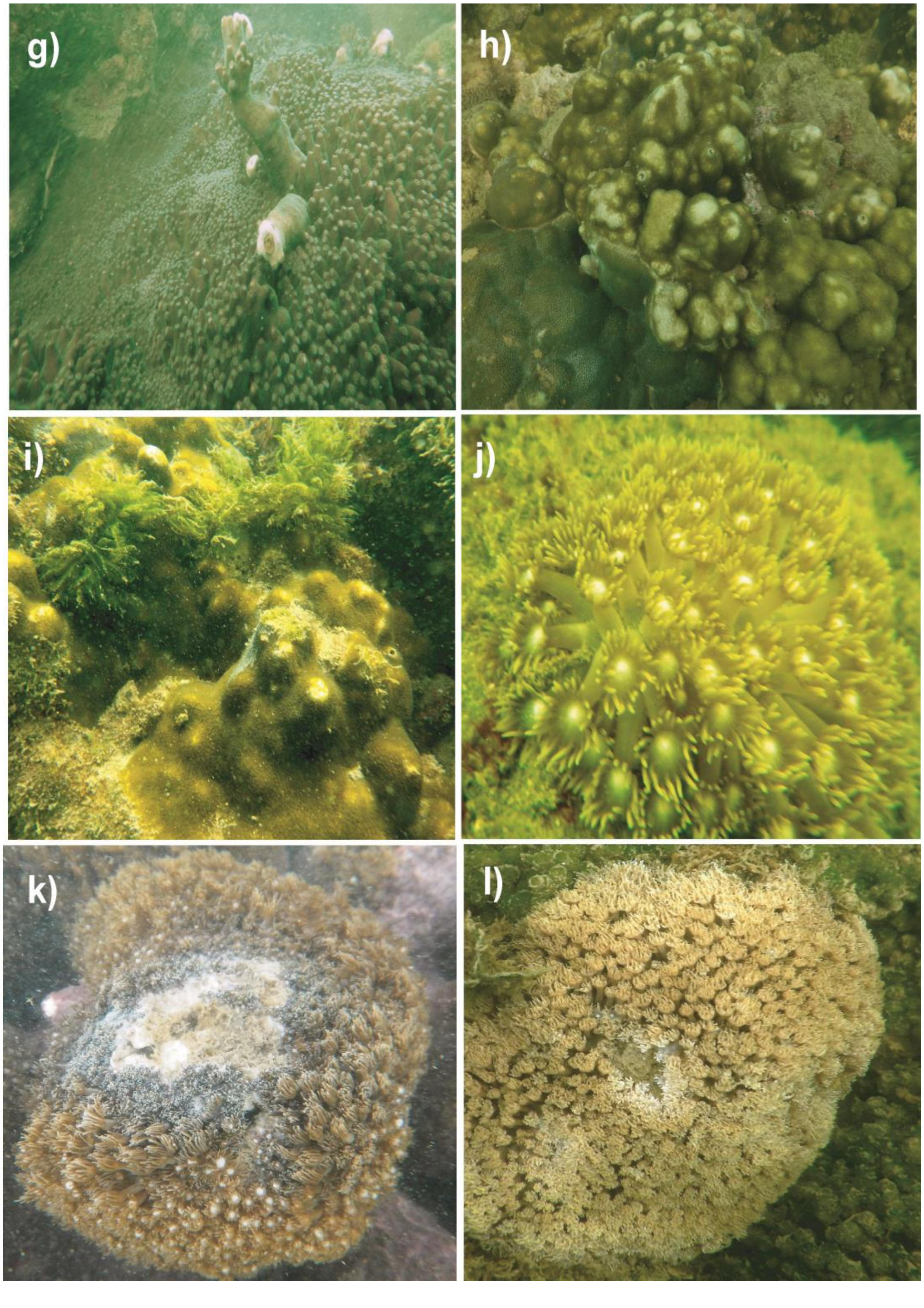

Fig 4.2: g. ***Turbinaria frondens;*** **h.** ***Porites compressa*****; i.** ***Porites*** **sp.; j.** ***Goniopora stokesi;*** **k.** ***Goniopora pedunculata*****; l.** ***Bernardpora stutchburyi***

Table 4.4. List of biotas documented in different literature from the Malvan marine sanctuary

Phylum	Class	Subclass	Order	Family	Species	Reference
Cnidaria	Anthozoa	Hexacorallia	Actiniaria	Edwardsiidae	*Edwardsia tinctrix*	Parulekar, 1981
				Haloclavidae	*Metapeachia tropica*	Parulekar, 1981
				Haliactinidae	*Pelocoetes exul*	Parulekar, 1981
					Pelocoetes gangeticus	Parulekar, 1981
				Actiniidae	*Anemonia indicus/ Anemonia indica*	Parulekar, 1981
					Bunodosoma granulifera	Parulekar, 1981
					Anthopleura midori/ Anthopleura anjunae	Parulekar, 1981
					Anthopleura asiatica	Parulekar, 1981
					Anthopleura panikkarii	Parulekar, 1981
					Anthopleura pacifica	UNDP, 2011
					Paracondylactis indicus/Paracondylactis sinensis	Parulekar, 1981
					Actinogeton sultana/ Gyractis sesere	Parulekar, 1981
					Cribrinopsis robertii	Parulekar, 1981
				Acontiophoridae	*Aconitiophorum bombayensi/Acontiophorum bombayense*	Parulekar, 1981
				Metridiidae	*Metridium senile* var. fimbriatum/*Metridium senile*	Parulekar, 1981
					Neoaiptasia commensali	Parulekar, 1981
				Aiptasiomorphidae	*Aiptasiomorpha luciae*	Parulekar, 1981
				Diadumenidae	*Didumene schilleriana*	Parulekar, 1981
			Zoantharia	Epizoanthidae	*Epizoanthus elongatum*	Parulekar, 1981
		Octocorallia	Pennatulacea	Veretillidae	*Cavernularia orientails/ Cavernulina orientails*	Parulekar, 1981
				Virgulariidae	*Virgularia rumphii*	Parulekar, 1981
		Hexacorallia	Scleractinia	Siderastreidae	*Siderastrea savignvana*	Parulekar, 1981, De *et al.,* 2015
					Pseudosiderastrea tayami	De *et al.,* 2015
					Pseudosiderastrea sp.	ICMAM, 2001
				Coscinaraeidae	*Coscinarea monile*	Parulekar, 1981, De *et al.,* 2015
				Poritidae	*Porites lutea*	Parulekar, 1981, De *et al.,* 2015
					Porites lichen	Parulekar, 1981, De *et al.,* 2015

				Goniopora minor	De *et al.*, 2015, Tripathy *et al.*, 2016
				Goniopora sp.	ICMAM, 2001, De *et al.*, 2015
			Merulinidae	*Favites halicora*	Parulekar 1981, De *et al.*, 2015
				Favites sp.	ICMAM, 2001
				Favites bestae	De *et al.*, 2015
				Goniastrea retiformes	Parulekar, 1981, De *et al.*, 2015
				Cyphastrea serailia	De *et al.*, 2015
				Cyphastrea sp.	Parulekar, 1981
			Scleractinia incertae sedis	*Leptastrea purpurea*	Tripathy *et al.*, 2016
			Plesiastreidae	*Plesiastrea versipora*	De *et al.*, 2015
			Agariciidae	*Pavona bipartita*	Tripathy *et al.*, 2016
			Lobophylliidae	*Cynaria lacrymalis*	Parulekar, 1981
			Dendrophylliidae	*Turbinaria crater*	Parulekar, 1981
				Turbinaria sp.	ICMAM, 2001
				Turbinaria mesenterina	De *et al.*, 2015
				Astraea stellata/Turbinaria stellulata	UNDP, 2011
				Tubastraea coccinea	De *et al.*, 2015
				Tubastrea sp.	ICMAM, 2001
			Caryophylliidae	*Polycyathus verrilli*	Tripathy *et al.*, 2016
Chordata	Elasmobranchii	Orectolobiformes	Hemiscylliidae	*Chiloscyllium griseum*	Barman *et al.*, 2007
			Rhincodontidae	*Rhincodon typus*	Mainkar, 1992
		Carcharhiniformes	Carcharhinidae	*Scoliodon laticaudus*	Barman *et al.*, 2007, Tike *et al.*, 2009
		Pristiformes	Pristidae	*Anoxypristis cuspidata*	Tike *et al.*, 2009
		Myliobatiformes	Myliobatidae	*Aetomylaeus maculatus*	Tike *et al.*, 2009
	Actinopterygii	Anguilliformes	Ophichthidae	*Neenchelys buitendijki*	Barman *et al.*, 2007
				Ophichthus apicalis	Barman *et al.*, 2007
				Pisodonophis boro	Parulekar, 1981
			Muraenesocidae	*Congresox talabonoides*	Barman *et al.*, 2007
				Muraenesox cinereus	Barman *et al.*, 2007
		Clupeiformes	Clupeidae	*Amblygaster clupeoides*	Barman *et al.*, 2007
				Hilsa kelee	Barman *et al.*, 2007
				Nematolosa nasus	Barman *et al.*, 2007

		Pellona ditchella	Barman *et al.*, 2007
		Sardinella fimbriata	Tike *et al.*, 2009
		Sardinella melaneura	Barman *et al.*, 2007
		Sardinella longiceps	Barman *et al.*, 2007
		Tenualosa ilisha	Barman *et al.*, 2007, Tike *et al.*, 2009
	Pristigasteridae	*Opisthopterus tardoore*	Barman *et al.*, 2007
		llisha melastoma	Barman *et al.*, 2007
	Engraulididae	*Stolephorus indicus*	Barman *et al.*, 2007, Tike *et al.*, 2009
		Stolephorus commersonnii	Tike *et al.*, 2009
		Thryssa hamilton	Barman *et al.*, 2007
		Thryssa dussumieri	Barman *et al.*, 2007
		Thryssa malabarica	Barman *et al.*, 2007
		Thryssa mystax	Barman *et al.*, 2007, Tike *et al.*, 2009
		Thryssa setirostris	Barman *et al.*, 2007
		Coilia dussumieri	Tike *et al.*, 2009
	Chirocentridae	*Chirocentrus dorab*	Barman *et al.*, 2007, Tike *et al.*, 2009
Siluriformes	Bagridae	*Mystus gulio*	Barman *et al.*, 2007
	Ariidae	*Arius maculatus*	Barman *et al.*, 2007
		Arius platystomus	Barman *et al.*, 2007
		Arius thalassinus	Barman *et al.*, 2007
		Osteogeneiosus militaris	Tike *et al.*, 2009
	Plotosidae	*Plotosus canius*	Barman *et al.*, 2007
Aulopiformes	Harpadontidae	*Harpadon nehereus*	Barman *et al.*, 2007
	Synodidae	*Saurida tumbil*	Barman *et al.*, 2007
Gadiformes	Bregmacerotidae	*Bregmaceros mcclellandii*	Barman *et al.*, 2007
Mugiliformes	Mugilidae	*Liza tade*	Barman *et al.*, 2007
		Mugil cephalus	Barman *et al.*, 2007, Tike *et al.*, 2009
Beloniformes	Hemiramphidae	*Hemiramphus archipelagicus*	Barman *et al.*, 2007
		Hemiramphus far	Barman *et al.*, 2007

			Hemiramphus lutkei	Barman *et al.*, 2007
	Scorpaeniformes	Platycephalidae	*Platycephalus indicus*	Barman *et al.*, 2007, Tike *et al.*, 2009
	Perciformes	Centropomidae	*Lates calcarifer*	Barman *et al.*, 2007
		Serranidae	*Epinephelus bleekeri*	Barman *et al.*, 2007
			Epinephelus dicanthus	Tike *et al.*, 2009
		Teraponidae	*Terapon jarbua*	Parulekar, 1981, Barman *et al.*, 2007
			Terapon theraps	Barman *et al.*, 2007
		Apogonidae	*Apogon thermalis*	Barman *et al.*, 2007
		Sillaginidae	*Sillago sihama*	Barman *et al.*, 2007, Tike *et al.*, 2009
		Carangidae	*Alepes ciliaris*	Barman *et al.*, 2007
			Alepes djedaba	Barman *et al.*, 2007
			Alepeskleinii	Barman *et al.*, 2007
			Atropus atropos	Barman *et al.*, 2007
			Caranx carangus	Barman *et al.*, 2007
			Caranx ignobilis	Barman *et al.*, 2007
			Caranx sexfasciatus	Barman *et al.*, 2007
			Decapterus russelli	Barman *et al.*, 2007
			Gnathonodon speciosus	Barman *et al.*, 2007
			Megalaspis cordyla	Barman *et al.*, 2007
			Naucrates ductor	Barman *et al.*, 2007
			Parastromus niger	Barman *et al.*, 2007
			Scomberoides lysan	Barman *et al.*, 2007
			Scomberoides tol	Barman *et al.*, 2007
			Trachinotus blochii	Barman *et al.*, 2007
			Carangoides sp.	Tike *et al.*, 2009
		Coryphaenidae	*Coryphaena hippurus*	Barman *et al.*, 2007
		Leiognathidae	*Gazza minuta*	Barman *et al.*, 2007, Tike *et al.*, 2009
			Leiognathus bindus	Barman *et al.*, 2007
			Leiognathus brevirostris	Barman *et al.*, 2007

	Leiognathus daura	Barman *et al.*, 2007
	Leiognathus equulus	Barman *et al.*, 2007
	Secutor insidiator	Barman *et al.*, 2007
	Lutjanus argentimaculatus	Barman *et al.*, 2007
	Lutjanus johnii	Barman *et al.*, 2007, Tike *et al.*, 2009
	Lutjanus russellii	Barman *et al.*, 2007
Gerreidae	*Gerres erythrourus*	Barman *et al.*, 2007
	Gerres filamentosus	Tike *et al.*, 2009
	Gerres limbatus	Barman *et al.*, 2007
	Gerres macracanthus	Barman *et al.*, 2007
Haemulidae	*Pomadasys kaakan*	Barman *et al.*, 2007
	Pomadasys maculatus	Barman *et al.*, 2007
Nemipteridae	*Nemipterus japonicus*	Barman *et al.*, 2007
Sciaenidae	*Chrysochir aureus*	Barman *et al.*, 2007
	Jahnius carutta	Barman *et al.*, 2007
	Otolithes rubber	Barman *et al.*, 2007
	Johnius dussumieri	Tike *et al.*, 2009
	Protonibea dicanthus	Tike *et al.*, 2009
Mullidae	*Upeneus vittatus*	Barman *et al.*, 2007
Monodactylidae	*Monodactylus argenteus*	Barman *et al.*, 2007
Pempherididae	*Pempheris vanicolensis*	Barman *et al.*, 2007
Scatophagidae	*Scatophagus argus*	Barman *et al.*, 2007
Cichlidae	*Etrophus suratensis*	Barman *et al.*, 2007, Tike *et al.*, 2009
Sphyraenidae	*Sphyraena jella*	Barman *et al.*, 2007
Polynymidae	*Eleutheronema tatradactylum*	Barman *et al.*, 2007
	Filimanus xanthonema	Barman *et al.*, 2007
	Polydactylus mullani	Barman *et al.*, 2007
Gobiidae	*Caragobius urolepis*	Barman *et al.*, 2007
	Odontamblyopus rubicundus	Barman *et al.*, 2007
	Callogobius sp.	Parulekar, 1981
	Batrachus sp.	

				Trypauchenidae	*Trypauchen vagina*	Barman *et al.*, 2007
				Siganidae	*Siganus canaliculatus*	Barman *et al.*, 2007
					Siganus vermiculatus	Tike *et al.*, 2009
				Trichiuridae	*Lepturacanthus pantului*	Barman *et al.*, 2007
					Trichiurus lepturus	Tike *et al.*, 2009
				Scombridae	*Auxis thazard*	Barman *et al.*, 2007
					Rastrelliger kanagurta	Barman *et al.*, 2007
					Scomberomorus commerson	Barman *et al.*, 2007
					Scomberomorus guttatus	Barman *et al.*, 2007
					Scomberoides tol	Barman *et al.*, 2007
					Thunnus albacares	Barman *et al.*, 2007
					Thunnus tonggol	Barman *et al.*, 2007
				Stromateidae	*Pampus argenteus*	Barman *et al.*, 2007
					Pampus chinensis	Barman *et al.*, 2007
				Menidae	*Mene maculata*	Tike *et al.*, 2009
				Sparidae	*Rhabdosargus sarba*	Tike *et al.*, 2009
					Acanthopagrus berda	Tike *et al.*, 2009
				Lactariidae	*Lactarius lactarius*	Tike *et al.*, 2009
				Blenniidae	*Petroscirtes punctatu, accepted name Omobranchus punctatuss*	Parulekar, 1981
			Pleuronectiformes	Paralichthyidae	*Pseudorhombus javanicus*	Barman *et al.*, 2007
					Pseudorhombus triocellatus	Barman *et al.*, 2007
				Cynoglossidae	*Cynoglossus puncticeps*	Barman *et al.*, 2007
					Cynoglossus lingua	Tike *et al.*, 2009
				Soleidae	*Solea ovata*	Barman *et al.*, 2007
					Synaptura albomaculata	Barman *et al.*, 2007
			Tetraodontiformes	Tetraodontidae	*Lagocephalus inermis*	Barman *et al.*, 2007
					Torquigener hypselogenion	Barman *et al.*, 2007
				Diodontidae	*Diodon hystrix*	Barman *et al.*, 2007
					Diodon holocanthus	Barman *et al.*, 2007
Mammal	Chordata	Mammalia	Cetartiodactyla	Delphinidae	*Delphinus delphis*	UNDP, 2011, Jog *et al.*, 2015
				Phocoenidae	*Neophocaena phocaenoides*	UNDP, 2011, Jog *et al.*, 2015

				Balaenopteridae	*Balaenoptera edeni*	Jog *et al.*, 2015
					Balaenoptera musculus	Jog *et al.*, 2015
Turtle		Reptilia	Testudines	Cheloniidae	*Chelonia mydas*	UNDP, 2011
					Lepidochelys olivacea	UNDP, 2011
			Squamata	Elapidae	*Hydrophis schistosus*	Rao & Muralidharan, 2017
					Hydrophis curtus	Rao & Muralidharan, 2017
				Acrochordidae	*Acrochordus granulatus*	Rao & Muralidharan, 2017
Arthropoda	Crustacea	Malacostraca	Stomatopoda	Squillidae	*Squilla raphidea*	Parulekar, 1981
					Squilla nepa	Parulekar, 1981
					Squilla scorpio	Parulekar, 1981
					Squilla interrupta	Parulekar, 1981
				Gonodactylidae	*Gondactylus chircra*	Parulekar, 1981
			Isopoda	Cirolanidae	*Cirolana* sp.	Parulekar, 1981
				Limnoriidae	*Limnoria (Limnoria) bombayensis*	Parulekar, 1981
				Sphaeromatidae	*Sphaeroma walkeri*	Parulekar, 1981
				Idoteidae	*Sybidotea variegate/ Synidotea variegata*	Parulekar, 1981
				Ligiidae	*Ligia exotica*	Parulekar, 1981
			Decapoda	Matutidae	*Matuta victor*	Parulekar, 1981
					Matuta planipes	Parulekar, 1981
				Dorippidae	*Dorippe astute*	Parulekar, 1981
				Hymenosomatidae	*Elamena cristatipes*	Parulekar, 1981
				Majidae	*Schizophrys aspera*	Parulekar, 1981
				Portunidae	*Portunus pelagicus*	Parulekar, 1981
					Portunus sanguinolentus	Parulekar, 1981
					Charybdis annulata	Parulekar, 1981
					Thalamita crenata	Parulekar, 1981
				Leucosiidae	*Philyra globosa/Philyra globus*	UNDP, 2011
				Xanthidae	*Leptodius arassimanus*	Parulekar, 1981
				Oziidae	*Ozius rugulosus*	Parulekar, 1981
				Pinnotheridae	*Pinnotheres*sp.	Parulekar, 1981
				Ocypodidae	*Uca annulipes/Austruca annulipes*	Parulekar, 1981

			Ocypodidae	*Ocypod ceratopthalma*	Parulekar, 1981
			Dotillidae	*Dotilla myctiroides*	Parulekar, 1981
			Macrophthalmidae	*Macropthalamus sulcatus/Macrophthalmus (Macrophthalmus) sulcatus*	Parulekar, 1981
			Hippidae	*Emerita holthuisi*	Parulekar, 1981
			Grapsidae	*Metagrapsus messor/Metopograpsus messor*	Parulekar, 1981
			Sesarmidae	*Sesarme oceanica?*	Parulekar, 1981
			Porcellanidae	*Petrolisthes boscii*	Parulekar, 1981
			Diogenidae	*Clibanarius intraspinatus*	Parulekar, 1981
				Clibanarius padavensis	Parulekar, 1981
			Diogenidae	*Diogenes custus/Diogenes custos*	Parulekar, 1981
				Diogenes miles	Parulekar, 1981
			Sergestidae	*Acetes indicus*	Parulekar, 1981
			Penaeidae	*Penaeus japonicus*	Parulekar, 1981
				Metapenaeus monoceros	Parulekar, 1981
				Parapeneopsis stylifera/Parapenaeopsis stylifera	Parulekar, 1981
			Lysmatidae	*Hippolysmata ensiostris/Exhippolysmata ensirostris*	Parulekar, 1981
	Hexanauplia	Ibliformes	Iblidae	*Ibla cumingi*	Parulekar, 1981
		Lepadiformes	Lepadidae	*Lepas*sp.	Parulekar, 1981
		Sessilia	Balanidae	*Balanus tintinnabulum/Megabalanus tintinnabulum*	Parulekar, 1981
				Balanus amphitrite/Amphibalanus amphitrite	Parulekar, 1981
			Tetraclitidae	*Tetraclita purpurascens*	Parulekar, 1981
			Chthamalidae	*Cthamalus withersi/Microeuraphia withersi*	Parulekar, 1981
	Insecta	Hemiptera	Gerridae	*Halobates* sp.	Parulekar, 1981
Mollusca	Polyplacophora	Chitonida	Ischnochitonidae	*Ischnochiton computus*	Parulekar, 1981, Tike *et al.*, 2009
	Gastropoda	Lepetellida	Fissurellidae	*Diodora bombayana*	Parulekar, 1981, Tike *et al.*, 2009
				Scutus unguis	Parulekar, 1981, Tike *et al.*, 2009
			Nacellidae	*Cellana radiata*	Parulekar, 1981, Tike *et al.*, 2009
			Haliotidae	*Haliotis varia*	UNDP, 2011

Seguenziida	Chilodontaidae	*Euchelus asper*	Parulekar, 1981, Tike *et al.*, 2009
		Euchelus tricarinatus/Euchelus tricarinata	Parulekar, 1981, Tike *et al.*, 2009
Trochida	Calliostomatidae	*Calliostoma scobinatum*	Parulekar, 1981, Tike *et al.*, 2009
		Gibbula swainsonnii?	Parulekar, 1981, Tike *et al.*, 2009
	Trochidae	*Clanculus depictus*	Parulekar, 1981, Tike *et al.*, 2009
		Clanculus ceylanicus	UNDP, 2011
		Trochus radiatus	Parulekar, 1981, Tike *et al.*, 2009
		Umbonium vestiarum/Umbonium vestiarium	Parulekar, 1981, Tike *et al.*, 2009
	Turbinidae	*Turbo intercostalis*	Parulekar, 1981, Tike *et al.*, 2009
		Turbo brunneus	UNDP, 2011, Tike *et al.*, 2009
		Turbo coronatus	Tike *et al.*, 2009
		Astraea semicostata/Astralium semicostatum	UNDP, 2011
		Isanda crenulifera/Microthyca crenellifera	UNDP, 2011
Cycloneritida	Neritidae	*Nerita oryzarum*	Parulekar, 1981, Tike *et al.*, 2009
		Nerita polita	Parulekar, 1981, Tike *et al.*, 2009
		Nerita crepidularia	UNDP, 2011
		Nerita albicella	UNDP, 2011
Ellobiida	Ellobiidae	*Cossidula nucleus*	UNDP, 2011
		Melampus coffea	UNDP, 2011
		Melampus sincaporensis	UNDP, 2011
		Eassidula nucleus	UNDP, 2011
		Ellobium aurisjudae	UNDP, 2011
Pylopulmonata	Pyramidellidae	*Pyramidella pulchella/Tiberia pulchella*	UNDP, 2011
Siphonariida	Siphonariidae	*Siphonaria basseineusis/Siphonaria basseinensis*	UNDP, 2011
Caenogastropoda	Planaxidae	*Planaxis sulcatus*	Parulekar, 1981, Tike *et al.*, 2009
		Planaxis acutus	UNDP, 2011
		Planaxis similes	UNDP, 2011
	Potamididae	*Cerithidea fluviatilis/Potamides cingulatus/Pirenella cingulata*	Parulekar, 1981, Tike *et al.*, 2009, UNDP, 2011
		Telescopium telescopium	Parulekar, 1981

		Alaba rectangulata?	Parulekar, 1981
	Cerithiidae	*Cerithium morus*	Parulekar, 1981, Tike *et al.*, 2009
		Cerithium rubus	Parulekar, 1981, Tike *et al.*, 2009
	Epitoniidae	*Janthina roseola/Janthina janthina*	UNDP, 2011
		Acrilla acuminate/Acrilla acuminata	UNDP, 2011
	Turritellidae	*Turritella terebra*	Tike *et al.*, 2009
		Turritella duplicata	Parulekar 1981, Tike *et al.*, 2009
Littorinimorpha	Littorinidae	*Littorina subgranosa/Echinolittorina leucosticta*	Parulekar 1981, Tike *et al.*, 2009
		Littorina intermedia	Parulekar 1981, Tike *et al.*, 2009
		Littorina ventricosa	Parulekar, 1981, Tike *et al.*, 2009
		Littorina undulate	UNDP, 2011
		Tectarius malaccanus?/Echinolittorina malaccana	Parulekar, 1981, Tike *et al.*, 2009
	Vermetidae	*Vermetus* sp.	Parulekar, 1981, UNDP, 2011, Tike *et al.*, 2009
	Calyptraeidae	*Ergoea walshii*	Parulekar, 1981, Tike *et al.*, 2009
	Rostellariidae	*Tibia Curta*	Parulekar, 1981, Tike *et al.*, 2009
	Naticidae	*Natica lineata/Tanea lineata*	Parulekar, 1981, Tike *et al.*, 2009
		Natica maculosa	Parulekar, 1981, Tike *et al.*, 2009
		Natica didyma/Neverita didyma	UNDP, 2011
		Natica picta/ Tanea picta	UNDP, 2011
		Natica pulcaria/Natica cincta	UNDP, 2011
		Natica ruf/Natica vitellus	UNDP, 2011
	Cypraeidae	*Cyprea pallida*	Parulekar, 1981
		Cypraea lentiginosa/Palmadusta lentiginosa	UNDP, 2011
		Cypraea arabica/Mauritia arabica	UNDP, 2011
		Cyprea sp.	Tike *et al.*, 2009
	Bursidae	*Bursa spinosa/Bufonaria echinata*	UNDP, 2011

		Bursa tuberculata/Gyrineum natator	Parulekar, 1981, Tike *et al.*, 2009
	Tonnidae	*Tonna allium*	UNDP, 2011
		Tonna fasciata/Tonna sulcosa	UNDP, 2011
		Tonna dolium	UNDP, 2011
	Xenophoridae	*Xenephora solaris/Stellaria solaris*	UNDP, 2011
	Ficidae	*Ficus ficus*	UNDP, 2011
	Ovulidae	*Volva sowerbyna/Phenacovolva rosea*	UNDP, 2011
Neogastropoda	Columbellidae	*Pyrene terpsichore*	Parulekar, 1981, Tike *et al.*, 2009
		Pyrene scripta/Mitrella scripta	UNDP, 2011
	Babyloniidae	*Babylonia spirata*	Parulekar, 1981, Tike *et al.*, 2009
	Aplustridae	*Bullia lineolata*	Parulekar, 1981, Tike *et al.*, 2009
	Nassariidae	*Nassarius ornatus/Nassarius stolatus*	Parulekar, 1981, Tike *et al.*, 2009
		Nassarius canaliculata/Nassarius siquijorensis	UNDP, 2011
		Nassarius jacksoniana	UNDP, 2011
		Nassarius lentiginosis/Nassarius reeveanus	UNDP, 2011
		Nassarius mucronatus/Nassarius gaudiosus	UNDP, 2011
		Nassarius olivaceous	UNDP, 2011
		Nassarius pictus/Nassarius sufflatus	UNDP, 2011
		Cyllene fuscata	UNDP, 2011
		Nassa thersitis/Nassarius pullus	UNDP, 2011
		Zeuxis caelatus/Nassarius caelatus	UNDP, 2011
	Nassariidae	*Nassarina suturalis/Nassarius glans*	UNDP, 2011
	Clavatulidae	*Surcula javana/Turricula javana*	Parulekar, 1981, Tike *et al.*, 2009
		Surcula amicta/Turris amicta	UNDP, 2011
		Surcula fulminata/Turricula tornata fulminata	UNDP, 2011

Terebridae	*Terebra capensis/Euterebra capensis*	UNDP, 2011
Mitridae	*Chrysame ambigua/Strigatella ambigua*	UNDP, 2011
	Mitra obeliscus/Vexillum obeliscus	UNDP, 2011
	Mitra circula/Domiporta circula	UNDP, 2011
Muricidae	*Murex adustus/Chicoreus brunneus*	Parulekar, 1981, Tike *et al.*, 2009
	Murex tribulus	UNDP, 2011
	Drupa konkanensis?	Parulekar, 1981, Tike *et al.*, 2009
	Drupa contracta/Ergalatax contracta	UNDP, 2011
	Drupa hippocastanum?	UNDP, 2011
	Thais caranifera/Indothais lacera	Parulekar, 1981, Tike *et al.*, 2009
	Thais rudolphi/Purpura persica	Parulekar, 1981, Tike *et al.*, 2009
	Thais tissoti/Semiricinula tissoti	Parulekar, 1981, Tike *et al.*, 2009
	Thais sacellum/Indothais sacellum	UNDP, 2011
	Thais bufo/Purpura bufo	UNDP, 2011
	Ocinebra bombayana/Lataxiena bombayana	UNDP, 2011
Cancellariidae	*Cancellaria costifera/Scalptia scalariformis*	UNDP, 2011
Olividae	*Oliba nebulosa/Agaronia gibbosa*	UNDP 2011
	Oliva gibbosa/Agaronia gibbosa	UNDP 2011
Conidae	Conus *monachus*	UNDP, 2011
	Conus *mutabilis/Conus hyaena*	UNDP, 2011
	Conus *piperatus*	UNDP, 2011
	Conus cumingii	UNDP, 2011
Architectonicidae	*Architectonica laevigate/Architectonica laevigata*	UNDP, 2011
Pisaniidae	*Cantharus spiralis*	UNDP, 2011

			Engina zea	UNDP, 2011
			Polia rubiginosa	UNDP, 2011
		Melongenidae	*Hemifusus pugilinus/Volegalea cochlidium*	UNDP, 2011
		Drilliidae	*Drilla atkinsoni/Etrema denseplicata*	UNDP, 2011
			Clavus crassa/Clavus aglaia	UNDP 2011
	Aplysiida	Aplysiidae	*Aplysia cornifera/Aplysia cornigera*	Parulekar, 1981, Tike *et al.*, 2009
			Bursatella leachii	Parulekar, 1981, Tike *et al.*, 2009
	Systellommatophora	Onchidiidae	*Onchidium verraculatum/Peronia verruculata*	Parulekar, 1981, Tike *et al.*, 2009
Bivalvia	Arcida	Arcidae	*Arca bistrigata/Mesocibota bistrigata*	Parulekar, 1981, Tike *et al.*, 2009
			Arca symmetrica	Parulekar, 1981, Tike *et al.*, 2009
			Anadara granosa/Tegillarca granosa	Parulekar, 1981, Tike *et al.*, 2009
	Mytilida	Mytilidae	*Mystilus viridis/Perna viridis*	Parulekar, 1981
			Modiolus striatulus/Brachidontes striatulus	Parulekar, 1981, Tike *et al.*, 2009
			Brachyodontes karachiensis/Brachidontes pharaonis	Parulekar, 1981, Tike *et al.*, 2009
	Pectinida	Pectinidae	*Chlamys tranquebaricus/Volachlamys tranquebaria*	Parulekar, 1981, Tike *et al.*, 2009
		Anomiidae	*Anomis achaeus*	Parulekar, 1981, Tike *et al.*, 2009
	Limida	Limidae	*Lima lima*	Parulekar, 1981, Tike *et al.*, 2009
	Ostreida	Ostreidae	*Astrea stellata/Crassostrea cucullata/ Saccostrea cuccullata*	Parulekar, 1981, Tike *et al.*, 2009
			Crassostrea bicolor/Crassostrea tulipa	Parulekar, 1981, Tike *et al.*, 2009
			Crassostrea lacerata/Ostrea lacerata	Parulekar, 1981, Tike *et al.*, 2009
			Crassostrea gryphoides/Magallana gryphoides	Tike *et al.*, 2009
		Margaritidae	*Pinctada chemnitzi*	Parulekar, 1981, Tike *et al.*, 2009
	Venerida	Trapezidae	*Trapezium vellicatum?*	Parulekar, 1981

				Veneridae	*Gafrarium divericatum*	Parulekar, 1981/Tike *et al.*, 2009
					Sunetta solandri	Parulekar, 1981
					Sunetta effossa	Tike *et al.*, 2009
					Meretrix meretrix	Parulekar, 1981, Tike *et al.*, 2009
					Meretrix casta	Parulekar, 1981, Tike *et al.*, 2009
					Paphia textile/Paratapes textilis	Parulekar, 1981, Tike *et al.*, 2009
					Paphia malabarica/Protapes gallus	Tike *et al.*, 2009
					Paphia alapapilioni/Paphia rotundata	Tike *et al.*, 2009
					Catelysia opima/Marcia opima	Tike *et al.*, 2009
				Mesodesmatidae	*Coecella transversalis/Coecella horsfieldii*	Parulekar, 1981, Tike *et al.*, 2009
			Cardiida	Cardiidae	*Cardium asiaticum/Vepricardium asiaticum*	Parulekar, 1981, Tike *et al.*, 2009
					Cardium setosum/Maoricardium setosum	Tike *et al.*, 2009
					Cardium antiqualata?	Tike *et al.*, 2009
				Donacidae	*Donax incarnatus*	Parulekar, 1981, Tike *et al.*, 2009
					Donax scortum	Parulekar, 1981, Tike *et al.*, 2009
			Adapedonta	Solenidae	*Solen truncatus/Solen guinensis*	Parulekar, 1981, Tike *et al.*, 2009
			Myida	Pholadidae	*Martesia striata*	Parulekar, 1981, Tike *et al.*, 2009
	Echinodermata	Asteroidea	Paxillosida	Astropectinidae	*Astropecten indicus*	Parulekar, 1981
		Ophiuroidea	Amphilepidida	Ophiactidae	*Ophiactis savignyii*	Parulekar, 1981
		Echinoidea	Camarodonta	Temnopleuridae	*Temnopleurus toreumaticus*	Parulekar, 1981
		Holothuroidea	Holothuriida	Holothuriidae	*Holothuria scabra*	Parulekar, 1981
			Apodida	Synaptidae	*Synopta* sp./*Synapta* sp.	Parulekar, 1981
	Porifera	Demospongiae	Clionaida	Clionaidae	*Cliona thomasi*	Mote *et al.*, 2019
			Tethyida	Tethyidae	*Tethys Iyncurium/Tethya aurantium*	Parulekar, 1981
			Tetractinellida	Tetillidae	*Tetilla dactyloides*	Parulekar, 1981
Annelida	Polychaeta	Errantia	Phyllodocida	Polynoidae	*Lepidonotus carinulatus*	Parulekar, 1981

Gattyana deludens	Parulekar, 1981
Harmothoe ampullifera	Parulekar, 1981
Leanira japonica	Parulekar, 1981
Polyodontes melanonotus	Parulekar, 1981
Panthalis oerstedi	Parulekar, 1981
Bhawania cryptocephala	Parulekar, 1981
Eurythoe complanata	Parulekar, 1981
Chloeia rosea	Parulekar, 1981
Hesionidae (gen. sp.)	Sukumaran *et al.*, 2016
Hesione pantherina	Parulekar, 1981
Leocrates claparedii	Parulekar, 1981, Sukumaran *et al.*, 2016
Leocratides ehlersi	Parulekar, 1981
Podarke angustifrons	Parulekar, 1981
Syllis amica	Sukumaran *et al.*, 2016
Syllis (Haplasylle's) spongicola	Parulekar, 1981
Syllis (Sylle's) gracilis	Parulekar, 1981
Syllis (Typosylle's) closterobranchia	Parulekar, 1981
Syllis (Typosylle's) variegata	Parulekar, 1981
Syllis sp.	Sukumaran *et al.*, 2016
Nereididae (gen. sp.)	Sukumaran *et al.*, 2016
Nereis (Nereis) Chingrighattensis	Parulekar, 1981
Nereis (Nereis) chilkaensis	Parulekar, 1981
Nereis (Ceratonereis) costae	Parulekar, 1981
Nereis (Ceratonereis) mirabilis	Parulekar, 1981
Perinereis vancaurica var. Typica	Parulekar, 1981
Perinereis vancaurica var. Indica	Parulekar, 1981
Perinereis cultrifera var. Typica	Parulekar, 1981
Perinereis aibuhitensis	Parulekar, 1981
Perinereis nigro-punctata	Parulekar, 1981
Perinereis nuntia var. *Typica*	Parulekar, 1981
Perinereis nuntia var. Brevicirris	Parulekar, 1981
Eunice tentaculata	Parulekar, 1981

Eunice antennata	Parulekar, 1981
Eunicidae (gen. sp.)	Sukumaran *et al.*, 2016
Eunice sp.	Sukumaran *et al.*, 2016
Eunice pennata	Sukumaran *et al.*, 2016
Marphysa sanguines	Parulekar, 1981
Diopatra neopolitana	Parulekar, 1981
Lumbriconeries heteropoda	Parulekar, 1981
Arabella iricolor	Parulekar, 1981
Glyceridae (gen. sp.)	Sukumaran *et al.*, 2016
Glycera alba	Parulekar, 1981, Sukumaran *et al.*, 2016
Glycera longipinnis	Sukumaran *et al.*, 2016
Palydora (Polydora) coeca	Parulekar, 1981
Cirriformia limnoricola	Parulekar, 1981
Phyllochfietopterus socialis	Parulekar, 1981
Sabellaria sp.	Parulekar, 1981
Sabellariidae (gen. sp.)	Sukumaran *et al.*, 2016
Sabellidae (gen. sp.)	Sukumaran *et al.*, 2016
Pista sp.	Parulekar, 1981
Spirographis spallanzanii	Parulekar, 1981
Dasychone cingulatus	Parulekar, 1981
Dasychone serratibranchis	Parulekar, 1981
Potamilla leptochaeta	Parulekar, 1981
Onuphis sp.	UNDP, 2011
Sthenelais boa	UNDP, 2011
Vemiliopsis glandigerus	Parulekar 1981
Dendrostoma signifer	Parulekar 1981
Ochetostoma bombayense	Parulekar 1981
Polynoidae (gen. sp.)	Sukumaran *et al.*, 2016
Pholoe sp.	Sukumaran *et al.*, 2016
Eteone ornata	Sukumaran *et al.*, 2016
Phyllodoce capensis	Sukumaran *et al.*, 2016
Sigambra parva	Sukumaran *et al.*, 2016
Sigambra constricta	Sukumaran *et al.*, 2016

Oxydromus sp.	Sukumaran *et al.*, 2016
Oxydromus spinosus	Sukumaran *et al.*, 2016
Oxydromus agilis	Sukumaran *et al.*, 2016
Oxydromus angustifrons	Sukumaran *et al.*, 2016
Aglaophamus dibranchis	Sukumaran *et al.*, 2016
Micronephthys oculifera	Sukumaran *et al.*, 2016
Scoloplos uniramus	Sukumaran *et al.*, 2016
Scoloplos armiger	Sukumaran *et al.*, 2016
Armandia sp.	Sukumaran *et al.*, 2016
Cossura coasta	Sukumaran *et al.*, 2016
Cossura longocirrata	Sukumaran *et al.*, 2016
Capitellidae (gen. sp.)	Sukumaran *et al.*, 2016
Mediomastus sp.	Sukumaran *et al.*, 2016
Notomastus sp.	Sukumaran *et al.*, 2016
Parheteromastus sp.	Sukumaran *et al.*, 2016
Capitella capitata	Sukumaran *et al.*, 2016
Maldanidae (gen. sp.)	Sukumaran *et al.*, 2016
Sternaspis scutata	Sukumaran *et al.*, 2016
Pherusa sp.	Sukumaran *et al.*, 2016
Isolda pulchella	Sukumaran *et al.*, 2016
Terebellidae (gen. sp.)	Sukumaran *et al.*, 2016
Loimia batilla	Sukumaran *et al.*, 2016
Sabellastarte longa	Sukumaran *et al.*, 2016
Novafabricia bansei	Sukumaran *et al.*, 2016
Spirobranchus kraussii	Sukumaran *et al.*, 2016
Hydroides homoceros	Sukumaran *et al.*, 2016
Avifauna	
Haliastur indus	Khot, 2016
Milvus migrans	Khot, 2016
Egretta garzetta	Khot, 2016
Mesophoyx intermedia	Khot, 2016
Casmerodius albus	Khot, 2016

			Ardeola grayii	Khot, 2016
			Nycticorax nycticorax	Khot, 2016
			Butorides striata	Khot, 2016
			Ardea cinerea	Khot, 2016
			Egretta gularis	Khot, 2016
			Charadrius dubius	Khot, 2016
			Actitis hypoleucos	Khot, 2016
			Tringa	Khot, 2016
			Gallinago gallinago	Khot, 2016
Phytoplankton	Bacillariophyceae	Naviculales	*Amphiprora alata*	Hardikar *et al.*, 2017
			Amphora sp.	Hardikar *et al.*, 2017
			Asterionella glacialis	Hardikar *et al.*, 2017
			Bacillaria sp.	Hardikar *et al.*, 2017
			Bacteriastrum delicatula	Hardikar *et al.*, 2017
			Chaetoceros sp.	Hardikar *et al.*, 2017
			Corethron sp.	Hardikar *et al.*, 2017
			Coscinodiscus sp.	Hardikar *et al.*, 2017
			Cyclotella sp.	Hardikar *et al.*, 2017
			Diploneis sp.	Hardikar *et al.*, 2017
			Ditylum brightwellii	Hardikar *et al.*, 2017
			Eucampia zodiacus	Hardikar *et al.*, 2017
			Fragillaria sp.	Hardikar *et al.*, 2017
			Grammatophora sp.	Hardikar *et al.*, 2017
			Guinardia striata	Hardikar *et al.*, 2017
			Gyrosigma sp.	Hardikar *et al.*, 2017
			Hemiaulus hauckii	Hardikar *et al.*, 2017
			Leptocylindrus danicus	Hardikar *et al.*, 2017
			Licmophora juergensii	Hardikar *et al.*, 2017
			Lithodesmium undulatum	Hardikar *et al.*, 2017
			Melosira sp.	Hardikar *et al.*, 2017
			Meuniera sp.	Hardikar *et al.*, 2017
			Navicula delicatual	Hardikar *et al.*, 2017
			Navicula sp.	Hardikar *et al.*, 2017

		Odontella mobiliensis	Hardikar *et al.*, 2017
		Odontella sinensis	Hardikar *et al.*, 2017
		Planktoniella sol	Hardikar *et al.*, 2017
		Pleurosigma angulatum	Hardikar *et al.*, 2017
		Pseudonitzschia sp.	Hardikar *et al.*, 2017
		Rhizosolenia hebetata	Hardikar *et al.*, 2017
		Rhizosolenia imbricata	Hardikar *et al.*, 2017
		Skeletonema costatum	Hardikar *et al.*, 2017
		Skeletonema tropicum	Hardikar *et al.*, 2017
		Streptotheca thamensis	Hardikar *et al.*, 2017
		Surirella sp.	Hardikar *et al.*, 2017
		Synedra ulna	Hardikar *et al.*, 2017
		Thalassionema nitzschioides	Hardikar *et al.*, 2017
		Thalassiosira sp.	Hardikar *et al.*, 2017
		Thalassiothrix sp.	Hardikar *et al.*, 2017
		Ceratium furca	Hardikar *et al.*, 2017
		Dinophysis miles	Hardikar *et al.*, 2017
		Noctiluca sp.	Hardikar *et al.*, 2017
		Ornithoceros magnificus	Hardikar *et al.*, 2017
		Peridinium sp.	Hardikar *et al.*, 2017
		Prorocentrum lima	Hardikar *et al.*, 2017
		Prorocentrum micans	Hardikar *et al.*, 2017
		Protoperidinium sp.	Hardikar *et al.*, 2017
		Pyrophacus holorogicum	Hardikar *et al.*, 2017
		Actinastrum sp.	Hardikar *et al.*, 2017
		Cosmarium botrytis	Hardikar *et al.*, 2017
		Pediastrum duplex	Hardikar *et al.*, 2017
		Scenedesmus dimorphus	Hardikar *et al.*, 2017
		Staurastrum sp.	Hardikar *et al.*, 2017
		Anabaena sp.	Hardikar *et al.*, 2017
		Oscillotaria sp.	Hardikar *et al.*, 2017

				Dictyocha sp.	Hardikar *et al.*, 2017
Chlorophyta	Ulvophyceae	Dasycladales	Polyphysaceae	*Acetabularia* sp.	UNDP, 2011
		Bryopsidales	Bryopsidaceae	*Bryopisis hypnoides*	UNDP, 2011
				Bryopsis plumosa	Rode & Sabale, 2015
			Caulerpaceae	*Caulerpa scalpelliformes*	UNDP, 2011
				Caulerpa sertulariodes	UNDP, 2011
				Caulerpa mexicana	UNDP, 2011
				Caulerpa verticilata	UNDP, 2011
				Caulerpa peltata	UNDP, 2011
				Caulerpa racemosa	UNDP, 2011
			Codiaceae	*Codium indicum*	UNDP, 2011
		Ulvales	Ulvaceae	*Ulva lactuca*	Rode & Sabale, 2015
				Ulva fasiata	Rode & Sabale, 2015
				Ulva reticulata	Rode & Sabale, 2015
				Ulva taeniata	Mantri *et al.*, 20107
				Enteromorpha clathrata/Ulva clathrata	UNDP, 2011
				Enteromorpha flexuosa/Ulva flexuosa	UNDP, 2011
				Enteromorpha compressa/ Ulva compressa	Rode & Sabale, 2015
		Cladophorales	Cladophoraceae	*Chaetomorpha linum*	UNDP, 2011
				Chaetomorpha media/Chaetomorpha antennina	UNDP, 2011
				Cladophora sp.	UNDP, 2011
			Siphonocladaceae	*Ernodesmis verticillata*	UNDP, 2011
			Anadyomenaceae	*Microdictyon* sp.	UNDP, 2011
	Chlorophyceae	Chaetophorales	Chaetophoraceae	*Steigeoclonium* sp.	UNDP, 2011
				BROWN ALGAE	
Ochrophyta	Phaeophyceae	Ectocarpales	Scytosiphonaceae	*Colpomenia sinuosa*	UNDP, 2011
		Dictyotales	Dictyotaceae	*Dictyopteris australis*	UNDP, 2011
				Dictyota dichotoma	UNDP, 2011
				Dictyota dumosa	UNDP, 2011
				Dictyota bartayresiana	UNDP, 2011
				Dictyota maxima	Rode & Sabale, 2015

				Padina gymnospora	UNDP, 2011
				Padina tetrastromatica	Rode & Sabale, 2015
				Pocockiella variegata/Lobophora variegata	UNDP, 2011
				Stocehospermum marginatum/ Stoechospermum polypodioides	UNDP, 2011
		Ectocarpales	Ectocarpaceae	*Ectocarpus coniger*	UNDP, 2011
				Ectocarpus siliculosus	Rode & Sabale, 2015
			Scytosiphonaceae	*Iyengaria* sp.	UNDP, 2011
		Fucales	Sargassaceae	*Sargassum cinereum*	Rode & Sabale, 2015
				Sargassum ilicifolium	UNDP, 2011, Rode & Sabale, 2015
				Sargassum tenerrimum	Rode & Sabale, 2015
				Spatoglossum asperum	UNDP, 2011, Rode & Sabale, 2015
				Saragassum cinereum	UNDP, 2011, Rode & Sabale, 2015
				Saragassum piluliferum	UNDP, 2011, Rode & Sabale, 2015
		Sphacelariales	Sphacelariaceae	*Sphacelaria furcigera/ Sphacelaria rigidula*	UNDP, 2011
Rhodophyta	Florideophyceae	Ceramiales	Rhodomelaceae	*Acanthopora specifera*	UNDP, 2011
		Corallinales	Lithophyllaceae	*Amphiroa anceps*	Rode & Sabale, 2015
				Amphiroa sp.	UNDP, 2011
			Corallinaceae	*Corallina berteroi*	Rode & Sabale, 2015
				Cheliosporum sp.	UNDP, 2011
		Gelidiales	Gelidiaceae	*Gelidium pusillum*	UNDP, 2011
				Gelediella acerosa	Rode & Sabale, 2015
		Nemaliales	Scinaiaceae	*Scinaia hatei*	UNDP, 2011
			Liagoraceae	*Liagora albicans*	Rode & Sabale, 2015
		Gigartinales	Solieriaceae	*Soliera robusta*	UNDP, 2011
		Rhodymeniales	Lomentariaceae	*Gelidiopsis variabillis*	UNDP, 2011
		Gracilariales	Gracilariaceae	*Gracillaria corticata*	UNDP, 2011, Rode & Sabale, 2015
		Halymeniales	Halymeniaceae	*Grateloupia filicina*	UNDP, 2011
		Ceramiales	Delesseriaceae	*Martensia fragillis*	UNDP, 2011

		Ahnfeltiales	Ahnfeltiaceae	*Ahnfeltia plicata*	Rode & Sabale, 2015
		Gigartinales	Cystocloniaceae	*Hypnea musciformis*	Rode & Sabale, 2015
	Bangiophyceae	Bangiales	Bangiaceae	*Bangia fuscopurpurea*	Untawale
				Porphyra vietnamensis	Rode & Sabale, 2015
	Florideophyceae	Ceramiales	Ceramiaceae	*Centroceros clavulatum*	UNDP, 2011

Chapter 5

5. Coral reef-associated ichthyofaunal diversity in the Malvan Marine Sanctuary

5.1. Introduction

Reef fishes are one of the most numerous, colorful, intriguing, and heterogeneous species that inhabit coral reefs, the most biodiverse ecosystem on Earth (Moberg & Folke, 1999). However, reef fishes are vulnerable to declines globally due to coral reef degradation (Graham *et al.,* 2006, 2007; Wilson *et al.,* 2008). The complex physical structures formed by reef-forming corals provide a protected environment from the surrounding open ocean, which influences the fish population by providing a conducive habitat and food resources (Ménard *et al.,* 2012). Reef fishes are the conspicuous component of the healthy coral reef ecosystem and recognized as a keystone group in the coral reef for their significant role in structuring coral communities and enhancing reef resilience (Graham *et al.,* 2006; Green and Bellwood 2009). Herbivores and grazers in the coral reef control macroalgal and turf algal growth and, therefore, aid coral larval settlement, reduced benthic space competition, overall reef health and resilience (Heenan & Williams, 2013). Globally, the depletion of herbivore fish community through intensive fishing and habitat loss has often resulted in the undesirable state in coral reefs from coral dominance to the algal proliferation (Mumby *et al.,* 2006). Additionally, diminution of herbivores may trigger to overgrow coral pathogenic microbes, eventually negatively impact the reef health by triggering coral disease outbreak (Graham *et al.,* 2012; Sweet *et al.,* 2013).

Information on distribution, biodiversity, and ecology of coral reef fishes have been reported from the major coral reef areas of India, viz., Andaman and Nicobar Islands, Gulf of Mannar Biodiversity Reserves, Lakshadweep Islands, and Gulf of Kachchh. However, little is known on reef-dwelling fishes from the minor patch coral reefs along the West coast of India. This investigation aimed to document the reef-associated fishes in the coral reef of the Malvan Marine Sanctuary (MMS), a Marine Protected Area on the West coast of India. Malvan is also one of the major fish landing centers of economic importance on the west coast of India, known for its high marine fish and shellfish diversity (Tike *et al.,* 2009). Previous studies on fish diversity from the MMS region were mostly focused on the fishes of commercial importance and landed by the fishing activities of local fishers (Tike *et al.,* 2009; Parulekar, 1981; Barman *et al.,* 2007). Instead of the presence of fragile coral reef region with enormous ecological and socio-economic significance, very little information is known on the occurrence and diversity of reef-associated fishes from this Marine Protected Area. Considering the increasing environmental and anthropological pressure on corals at MMS, and the complex functioning roles of reef fishes in the reef environment, documentation of the occurrence and distribution associated ichthyofauna is critically essential.

The present study provides baseline information on the occurrence and distribution of reef fishes in the MMS, based on systematic underwater surveys conducted around major patch reefs. This study also points out the threats on the MMS and emphasis on urgent management intervention to protect the vulnerable coral reef habitat.

5.2.Materials and methods

5.2.1. Study area

The Malvan Marine Sanctuary (MMS), a designated Marine Protected Area (MPA), located along the central west coast of India and spread over 29.122 km2 area. Malvan Bay is protected by numerous rocky outcrops and a small low fortified Sindhudurg Island. Patch reefs are distributed in shallow protected areas along with the MPA. Underwater surveys by SCUBA diving were conducted at five patch reefs (T1 to T5) across the MMS from October 2015 to April 2019 (Table 4.1; Fig. 4.1). Reef T1, T2, and T3 are located at the core zone of the MPA and are extensively used as commercial, recreational diving, and snorkeling sites by local tourism operators, whereas T4 and T5 are fishing sites for local fishers. Recently, T5 also become a recreational diving site. MMS is well protected from wave action by Sindhudurg Island and a chain of submerged and exposed rocks. The patch reefs in the Malvan bay are primarily composed of a massive and encrusting form of Porites spp. and foliose form of *Turbinaria mesenterina* (De *et al.,* 2015, see species list in Table 2). High rainfall, rapidly changing land use, and terrigenous runoff cause relatively higher suspended particles and dissolved nutrients in the MMS (Hussain *et al.,* 2016). Average water depth ranges between 2-10 meters along the study area (De *et al.,* 2020).

5.2.2. Ichthyofaunal survey and data collection

Underwater visual census was conducted by two SCUBA divers at each patch reef following the method described by Halford and Thompson (1994), one of the non-destructive approaches offered to assess coral reef fish communities. Underwater digital photographs were clicked using Nikon AW120 (14 megapixels, Japan) and GoPro Hero4 (12 megapixels, USA) underwater camera. Initial identification of the fishes was carried out during underwater observation. Images were used for further confirmation at best possible taxonomic resolution by using available identification keys (Talwar & Kacker, 1984; Murugan & Namboothri, 2012; Allen, 1991; Randall *et al.*, 1990). Online database resources, such as FishBase ver. 02/2019 (www.fishbase.org), World Register of Marine Species: WoRMS (http://www.marinespecies.org/), the IUCN red list of threatened species (https://www.iucnredlist.org/), and bioSearch (http://www.biosearch.in) were also used for identification keys, taxonomic validity, distribution range, and ecology. No collection of fish specimens or damage to coral reefs was done during the surveys.

5.3. Results

Patterns in species richness

During the present study, 47 species of reef fishes belonging to 35 genera and 26 families were encountered from the MMS coral reefs (Fig. 5.4-5.5; Table 3). Maximum representation was shown by the Pomacentridae family with four genera and seven species, followed by Chaetodontidae with two genera and six species (Fig. 5.1). We observed that the distribution of reef fishes varies between the studied sites, the highest species richness was observed at T2 (n= 21), followed by T1 (n=17), T3 (n= 16) around the Sindhudurg Island. However, 15 and 14 numbers of species were reported at reef stations T4 and T5, respectively. Whereas highest density of reef fishes was observed at patch reef location T1, T2, T3, mostly comprised of *Neopomacentrus violascens* (Bleeker, 1848), *N. cyanomos* (Bleeker, 1856), *Abudefduf bengalensis* (Bloch, 1787), *A. sordidus* (Forsskål, 1775), and *Cheiloprion labiatus* (Day, 1877), these species form large school and attract thousands of SCUBA diving enthusiastic.

Ten fish species were observed to be common in all the study sites, namely, *Halichoeres leucurus* (Walbaum, 1792), *C. labiatus* (Day, 1877), *N. violascens* (Bleeker, 1848), *N.*

cyanomos (Bleeker, 1856), *Cryptocentrus caeruleopunctatus* (Rüppell, 1830), *Scatophagus argus* (Linnaeus, 1766), *Cephalopholis formosa* (Shaw, 1812), *Blenniella periophthalmus* (Valenciennes, 1836), *Salarias faciatus* (Bloch, 1786), and *Chaetodon collare* Bloch, 1787.

Site T5, showed a unique species assemblage (n=8) includes *Cephalopholis sonnerati* (Valenciennes, 1828), *Karalla daura* (Cuvier, 1829), *Zanclus cornutus* (Linnaeus, 1758), *Chilomycterus reticulatus* (Linnaeus, 1758), *Caesio teres* Seale, 1906, *Plectorhinchus chubbi* (Regan, 1919), *Gymnothorax griseus* (Lacepède, 1803), *Gymnothorax favagineus* Bloch & Schneider, 1801. Followed by Site T4 (n=4) includes *Scolopsis vosmeri* (Bloch, 1792), *Pomacanthus annularis* (Bloch, 1787), *Lutjanus argentimaculatus* (Forsskål, 1775), *Scorpaenopsis venosa* (Cuvier, 1829) and T1 (n=4) comprises *Heniochus monoceros* Cuvier, 1831, *Pomadasys guoraca* (Cuvier, 1829), *Odonus niger* (Rüppell, 1836), *Epinephelus coioides* (Hamilton, 1822). The presence of three unique species was recorded at the T2, includes *Labroides dimidiatus* (Valenciennes, 1839), *Caesio xanthonota* Bleeker, 1853, *Chaetodon lineolatus* Cuvier, 1831. *Archamia fucata* (Cantor, 1849), and *Scarus ghobban* Forsskål, 1775, were only found at the T3 location (Fig.5.2).

Among the reported fish species, *N. cyanomos* belong to the vulnerable category of IUCN, whereas 28 fishes belong to Least Concern (LC), and six have not been evaluated.

Taxonomic description of all the recorded species is discussed below:

Phylum: Chordata

Subphylum: Vertebrata

Superclass: Osteichthyes

Class: Actinopterygii

Subclass: Neopterygii

Division: Teleostei

Order Tetraodontidornmes

Family Anguilliformes

Genus *Gymnothorax* Bloch, 1795

Gymnothorax griseus (Lacepède, 1803)

Gymnothorax favagineus Bloch & Schneider, 1801

Order: Perciformes

Suborder: Percoidei

Family: Serranidae

Genus: *Cephalopholis* Bloch and Schneider, 1801

Cephalopholis formosa (Shaw, 1812)

Cephalopholis sonnerati (Valenciennes, 1828)

Genus: *Epinephelus* Bloch, 1793

Epinephelus malabaricus (Bloch and Schneider, 1801)

Family Apogonidae

Genus *Archamia* Gill, 1863

Archamia fucata (Cantor, 1849)

Genus *Ostorhinchus* Lacepède, 1802

Ostorhinchus fasciatus (White, 1790)

Family Pempheridae

Genus *Pempheris* Cuvier, 1829

Pempheris vanicolensis Cuvier, 1831

Family Leiognathidae

Genus *Leiognathus* Lacepède, 1802

Karalla daura (Cuvier, 1829)

Family Pomacanthidae

Genus *Pomacanthus* Lacepede, 1802

Pomacanthus annularis (Bloch, 1787)

Family Chaetodontidae

Genus *Chaetodon* Linnaeus, 1758

Chaetodon collare Bloch, 1787

Chaetodon dolosus Ahl, 1923

Chaetodon decussatus Cuvier, 1829

Chaetodon lineolatus Cuvier, 1831

Genus Heniochus Cuvier, 1816

Heniochus acuminatus (Linnaeus, 1758)

Heniochus monoceros Cuvier, 1831

Family Lutjanidae

Genus *Lutjanus* Bloch, 1790

Lutjanus argentimaculatus (Forsskål, 1775)

Family Nemipteridae

Genus: *Scolopsis* Cuvier, 1814

Scolopsis vosmeri (Bloch, 1792)

Family Caesionidae

Genus *Caesio* Lacepède, 1801

Caesio xanthonota Bleeker, 1853

Caesio teres Seale, 1906

Family Sparidae

Genus *Acanthoparus* Peters, 1855

Acanthopagrus berda (Forsskål, 1775)

Family Scatophagidae

Genus *Scatophagus* Cuvier, 1831

Scatophagus argus (Linnaeus, 1766)

Family Haemulidae

Genus *Pomadasys* Lacepède, 1802

Pomadasys guoraca (Cuvier, 1829)

Genus Plectorhinchus Lacepède, 1801

Plectorhinchus chubbi (Regan, 1919)

Suborder Labroidei

Family Pomacentridae

Genus: *Abudefduf* Forsskal, 1775

Abudefduf bengalensis (Bloch, 1787)

Abudefduf sordidus (Forsskål, 1775)

Abudefduf vaigiensis (Quoy & Gaimard, 1825)

Genus: *Cheiloprion* Weber, 1913

Cheiloprion labiatus (Day, 1877)

Genus: *Chrysiptera* Swainson, 1839

Chrysiptera unimaculata (Cuvier, 1830)

Genus *Neopomacentrus* Allen, 1975

Neopomacentrus cyanomos (Bleeker, 1856)

Neopomacentrus violascens (Bleeker, 1848)

Family: Labridae

Genus: *Halichoeres* Rüppell, 1835

Halichoeres leucurus (Walbaum, 1792)

Genus: Thalassoma Swainson, 1839

Thalassoma lunare (Linnaeus, 1758)

Genus: Labroides Bleeker, 1851

Labroides dimidiatus (Valenciennes, 1839)

Family Scaridae

Genus Scarus Forsskål, 1775

Scarus ghobban Forsskål, 1775

Family Acanthuridae

Genus *Acanthurus* Forsskål, 1775

Acanthurus gahhm (Forsskål, 1775)

Family Siganidae

Genus *Siganus* Forsskål, 1775

Siganus vermiculatus (Valenciennes, 1835)

Family Zanclidae

Genus *Zanclus* Cuvier, 1831

Zanclus cornutus (Linnaeus, 1758)

Suborder Gobioidei

Family Gobiidae

Genus Istigobius Whitley, 1932

Istigobius ornatus (Rüppell, 1830)

Suborder Blennioidei

Family Blenniidae

Genus *Blenniella* Reid, 1943

Blenniella periophthalmus (Valenciennes, 1836)

Genus *Salarias* Cuvier, 1816

Salarias fasciatus (Bloch, 1786)

Suborder Mugiloidei

Family Mugilidae

Genus *Mugil* Linnaeus, 1758

Mugil cephalus Linnaeus, 1758

Order Scorpaeniformes

Suborder Scorpaeninae

Family Scorpaenidae

Genus *Scorpaenopsis* Heckel, 1839

Scorpaenopsis venosa (Cuvier, 1829)

Genus *Pterois* Oken, 1817

Pterois volitans (Linnaeus, 1758)

Order Tetraodontidornmes

Family Balistidae

Genus Odonus Gistel, 1848

Odonus niger (Ruppell, 1836)

Family Diodontidae

Genus *Chilomycterus* Brisoutde Barneville, 1846

Chilomycterus reticulatus (Linnaeus, 1758)

Family Tetradontidae

Genus *Arothron* Müller, 1841

Arothron immaculatus (Bloch & Schneider, 1801)

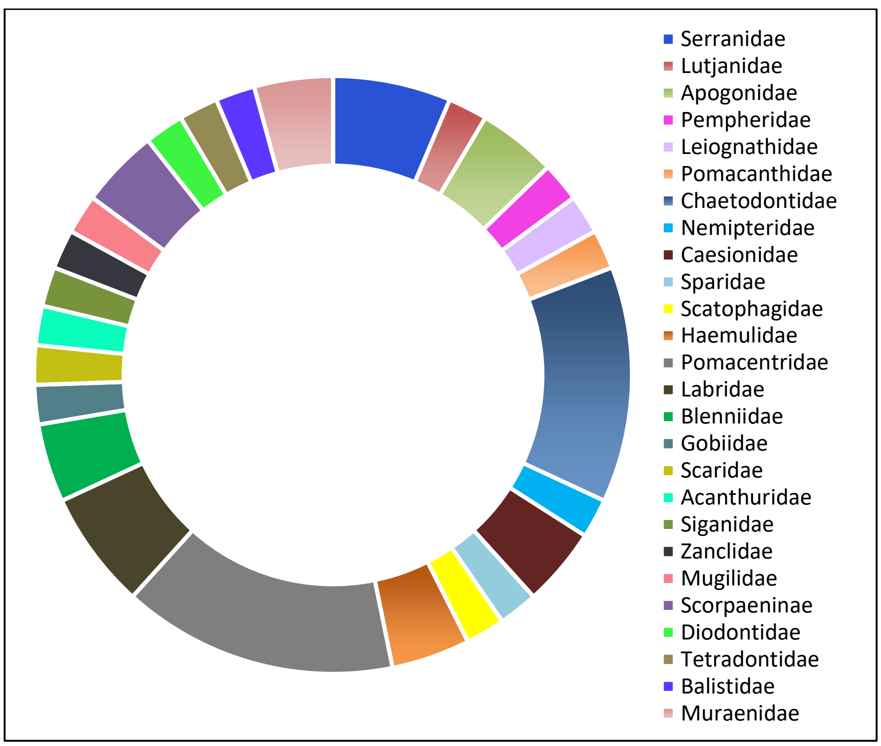

Fig. 5.1. Major families of fishes contributing to the reef fish diversity (%) in the MMS

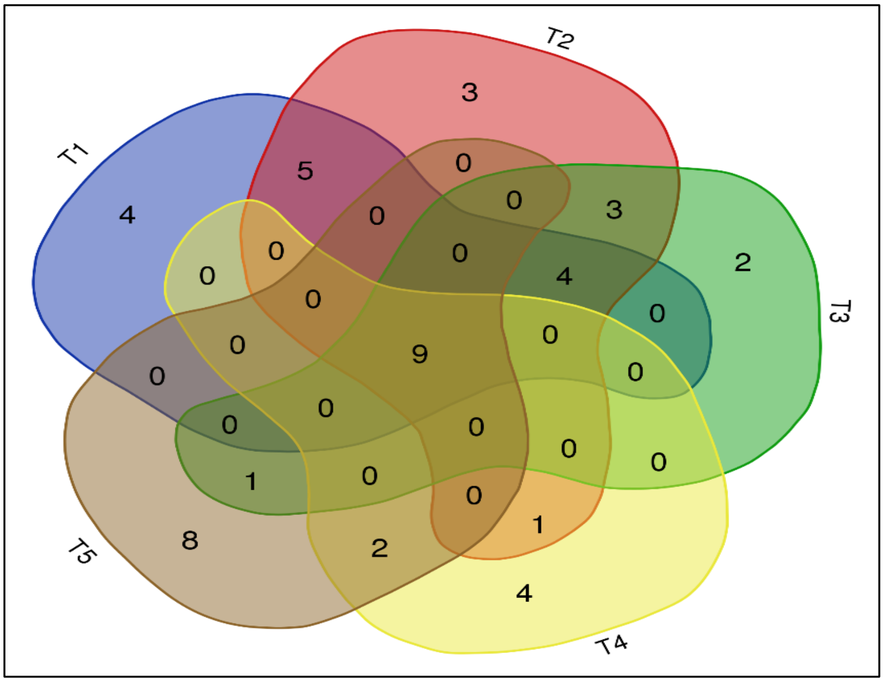

Fig. 5.2. Venn diagram representing the differences in reef fish assemblage at different patch reef site at the MMS

5.4.Discussion

Documentation and monitoring of fish assemblages in the coral reef is crucial to assess the health and resilience of a coral reef and also to achieve the conservation and management. The Malvan Marine Sanctuary was designated in 1987; however, little is known on the diversity and distribution of reef-associated fishes from the patch coral reefs across the MMS. During the present study, 47 species of reef-associated fishes were documented from the MMS. These reef fishes depend upon coral reefs for food, shelter, and breeding grounds. We observed that the presence of shallow water patch coral reefs provides ideal nurseries to the fishes by offering shelter for juvenile fishes and buffers the wave action, which is easier for the fish juveniles to locomotion, feeding, and identify predators (Ménard *et al.*, 2012).

The reef fish diversity in the MMS is comparatively lower when compared with the major coral reef ecosystems along the west coast of India, like the Gulf of Kachchh and the Lakshadweep Islands (Fig.5.3). However, the extent of diversity corroborated with the other patch coral reef habitats along the west coast of India, like Grande Island in Goa coast, where 36 reef-associated fish species were reported (Sreekanth *et al.*, 2019), and also from the preliminary findings from the Angriya bank coral reef off Maharashtra coast represented by 18 species reef-associated fishes (NIO, 2016). On the other hand, a relatively high number of fish species recorded from the patch reefs around Netrani Island in Karnataka coast accounted to be 92 fish species belongs to 35 families and 58 genera, which is maybe due to relatively better reef health and low human pressure like tourism, land-based pollution, and fishing (Zacharia *et al.*, 2002).

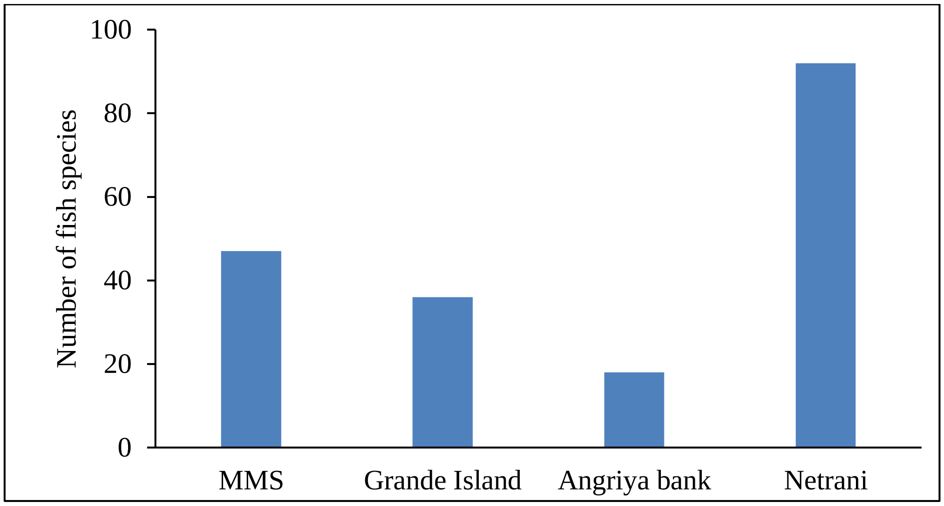

Fig. 5.3. Reef fish assemblage in different coral reef habitats along the central west coast of India (data source Sreekanth *et al.*, 2019; NIO, 2016; Zacharia *et al.*, 2008)

The presence of lionfish *Pterois volitans* is of particular concern for reef health as this predatory species prey on native herbivore reef fishes and invertebrates, posing a threat to herbivore abundance in the reef and negatively impacts the reef ecosystem (Green *et al.*, 2012). Fish families like Caesionidae, Serranidae, and Lutjanidae contributed to the abundance of commercial food fish species. In contrast, Pomacentridae, Chaetodontidae, Pomacanthidae, and Tetraodontidae contribute to ornamental species assemblage (Sreekanth *et al.*, 2019).

Corals in Malvan has been undergoing severe bleaching events and associated coral mortality in recent years because of elevated sea surface temperature triggered by climate change. During 2014, ~15% of coral colonies were affected by bleaching (De *et al.*, 2015). Further, the prevalence of coral bleaching increased drastically to 70.93% during December 2015, with a mortality of about 8.38% (Divyaraj *et al.*, 2018). Additionally, coral diseases like white syndrome and the coral tumor or growth anomalies were observed across the reefs in the MMS (Hussain *et al.*, 2016; De *et al.*, 2020 unpublish). The occurrence of recurrent bleaching events and coral mortality, as well as the outbreak of coral diseases, indicate corals in the MMS under severe stress (De *et al.*, 2020).

Moreover, we observed artisanal fishing activity in the reef area with the gill net and cast net during our surveys. Fishing activities inside the core protected area could lead to the declination of critical functional groups of reef species, which may result in a cascading impact on reef-associated species and may further disrupt the reef resilience to multiple stressors (Mumby *et al.,* 2007). The accordance of sanctuary status (MPA) to this region has always been opposed by the local fishermen because of their apprehension that they would be denied access to their traditional fishing grounds (Rajagopalan, 2008). Concurrently, in the absence of stringent management practice, destructive tourism practices, fishing, and poaching activity continues to date (De *et al.*, 2020). Fishing in the reef poses serious threats to corals by the removal of key herbivore fish species, as they control algal abundance by grazing (Chung *et al.*, 2019). Effective management of no-take region in coral reef areas where fishing is prohibited has proven to improve the reef health and resilience capacity. Therefore, it is crucial to devise a local fishery management policy and reef protection framework from local stressors in order to maintain the reef resilience in the face of global climate change and intensifying human influence (Weijerman *et al.,* 2018).

Coral reef and associated stunning colored reef fishes attract thousands of recreational tourists around the year and serve as one of the critical sources of livelihood to the local population in Malvan. In recent years, MMS has witnessed a dramatic increase in low-cost SCUBA tourism in the coral reef. However, it has improved the local economy, but, a large number of tourist (mostly untrained, and first-timer) on a relatively small coral reef has resulted in mass mechanical coral damage and detachment of fragile corals due to trampling and boat anchoring, which is already in severe stress due to catastrophic coral bleaching (De *et al.*, 2017; De *et al.*, 2020). Additionally, artificial feeding of the reef fishes by SCUBA diving operators was observed during the study period to attract fish aggregation around the diving tourist for photographs (De *et al.,* 2020). High abundance and unnatural aggregation of fish in big schools were observed at all the diving sites T1, T2, and T3, comprised of *A. bengalensis, A. sordidus, N. cyanomos, C. labiatus*, *A. berda*, these species also showed affinity the tourist recreational feeding. Feeding of reef fishes is reported to be detrimental to reef health for resulting in trophic alteration, malnutrition, and catalyze reef degradation (Brookhouse *et al.,* 2013; de Paula *et al.*, 2018). A high abundance of omnivorous generalists or opportunistic species leads to low species heterogeneity of the community (reviewed in de Mattos & Yeemin, 2018). Therefore, unsustainable tourism and fishing practice in the protected coral reef in the MMS poses serious threats to already stressed corals by recent mass bleaching events, and hinder recovery process and resilience to the disturbances.

Despite being a hotspot of ecological and economic importance, scientific documentation of the coral reef biodiversity and mapping of the threats to marine life yet to be complete in the MMS, which could assist in regulatory interventions from the conservation point of view. Hence, habitat restoration and management practice should be focused on improving the reef fish population with limiting reef resource exploitation will help the MMS coral reef to achieve resilience to the local and climate-induced stressors.

Table 5.1. Coral reef-associated ichthyofauna reported from MMS during the present study along with their occurrence, relative abundance feeding habits, and IUCN red list status.

Species Name	Common name	Occurrence site	Relative abundance	Feeding guild	Food/diet	IUCN status
Gymnothorax griseus (Lacepède, 1803)	Geometric moray eel	T5	Rare	Carnivore	crustaceans, small fish	LC
Gymnothorax favagineus Bloch & Schneider, 1801	Honeycomb moray	T5	Rare	Carnivore	crustaceans, small fish	LC
Cephalopholis formosa (Shaw, 1812)	Blue lined grouper	All sites	Rare	Carnivore	small fish, mollusk, crustacean	LC
Cephalopholis sonnerati (Valenciennes, 1828)	Tomato Hind	T5	Rare	Carnivore	small fish, mollusk, crustacean	LC
Epinephelus malabaricus (Bloch and Schneider 1801)	Malabar grouper	T1	Rare	Carnivore	small fish, molluscs, crustaceans	LC
Archamia fucata (Cantor, 1849)	Orangetailed cardinal fish	T3	Common	Carnivore	crustaceans, invertebrate	NE
Ostorhinchus fasciatus (White, 1790)	Broad-banded cardinalfish	T2, T3	Common	Invertivore	zooplankton, crustaceans, invertebrates	NE
Pempheris vanicolensis (Cuvier, 1831)	Vanikoro sweeper	T3, T5	Common	Planktivore	zooplankton, crustaceans	NE

Karalla daura (Cuvier, 1829)	Goldstripe ponyfish	T5	Rare	Omnivore	crustaceans, worms, molluscs, invertebrates	NE
Pomacanthus annularis (Bloch, 1787)	Blue-ringed angelfish	T4	Rare	Omnivore	algae, zooplankton, sponges, tunicates, coral polyps, small fishes	LC
Chaetodon collare (Bloch, 1787)	White collared butterfly fish	All sites	Common	Omnivore	algae, coral polyps, crustaceans, invertebrate	LC
Chaetodon decussatus Cuvier, 1829	Indian vagabond butterflyfish	T1, T2	Rare	Invertivore	benthic invertebrates, crustaceans	LC
Chaetodon dolosus (Ahl, 1923)	African butterfly fish	T1, T2	Rare	Invertivore	crustaceans, invertebrate	LC
Chaetodon lineolatus (Cuvier, 1831)	Lined butterflyfish	T2	Rare	Carnivore	coral polyps, anemones, algae, benthic invertebrate	LC
Heniochus acuminatus (Linnaeus, 1758)	Bannerfish/ Pennant coral fish	T2, T3	Rare	Carnivore	coral polyp, crustaceans, benthic invertebrates,	LC
Heniochus monoceros Cuvier, 1831	Masked bannerfish	T1	Rare	Carnivore	benthic invertebrates	LC
Lutjanus argentimaculatus (Forsskål, 1775)	Mangrove red snapper	T4	Rare	Carnivore	small fish, crustaceans	LC
Scolopsis vosmeri (Bloch, 1792)	Whitecheek monocle bream	T4	Rare	Invertivore	crustaceans, benthic invertebrate	NE
Caesio xanthonota (Bleeker, 1853)	Yellowback fusilier	T2	Rare	Invertivore	zooplankton, crustaceans, invertebrate	LC
Caesio teres (Seale, 1906)	Yellow and blueback fusilier	T5	Rare	Invertivore	zooplankton, crustaceans, invertebrate	NE
Acanthopagrus berda (Forsskål, 1775)	Goldsilk seabream	T2	Rare	Carnivore	small fish, echinoderms, worms, crustaceans, molluscs	LC
Scatophagus argus (Linnaeus, 1766)	Spotted Scat	All sites	Common	Omnivore	algae, zooplankton, crustaceans, invertebrates	LC
Pomadasys guoraca (Cuvier, 1829)	Guoraca Grunter	T1	Rare	Carnivore	crustaceans, molluscs, invertebrates	LC

Plectorhinchus chubbi (Regan, 1919)	Dusky Sweetlips	T5	Rare	Invertivore	benthic invertebrates, crustaceans, worms	LC
Abudefduf sordidus (Forsskål, 1775)	Black spot sergeant	T1, T2, T3	Common	Omnivore	algae, crustaceans, small invertebrates	LC
Abudefduf bengalensis (Bloch, 1787)	Bengal sergeant	T1,T2, T3	Common	Omnivore	algae, crustaceans, small invertebrates	LC
Abudefduf vaigiensis (Quoy & Gaimard, 1825)	Indo-Pacific sergeant	T1, T2, T3	Common	Omnivore	algae, crustaceans, invertebrates	LC
Cheiloprion labiatus (Day, 1877)	Big lip damsel	All sites	Common	Omnivore	algae, coral polyps, invertebrates	NE
Chrysiptera unimaculata (Cuvier, 1830)	One spot damsel	T1,T2, T3	Rare	Omnivore	algae, crustaceans, small invertebrates	LC
Neopomacentrus cyanomos (Bleeker, 1856)	Regal Demoiselle	All sites	Common	Omnivore	algae, zooplankton, crustaceans, small invertebrates	VU
Neopomacentrus violascens (Bleeker, 1848)	Violet demoiselle	All sites	Common	Omnivore	algae, zooplankton, crustaceans, small invertebrates	NE
Halichoeres leucurus (Walbaum, 1792)	Chain lined wrasse	All sites	Common	Invertivore	crustaceans, small invertebrates	LC
Thalassoma lunare (Linnaeus, 1758)	Moon Wrasse	T2, T3	Rare	Carnivore	benthic invertebrates, worms, fish egg	LC
Labroides dimidiatus (Valenciennes, 1839)	Bluestreak cleaner wrasse	T2	Common	Ectoparasite feeder	parasitic copepods, invertebrates	LC
Scarus ghobban Forsskål, 1775	Blue-barred parrotfish	T3	Rare	Herbivore	benthic algae/weeds	LC
Acanthurus gahhm (Forsskål, 1775)	Black Surgeon	T4, T5	Rare	Omnivore	zooplankton, benthic algae, detritus, invertebrates	LC
Siganus vermiculatus (Valenciennes, 1835)	Rabbit fish/ Vermiculated Spinefoot	T1, T2	Common	Herbivore	algae, seaweed	LC
Zanclus cornutus (Linnaeus, 1758)	Moorish idol	T5	Rare	Invertivore	sponges, invertebrates	LC
Mugil cephalus Linnaeus, 1758	Flathead grey mullet	T1, T2	Common	Omnivore	diatoms, algae, copepods, decayed organic matter	LC

Scorpaenopsis venosa (Cuvier, 1829)	Raggi scorpion fish	T4	Rare	Carnivore	small fish, crustacean	LC
Pterois volitans (Linnaeus, 1758)	Red lion fish	T4, T2	Rare	Carnivore	small fish, crustaceans, benthic invertebrates	LC
Istigobius ornatus (Rüppell, 1830)	Ornate goby	T1, T2, T3	Common	Invertivore	benthic invertebrates	LC
Blenniella periophthalmus (Valenciennes, 1836)	Blue-dashed rockskipper, Bullethead rockskipper	All sites	Common	Omnivore	algae, benthic invertebrates	LC
Salarias fasciatus (Bloch, 1786)	Jewelled blenny	All sites	Common	Omnivore	benthic algae, weeds, crustaceans, detritus	LC
Odonus niger (Rüppell, 1836)	Redtoothed trigger fish	T1	Rare	Carnivore	benthic invertebrates, small fish, molluscs, crustaceans, worms	NE
Chilomycterus reticulatus (Linnaeus, 1758)	Spotted burrfish	T5	Rare	Carnivore	molluscs, sea urchins, crustaceans	LC
Arothron immaculatus (Bloch & Schneider, 1801)	Immaculate Puffer	T4, T5	Rare	Carnivore	crustaceans, molluscs	LC

VU: Vulnerable, NE: Not Evaluated, DD: Data deficient, LC: Least concern

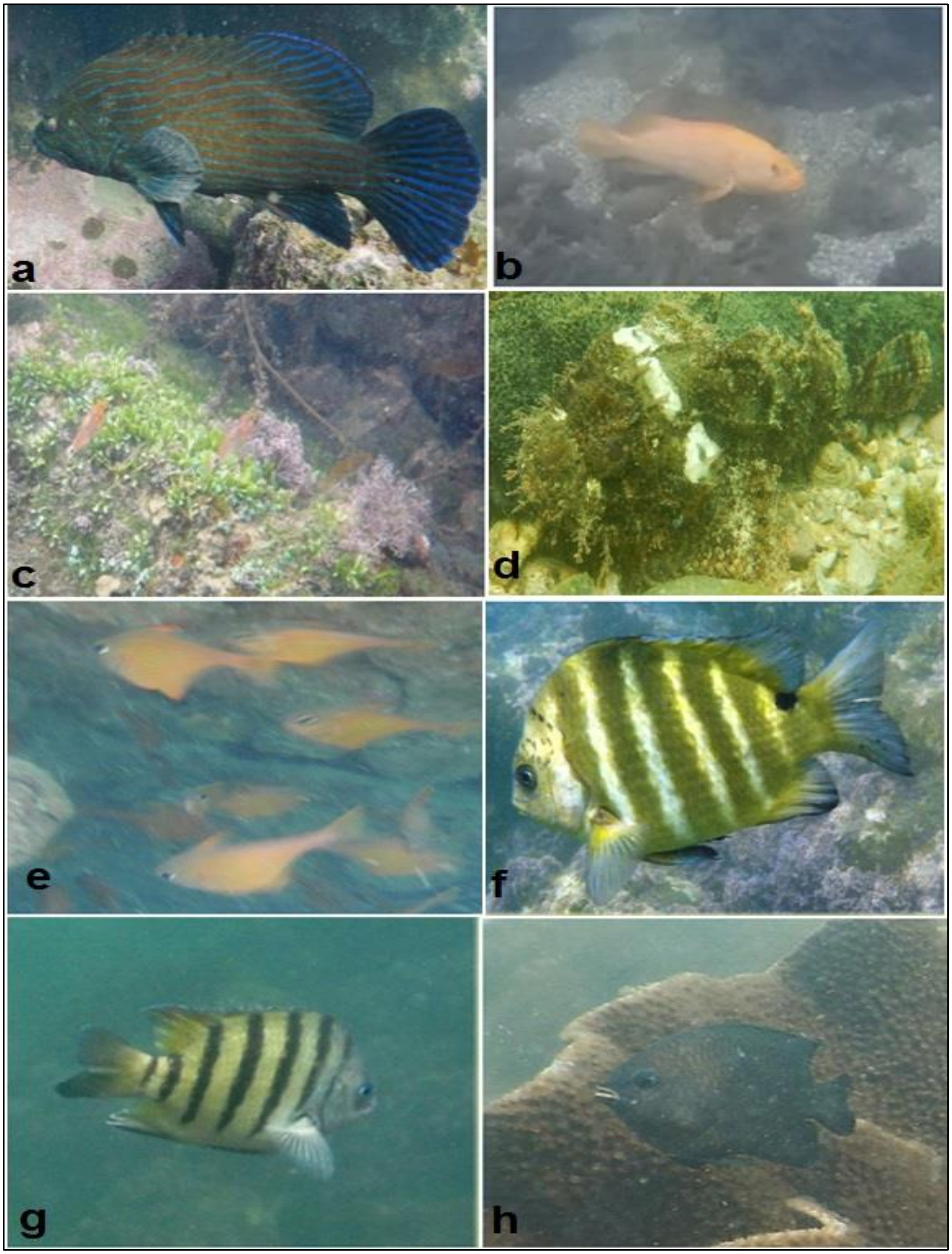

Fig. 5.4 a) Cephalopholis formosa; b) Cephalopholis sonnerati; c) Archamia fucata; d) Scorpaenopsis venosa; e) Pempheris vanicolensis; f) Abudefduf bengalensis; g) Abudefduf sordidus; h) Cheiloprion labiatus

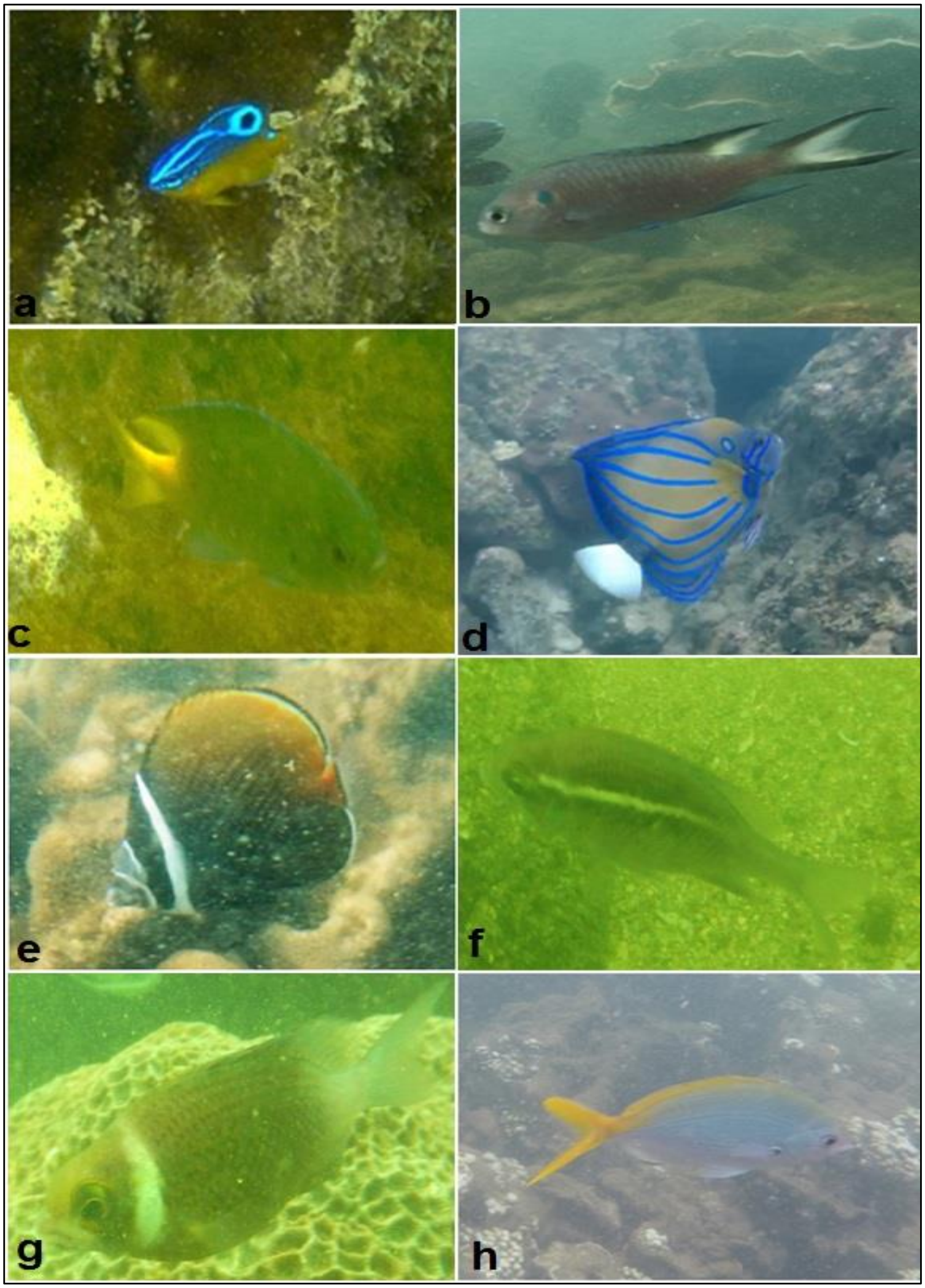

Fig. 5.5. a) Chrysiptera unimaculata; b) Neopomacentrus cyanomos; c) Neopomacentrus violascens; d) Pomacanthus annularis; e) Chaetodon collare; f) Pentapodusaureo fasciatus; g) Scolopsis vosmeri; h) Caesio xanthonota

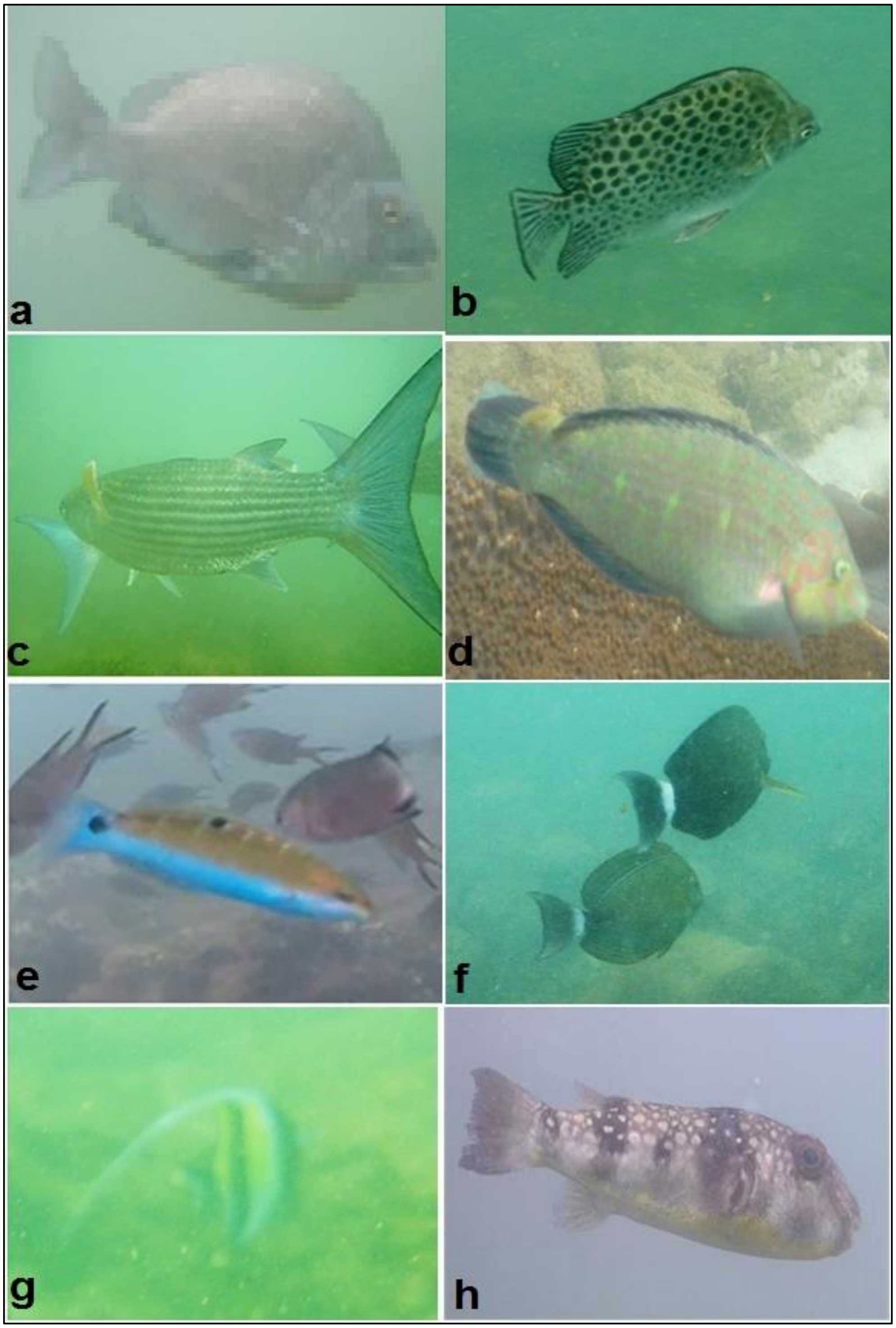

Fig. 5.6. a) Acanthoparus berda; b) Scatophagus argus; c) Mugil cephaluss; d) Halichoeres leucurus; e) Thalassoma lunare; f) Acanthurus gahhm; g) Zanclus cornutus; h) Chilomycterus reticulatus

Chapter 6

6. Abundance and seasonal variation in turf and macroalgal cover

6.1. Introduction

Globally, coral reefs are facing an existential crisis because of the intensified climate change and anthropogenic perturbations. The catastrophic impact of climate change over the global reef ecosystem is otherwise shown to benefit the proliferation of other associated communities of macroalgae. The events of community shift in the coral reef ecosystem potentially jeopardize many of their ecological, economic, and social services (Smith *et al.,* 2006). Corals and seaweeds are dominant habitat developers in the reef ecosystem, the substantial differences in their sensitivities to changing seawater chemistry have been reported to cause significant variation in their abundances. The increase in seaweed abundance and canopy cover with a concomitant decrease in coral colonies is a well-documented event in Jamaican reefs since the 1980s (Hughes, 1994).

Macroalgae are the major competitors of corals for space on tropical reefs (Rasher & Hay, 2010; Rasher *et al.,* 2011). They are taking over reefs that have been weakened by multiple stressors including recurrent bleaching events, sedimentation, eutrophication, overfishing, and storms (Figure 6.1) (Mccook *et al.,* 2001; Bellwood *et al.,* 2004). Being an opportunistic species with high reproductive capacity and fast growth rate, macroalgae often outcompete slow-growing coral species. Under the ecological conditions favouring coral health, the sovereignty of seaweed abundance is broadly governed by herbivorous fishes and nutrient availability (Rasher *et al.,* 2012). The eutrophication and intensive fishing are the major anthropogenic stressors that are shown to promote the seaweed growth in the reef ecosystem while severely affecting corals (Walsh, 2011; Rasher *et al.,* 2012; D'Angelo & Wiedenmann, 2014). The seaweed population, once increased in a reef system, causes detrimental effects to coral through allelopathic chemical extrudes and photosynthetic abrasion by the shading effect (Rasher & Hay, 2010; Del Monaco *et al.,* 2017). Thus, this competitive advancement of macro and turf algae accelerates the ecosystem regime shifts from coral to algal dominance in the reef ecosystem (Littler *et al.,* 2006; Fung *et al.,* 2011) (Fig. 6.1).

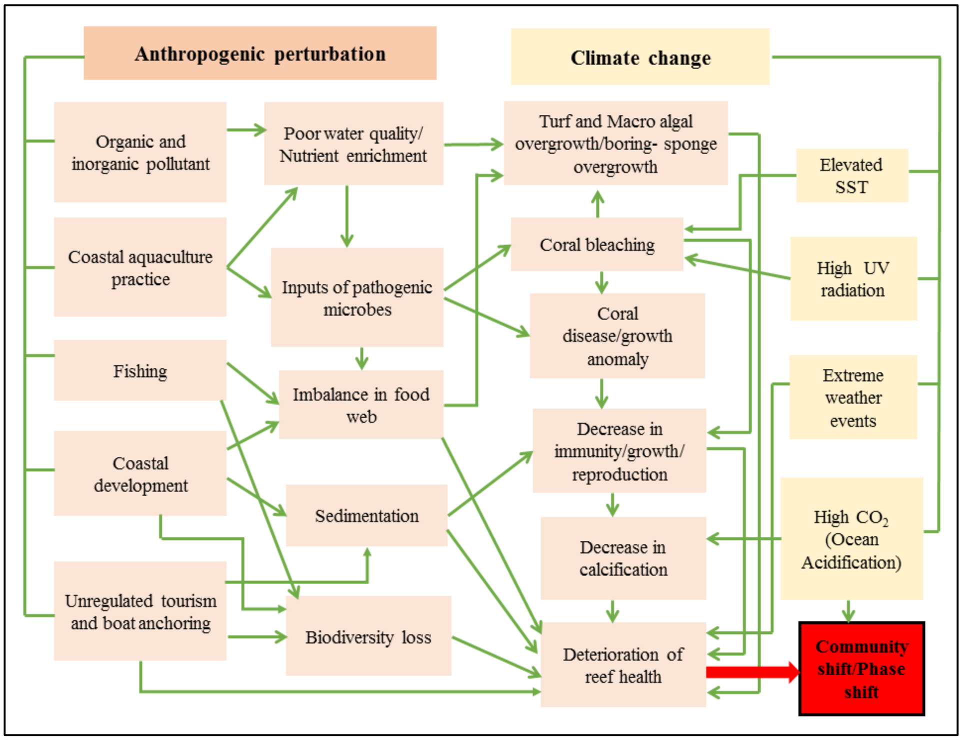

Fig. 6.1. Schematic presentation of the major impact of climatic and anthropogenic stressors on coral reef ecosystem and pathways to community shift or phase shift in the reef environment.

The term 'Coral-algal phase shifts' coined as the coral cover declines to their low levels and is replaced by algae in many reefs across worldwide. Studies have demonstrated that macroalgae would either benefit from or remain relatively unaffected by projected ocean warming (OW) and ocean acidification (OA), which is detrimental to the coral reefs (Hughes *et al.*, 2010). Evidence of coral-macroalgae regime shifts from field data has been supplemented by laboratory experimental investigation (e.g., Hughes *et al.*, 2007), theoretical (e.g., Knowlton, 1992; Mcmanus & Polsenberg, 2004) and through modelling (e.g., Mumby *et al.*, 2007). Studies demonstrated that nutrients enrichment in the coastal ocean from terrigenous sources promotes algal growth, and subsequent fishing of the key herbivores has resulted in the proliferation of macroalgae (Rasher *et al.*, 2012; Bozec *et al.*, 2016). Additionally, some macroalgae species like *Laurencia dendroidea*, *Fucus vesiculosus* can produce chemical defenses in response to herbivory, further resulted reduce grazing by herbivore (Jormalainen & Honkanen, 2008; Sudatti *et al.*, 2018).

Coral reefs in India are under tremendous stress due to bleaching events, disease outbreak, coastal pollution, bio-invasion, fishing, tourism and unregulated exploitation (De *et al.*, 2017; Raghuraman *et al.,* 2013; Rajan *et al.,* 2015; Venkataraman & Satyanarayana, 2003). In recent years macroalgae mediated coral health damage and the coral cover decline have been reported from the Gulf of Mannar and Palk Bay coral reefs (Machendiranathan *et al.*, 2016; Manikandan & Ravindran, 2017). However, most of the studies are based on short-term observation. Hence, the objectives of the present study were to provide insight on coral-algal interaction and to quantify spatio-temporal dynamic in the benthic cover of fleshy macroalgae and algal turf at four patch reefs in the Malvan Marine Sanctuary through long-term observation using consistent monitoring methodology.

6.2.Material and method

6.2.1. Study location and survey design

The present study was conducted in the Malvan Marine Sanctuary (MMS) (73º25'-73º28'.25N, 16º02'-16º02'.9E), a Marine Protected Area (MPA) located in the Central West coast of India known for its rich biodiversity (ICMAM 2001). Four patch reefs (ST#1-ST#4) were chosen for the survey (Fig. 8.1). Field surveys were conducted during pre-monsoon (PRM) and Post-monsoon (POM) seasons in October 2014 (POM14), Pre-monsoon May 2015 (PRM15), November 2015 (POM15), May 2016 (PRM16), October 2016 (POM16), May 2017 (PRM17), October 2018 (POM18), April 2019 (PRM19). Also, several sources of land-based pollution, including the discharge of untreated waste, and the sediment runoff (De *et al.*, 2015; Hussain *et al.*, 2016; Mote *et al.*, 2019). Additionally, two creeks (Kolam and Tarkarli) discharge terrigenous sediment within 1km on the north side and 4 km in the south of the MMS. Based on these observations, we employed a 20m underwater belt transect (English *et al.*, 1997) in triplicate at each site of the four permanent monitoring sites. The transect lines were deployed perpendicular to the shore and parallel to each other at a minimum distance of 10m. All the coral colonies and coral colonies in contact with fleshy macroalgae and algal turf falling within 1m of each side of the transect tape (20× (1+1) m) were counted and photographed. The benthic cover of the algae and coral within the belt transects was quantified by the line-intercept method (20m), following the central transect tape of the belt transects (English *et al.*, 1997). Underwater photographs of coral-sponge were taken using the Nikon AW130 camera. Site locations were recorded using Garmin handheld GPS, and the same coordinates were followed for the subsequent surveys. Corals were identified following Veron (2000), and macroalgae were identified following AlgaeBase (https://www.algaebase.org/). Algal turf was characterized by the presence of a dense consortium of different tiny and filamentous algae growing up to a height of about 1 cm (Wild *et al.*, 2014). The intensity of coral-algal contacts calculated by dividing the number of colonies with direct contacts with macroalgae and turf algae at each transect by the total number of colonies at that transect and expressed as a percentage. Site-level mean coral-algal contact intensity and standard errors were estimated from the belt transects. Three intensity values from replicate transects at each site further divided by the number of transects (n=3) to calculate a mean value for each sampling location. Data were presented as Mean±Standard Error (SE).

6.3. Result

Thirteen species of fleshy macroalgae were identified at the four surveyed reef sites during the present study. The macroalgal species were identified as *Amphiroa anceps, Amphiroa rigida, Chilosporum sagittatum, Dictyota cervicornis, Dictyota divericata, Caulerpa certulariodes, Caulerpa peltate, Sargassum cinereum, Sargassum crassifolium, Padina australis, Padina sp., Chnoospora minima, Dictyota faciola. Caulerpa* and *Sargassum* were the dominant genera.

6.3.1. Changes in macroalgal cover

During our first survey in the MPA in post-monsoon of 2014, fleshy macroalgal cover estimated to be 9.60%±0.49SE. In pre-monsoon of 2015, macroalgae cover increased 18.15%±1.17SE, and a slight increase was noted in post-monsoon of 2015, measured to be 16.88%±0.84SE. Further, in the pre-monsoon of 2016, macroalgae cover was measured to be 20.26%±0.93SE. In the post-monsoon of 2016, macroalgae cover expanded further and estimated to be 22.98%±0.88SE. During the pre-monsoon of 2017, macroalgal cover estimated to be 24.06%±0.89SE, which further increased to 24.54%±1.23SE in post-monsoon of 2018. During pre-monsoon of 2019, macroalgal cover elevated to 25.33%±1.01SE. From 2014 to 2019, a rapid relative increase of 163.96% in macroalgal cover was estimated in the MMS reefs.

6.3.2. Changes in algal turf cover

Benthic cover of algal turf showed a steady increasing trend in the MMS. In the post-monsoon of 2014, turf cover was 11.62%±0.58SE. Algal turf cover on the bottom substratum was estimated to be 11.65%±0.49SE in pre-monsoon of 2015. In post-monsoon of 2015, which increased to 13.45%±0.77SE. In pre-monsoon of 2016, algal turf spiked to 16.75%±0.75SE and 20.71%±0.88SE during the post-monsoon 2016. In pre-monsoon 2017 and post-monsoon of 2018, algal turf cover estimated to be 23.12%±0.88SE and 26.71%±0.97SE. During the pre-monsoon of 2019, algal turf cover increased to 27.45±1.10SE. During the study period, the relative increase in algal turf cover estimated to be 136.21% in the MMS.

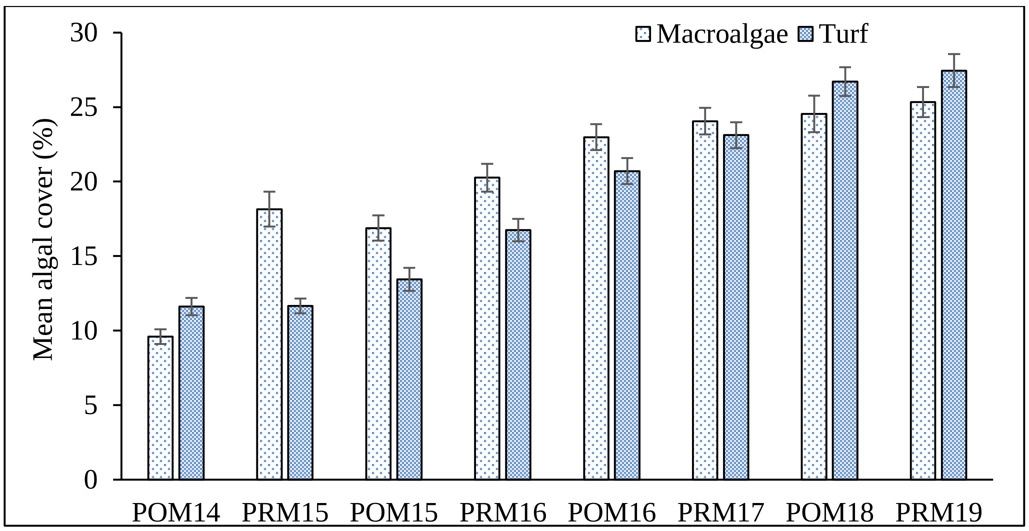

Fig. 6.2. Benthic algal cover (fleshy macroalgae and algal turf) in the MMS during different seasons (Mean±SE).

6.3.3. Total algal cover

In post-monsoon 2014, mean algal cover was recorded to be 21.21%±0.76SE, whereas, in pre-monsoon of 2015, the mean algal cover on the reef bottom was measured to be 29.80%±1.34SE. During the post-monsoon of 2015, the mean algae cover was estimated to be 30.33±1.06SE. In pre-monsoon of 2016, the mean algae cover was 37.01%±1.01SE, whereas in post-monsoon of 2016, algal cover estimated to be 43.69%±1.46SE. In pre-monsoon of 2017, algae cover increased to 47.17%±1.02SE. During post-monsoon 2018, the measured algal cover rose to 51.25%±1.31SE. While, in April 2019, algae cover spiked to 52.77%± 1.34SE with a broader spreading of macroalgae and turf algae over the recent dead coral colonies (Fig. 6.2-6.3). Relative increase in algal cover estimated to be 148.76% during the study period. A t-test of paired two samples for mean conducted to detect the correlation between the coral cover and algal cover (macroalgae and algal turf) in the reef, the Pearson correlation coefficient indicates an inverse correlation ($r = -0.94212$, $df=73$, $P>0.0005$) between the live coral cover and total algal cover, which confirms the coral cover has declined over the period while algal cover rose substantially in the reef (Fig. 6.4).

6.3.4. Coral algal contacts

In post-monsoon of 2014, 15.85%±1.04SE of coral colonies were in direct contact with macroalgae, whereas 10.52%±1.37SE of coral colonies were showed algal turf intrusion. In pre-monsoon of 2015, 15.51%±1.65SE of coral colonies were in direct contact with fleshy algae, and 14.47%±1.09SE of coral colonies were infested with algal turf. In the post-monsoon season of 2015, macroalgal contacts with coral colonies increased to 30.56%±1.97SE, and turf algal infestation increased to 26.70%±2.48SE. In pre-monsoon 2016, 28.58%±1.74SE of coral colonies were with macroalgal contacts, and 21.12%±2.38SE of colonies were invaded by tiny turf algae. In post-monsoon of 2016, 36.60%±2.44SE of the coral colonies were with direct contacts to macroalgae and 31.93%±1.32SE with turf-algae. In pre-monsoon of 2017, algal contacts further increased, and macroalgal and turf algal contacts with coral colonies were estimated to be 45.42%±3.72SE and 39.84%±3.09SE, respectively. During the post-monsoon of 2018, 51.08%±3.68SE of coral colonies showed direct contacts with macroalgae, and 49.37%±3.84SE of coral colonies were infested with algal turf. In pre-monsoon of 2019, macroalgal contacts with corals surged to 62.75%±3.96SE, and turf algal infestation on live coral colonies spiked to 52.75%±4.30SE.

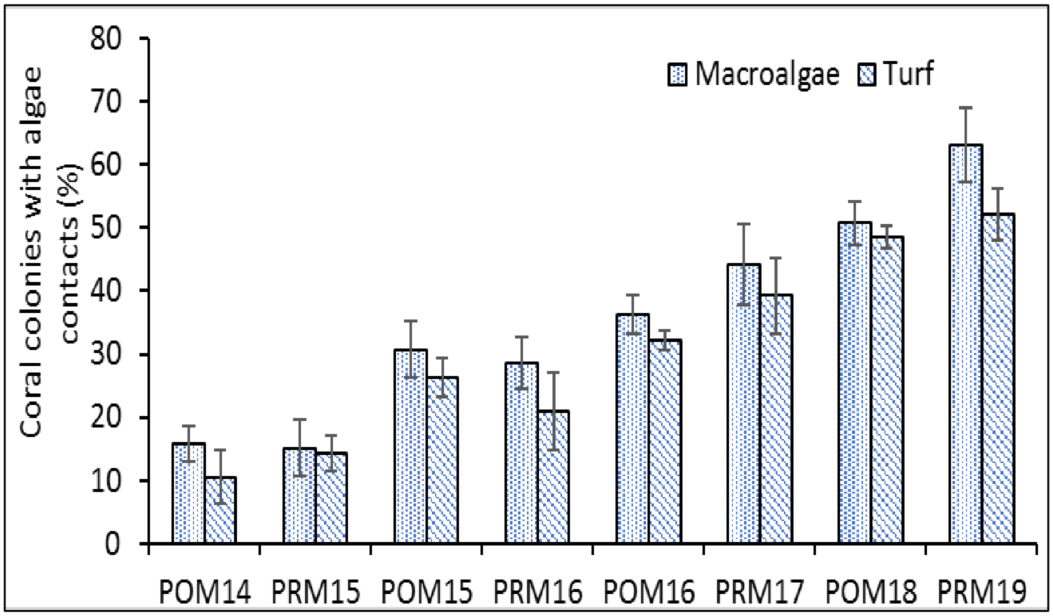

Fig. 6.3. Coral-algal contact intensity in the MMS during different seasons (Mean±SE).

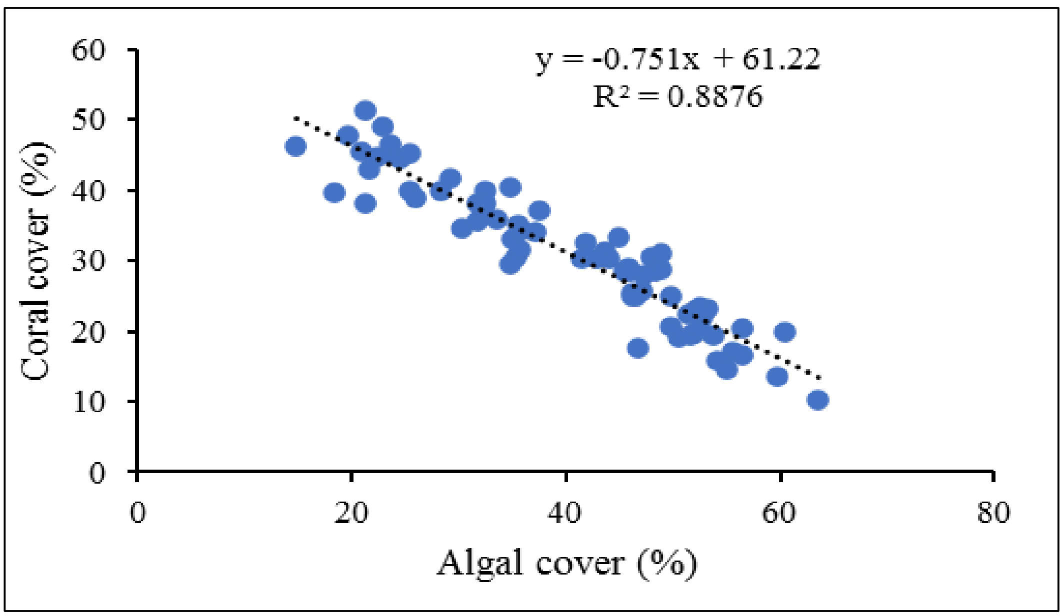

Fig. 6.4. Relationship between the mean live coral cover and benthic algal cover.

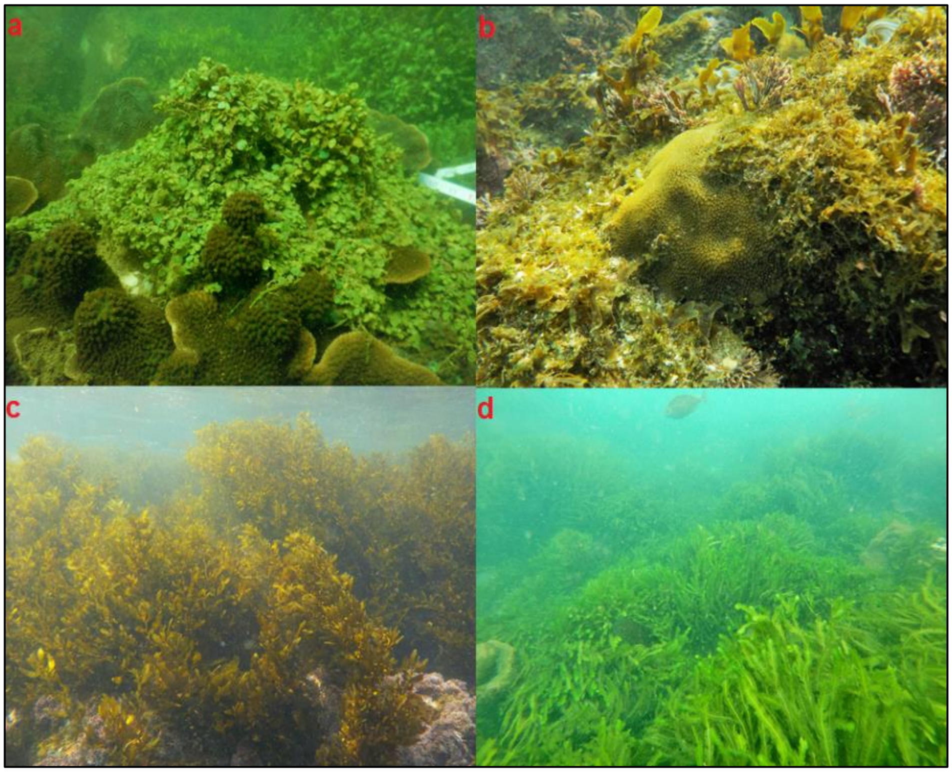

Fig. 6.5. An evidence of macro-algae proliferation in the MMS coral reefs. The magnitude of different macroalgal species growing over (a) *Turbinaria mesenterina;* (b) *Porites* sp.; (c) *Sargassum* and (d) *Caulerpa taxifolia* bed on the reef

6.4. Discussion

This study indicates increasing trends in benthic algal cover and intense coral-algal interaction. At both ends of the MMS (Fig. 8.1) two small rivers (Kolamb and Karli river) add sediment runoff, responsible for high suspended particulate matter (SPM), nutrients and turbidity into the coral reef habitat, thus increasing the organic matter load in the reef environment. Nutrient enrichment in the coral reef habitat has resulted in algal proliferation across the global reef (Rasher *et al.,* 2012; Jessen *et al.,* 2014; Gao *et al.*, 2017). On the other hand, artisanal fishing practice in the MMS has depleted the herbivore fish community, reduced herbivore biomass, also leading to algal growth in the reef environment (Rasher *et al.,* 2012; Welsh and Bellwood 2015). Studies showed that when protected from herbivores, approximately 40 to 70% of common macroalgae cause bleaching and death of coral tissue when in direct contact (Rasher & Hay, 2010). Further, macroalgae can cause immediate knock-off effect to corals by transfer of hydrophobic allelochemicals present on algal surfaces leading to coral bleaching, inhibiting photosynthesis, localized tissue death and occasionally death of corals (Rasher & Hay, 2010; Rasher *et al.,* 2011). Moreover, a recent study has demonstrated that increasing ocean acidification benefits some macroalgae over corals by enhancing the allelopathy (Del Monaco *et al.,* 2017). Additionally, benthic algae also host a variety of different potential coral pathogens and could transmit coral diseases like White Syndrome and Yellow Band Disease (Sweet *et al.,* 2013). These shifts from a coral to an algae-dominated state "phase-shifts," are likely to be accelerated and expected to occur more frequently (Fong & Paul, 2011). Moreover, this algal dominated alternative stable state is difficult to be reversed in the coral-dominated state (Briggs *et al.,* 2018; Schmitt *et al.,* 2019).

Furthermore, recent studies predicted that global environmental change is going to increase the frequency of extreme weather events like the mass coral bleaching events, tropical storms, and cyclones (Gutmann *et al.,* 2018). These natural calamities are widely known for mechanically disturbing the coral reefs (Hoegh-Guldberg *et al.,* 2018). Also, these events will cause a higher terrigenous nutrient influx in coastal reefs, which significantly supports the growth of algal biomass (Mejia *et al.,* 2012). This antagonistic impact over two communities will cause dramatic ecosystem changes (Hannah, 2015; Welsh & Bellwood, 2015). The reefs in the MMS have been witnessed severe heat stress during 2015-2017 due to El Niño Southern Oscillation (ENSO) event, which has resulted in intensive coral bleaching in these reefs. In October 2014, ~14% coral bleaching was noted in the MMS (De *et al.,* 2015), whereas, in December 2015, bleaching intensity increased to 71 % with a mortality of 8% (Raj *et al.*, 2018). The temperature anomaly causes bleaching in the significant coral colony while opening a niche for others to occupy (McManus and Polsenberg 2004; Bellwood *et al.,* 2006). This was an opportunistic chance for seasonally proliferating macroalgae, as they find vacant non-allelopathic substratum to establish. Seasonal seaweed proliferation brought the second wave of deleterious effects in post-monsoon of 2015/16, which covered sizable area predominantly by *Caulerpa* sp., *Sargassum* sp. (Fig.6.6). Algal phase shifts are known to have a significant negative impact on coral communities, which includes dominance of macroalgae, and turf algae (Bruno *et al.,* 2009; Fung *et al.,* 2011; Wild *et al.,* 2014). In a rapidly changing global environment, the consequences of increasing algal competition to coral may be severe, leading to elevated extinction risk and loss of critical reef habitat. However, further studies are required to elucidate the impact of increasing algal abundance on the coral resilience and recovery process.

6.4.1. Ecosystem services from macroalgae

In the marine ecosystem, seaweeds appear to be the most competitive and adaptive community confronting the climate change assaults. Seaweeds are the most extensive photoautotrophic vegetation of the coastal ecosystem and estimated to have an ocean cover of about 3.4 million Km^2 (Krause-Jensen *et al.,* 2018). Along with sizeable coastal distribution, the paraphyletic evolution of seaweeds represents their virtue of diversity compared to the homogenous diversity of other autotrophs like seagrass and mangrove. Although seaweed forests do not harbour the abundance and richness of biota that coral reefs do, but macroalgal canopy and understory offer three-dimensional living habitat and nursery grounds to plenty of associated communities, many of them with economic importance (e.g., Bartsch *et al.*, 2008; Christie *et al.*, 2009; Steneck *et al.*, 2002). They also serve as an ecosystem engineer and provide a range of ecosystem services (Bao *et al.,* 2019), such as habitat, food or nutrient regulation and biogeochemical cycling (Klinger, 2015) (Fig. 6.6-6.7).

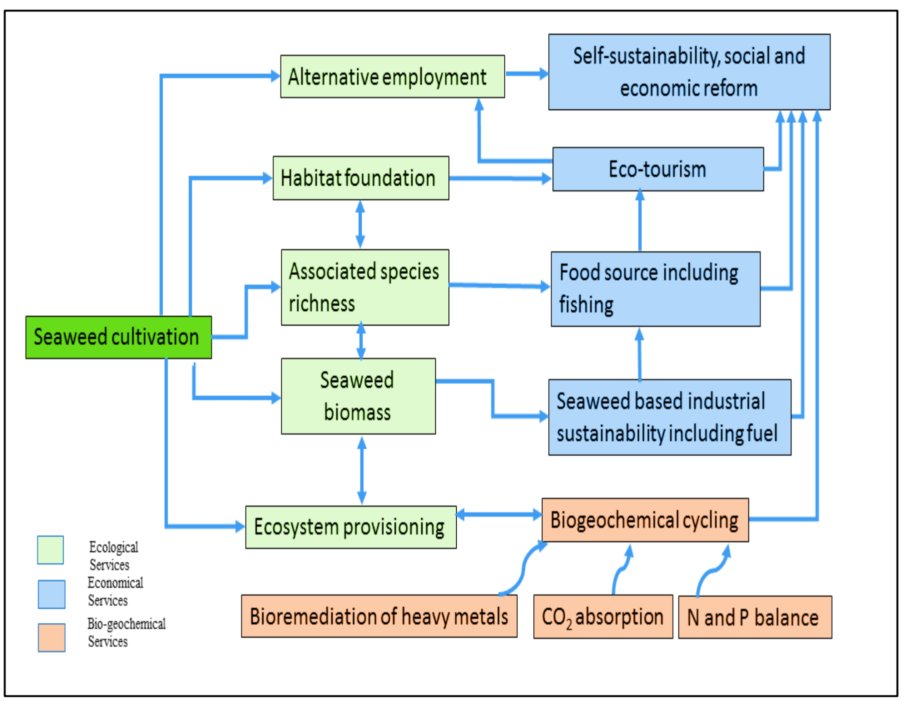

Fig. 6.6. Ecological, economic, and bio-geochemical services of the macroalgal ecosystem.

Habitat formation by macroalgae can boost the associated biodiversity and increase habitat complexity; therefore, it influences the functioning of local food webs (Irigoyen *et al.*, 2011; Salvaterra *et al.*, 2013). For example, macro-invertebrate densities often surpass 100000 numbers m^{-2} in macroalgal beds (Christie *et al.*, 2009), which point out a positive consequence on local biodiversity and food web. Simultaneously, high primary productivity of macroalgal ecosystems stimulates higher secondary production (Krumhansl & Scheibling, 2012), also attract fish aggregations by providing shelter and feeding ground for omnivores and invertebrate feeders (Anyango *et al.*, 2017; Eklöf *et al.*, 2006). Studies also noted that seaweeds are often preferred as food by herbivorous fishes like the Siganidae and Kyphosidae, a preferred plate fish, resulting in improved production (Tolentino-Pablico *et al.*, 2008; Hehre & Meeuwig, 2016), which could also benefit by reducing fishing pressure on coral reef fisheries. Additionally, habitat complexity of the macroalgal ecosystem is comparable with reefs, in terms of exchange and ecological interactions within nearby ecosystems (Lilley & Schiel, 2006). Hence, deterioration of the seaweed ecosystem could alter the community structure and decline in functional diversity and overall productivity.

Additionally, coastal macroalgal aquaculture contributes to shoreline protection and prevent coastal erosion by buffering wave energy (Mork, 1996; Løvås & Tørum, 2001). Studies have also shown that macroalgae are an excellent candidate for bioremediation of eutrophication as they remove excess inorganic nutrients and convert into biomass, thereby contributing to the improvement of coastal marine aquaculture environment and mitigate potentially adverse environmental impacts (Kim *et al.*, 2017; Zheng *et al.*, 2019). Furthermore, recent research has highlighted that macroalgae may form niches of high seawater pH, thereby, could be a potential medium for localizing remediation of ocean acidification (Krause-Jensen *et al.*, 2015) and additionally by contributing oxygen to the waters macroalgae could neutralize local de-oxygenation (Duarte *et al.*, 2017).

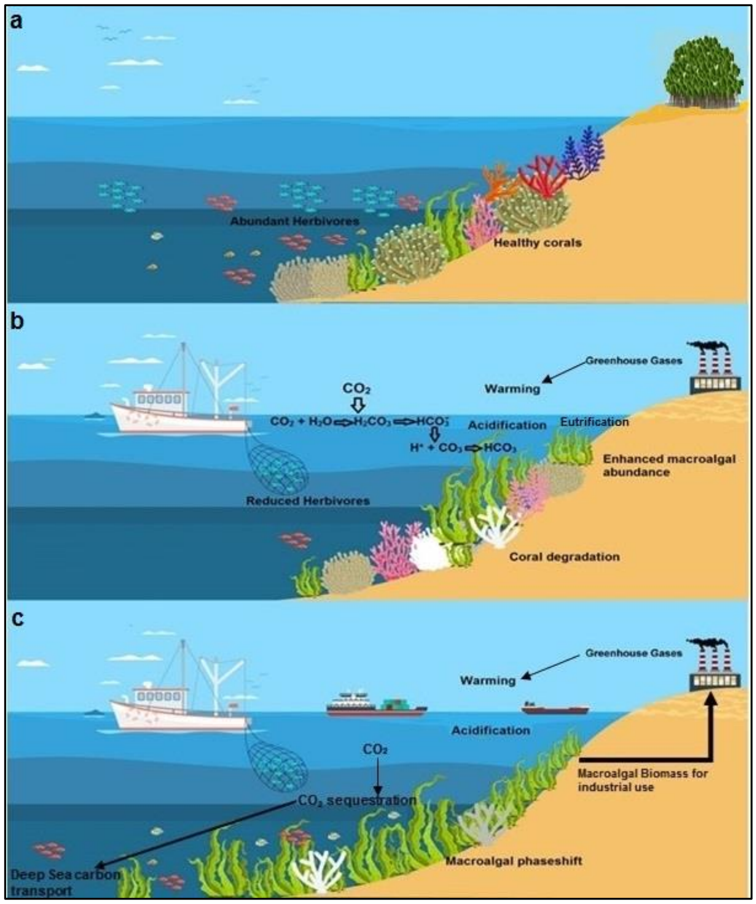

Fig. 6.7. Conceptual diagram on the gradual transformation of coral reef to seaweed dominated ecosystem. a) Healthy coral reef; b) Different stressors drove coral degradation and a gradual shift toward macroalgal dominance; c) Macroalgal phage shift in coral reef and ecological and economical services of macroalgal ecosystem

Chapter 7

7. Infestation of coral boring sponge

7.1. Introduction

Coral reefs are one of the most important ecosystems of the oceans. However, these ecologically sensitive habitats are diminishing worldwide at a faster pace due to various environmental stressors, including climate change and anthropogenic perturbation (Lough *et al.,* 2018). Coral reefs in India are already severely degraded and are under stress due to bleaching events, disease outbreak, coastal pollution, bio-invasion, fishing, tourism, and unregulated exploitation (Hussain *et al.,* 2016; De *et al.,* 2017; Nanajkar *et al.,* 2019).

Sponges are an essential component of coral reefs as they provide various ecosystem services such as the formation of 3D structures, water filtration, nutrient cycling, and living habitat to several organisms (Bell 2008). In tropical reefs, many sponge species compete with coral for space occupation (Schönberg *et al.*, 2017; Mote *et al.*, 2019). In recent years, several studies have reported competitive interactions between coral and sponge wherein increasing trend in sponge mediated bioerosion of coral skeleton is noted (Hernández-Ballesteros *et al.*, 2013; Bell *et al.*, 2013; Glynn and Manzello 2015). Sponges overgrow and out-compete corals, as corals are already in the cumbersome state due to multiple stressors like eutrophication, warming, and ocean acidification (Bell *et al.*, 2013; Schönberg *et al.*, 2017; Achlatis *et al.*, 2017).

In coral reef ecosystem, siliceous Clionaid sponges (Porifera, Demospongiae) are known to be the most aggressive space competitors and key bioeroder (Carballo *et al.,* 2013; Mote *et al.,* 2019). Among the Clionaid group, *Cliona* is the widespread and abundant genus of bio-eroding sponges (Schönberg *et al.,* 2017). These highly competitive excavating sponges degrade coral skeleton by their ability to penetrate and erode calcium carbonate ($CaCO_3$) framework and, eventually, kill the corals by overgrowing on the surface (Schönberg *et al.,* 2017; Mote *et al.,* 2019).

However, compared to other parts of the world, very little is known so far about the present ecological status of coral bio-eroding sponges on Indian reefs (Ashok *et al.,* 2018; Mote *et al.*, 2019). Therefore, the present study aims to understand the status of coral-killing bio-eroding Clionaid sponges, which is critical for the management and conservation of the important reef ecosystem.

7.2. Material and method

The present study was conducted in the Malvan Marine Sanctuary (MMS) (73^{o}25'-73^{o}28'.25N, 16^{o}02'-16^{o}02'.9E) in the Central West coast of India. Field surveys were conducted during pre-monsoon (PRM) and Post-monsoon (POM) seasons. However, the underwater observations were not possible during the monsoon months due to the rough sea condition and poor underwater visibility. Seasonal surveys were carried out in October 2014 (POM_2014), Pre-monsoon May 2015 (PRM_2015), November 2015 (POM_2015), May 2016 (PRM_2016), October 2016 (POM_2016), May 2017 (PRM_2017), October 2018 (POM_2018), April 2019 (PRM_2019). The MMS is the only Marine Protected Area (MPA) located in the Central West coast of India and known for its rich biodiversity (ICMAM 2001). Despite the protected status of the MMS, this coastal near-shore area is heavily impacted by the fishing activity and unregulated recreational SCUBA operations. Also, several sources of land-based pollution, including untreated waste, get discharged through the sediment runoff (De *et al.,* 2015; Hussain *et al.,* 2016; Mote *et al.,* 2019). Additionally, two creeks (Kolam and Tarkarli) discharge terrigenous sediment within 1km on the north side and 4 km in the south of the MMS (Fig.8.1). During our reef biodiversity monitoring program in the MMS (2016-2017), we observed the encrustation of dark brown sponge on the living coral colonies (Fig.7.3: A-F). Based on these observations, we employed a 20m underwater belt transect (English *et al.*, 1997) in triplicate at each site of the four permanent monitoring sites (Fig. 8.1). The transect lines were deployed perpendicular to the shore and parallel to each other at a minimum distance of 10m. All the coral colonies and the number of coral colonies invaded by boring sponge falling within 1m of each side of the transect tape ($20\times(1+1)$ m) were enumerated. The water depth at the study sites was ranged between 2-8 m and was characterised by patchy coral distribution, high turbidity, high wave action. Underwater photographs of coral-sponge were taken using the Nikon AW130 camera. Site locations were recorded using Garmin handheld GPS. The same coordinates were followed for the subsequent surveys. Corals were identified following Veron (2000) and sponge samples identified morphologically following Mote *et al.,* (2019).

7.2.1. Boring sponge infestation prevalence

Boring sponge infestation on corals was visually assessed by an external encrustation of sponge on the live coral colonies. Boring sponge infestation prevalence was estimated by dividing the number of sponges encrusted colonies at each transect by the over-all colonies at that transect and expressed as a percentage of the total number of sponges infested colonies. Site-wise mean sponge infestation prevalence and standard errors were estimated from the belt transects. Three prevalence values from replicate transects at each site further divided by the number of transects (n=3) to calculate a mean for each site. Data were presented as mean ±Standard Error (SE).

7.3. Result

In the MMS, coral reef formation is mostly dominated by massive and mounding corals such as *Porites* spp. *Favites* spp., *Plesiastrea* sp., *Cyphastrea* sp., *Goniastrea* sp. and foliose coral *Turbinaria* sp. (Fig.7.3). Notably, branching corals like Acroporids were missing from the study area. This is corroborated with the earlier findings by Qasim and Wafar (1979), suggested higher sedimentation and wave action in MMS suggested to inhibit the branching coral, whereas the stress-tolerant corals like *Porites, Goniastrea, Turbinaria* are adapted to such environment. *In-situ* observation showed that the brown encrusting sponge *Cliona thomasi* and *Cliona* spp. spread laterally on the surface of the colonies of *T. mesenterina*, *Favites* spp. and *Porites* spp. by forming a thin layer of 1-1.5 mm. Discoloration and dislodging of coral tissue was recorded in sponge infested coral colonies (Fig.7.3), indicating sponge mediated allelopathic interaction. Moreover, abnormal corallite structures were also observed on the infected *T. mesenterina*.

7.3.1. Annual trends in sponge infestation on corals

In POM 2014, the mean of sponge infestation on stony corals was estimated to be 3.20%±0.17SE of the total colonies surveyed. During PRM 2015, sponge infestation was recorded to be 3.66%±0.26SE, whereas, in POM 2015, the infection increases to 5.03%±0.50SE. In PRM 2016, sponge infestation prevalence did not change much and was estimated to be 5.00%±0.49SE. In POM 2016, sponge infestation increased to 5.24%±0.91SE.

Boring sponge infestation further increased to 6.39%±0.71SE in PRM 2017, 8.19%±0.56SE in POM 2018, and 11.20%±0.77SE during PRM 2019. A two-way ANOVA test revealed that boring sponge infestation prevalence shows the significant spatio-temporal difference, between reefs (df=3, MS=4.40, F=4.49, $p>0.005$) and between the seasons (df=7, MS=27.31, F=27.84, $p>0.05$).

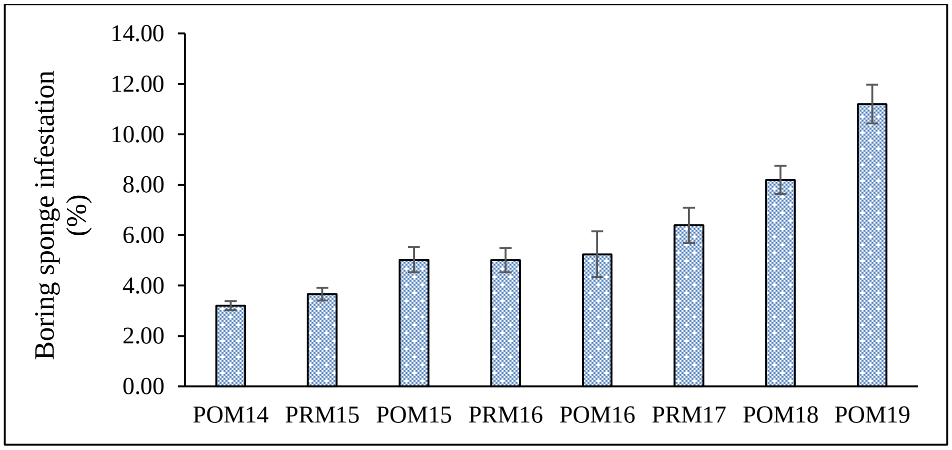

Fig. 7.1. Annual trends in boring sponge infestation in the MMS (Mean±SE).

It is worth noting that boring sponge species showed a habitat preference for laminar and plate form of *T. mesenterina* in MMS. A similar finding was also reported from the Gulf of Mannar, Southern part of India, where the Clionaid sponge *C. viridis* complex exhibited inclination for *T. mesenterina* (Ashok *et al.*, 2018). Likewise, encrustation on the *Porites* spp. and *Favites* spp. in the MMS corroborated with the findings in Southeast Florida, where Clionaid sponge *C. delitrix* showed a preference for massive coral species for settlement (Halperin *et al.*, 2016).

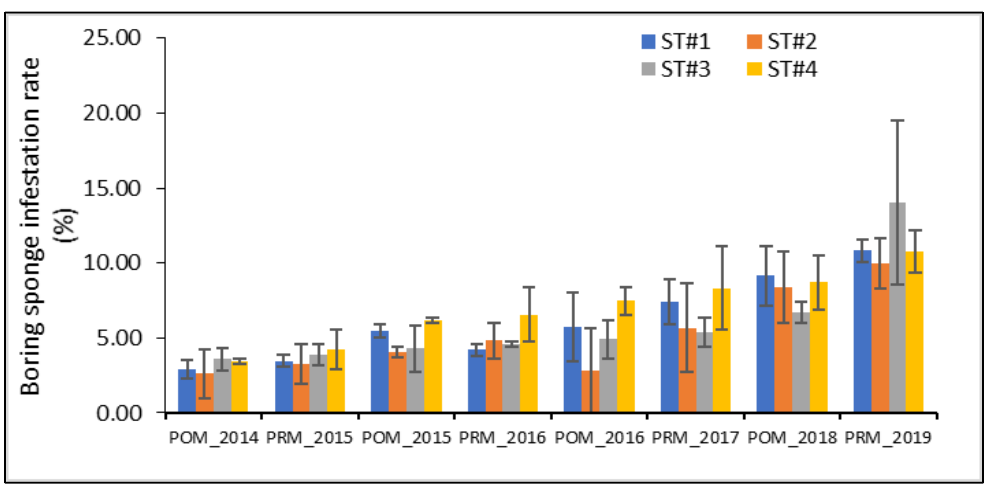

Fig. 7.2. Seasonal variation in sponge infestation in the MMS coral reef (Mean±SE)..

Results of the present study indicated about 249.84% relative increase in the prevalence of sponge encrustation on living coral over the study period from 2014 to 2019, which also coincided with the recent coral bleaching event and subsequent coral mortality. Thus, it indicates that *C. thomasi,* a dominant coral bio-eroding sponge in the MMN, may take advantage of coral bleaching and mortality (Bell *et al.,* 2013; Kelmo *et al.,* 2013).

Fig. 7.3. Coral boring sponges in the Malvan Marine Sanctuary; A) *Cliona* sp. growing on *Turbinaria mesenterina*; B) *Cliona* sp. growing on the bare rocks and coral rubble, C) *Cliona* sp. overgrowing on the massive *Porites* sp.; D) *Cliona* sp. growing on *Favites* sp.; E) *Cliona* sp. growing on *Turbinaria mesenterina;* F) *Cliona* sp. overgrowing on bleached *Porites* sp.

7.4. Discussion

Additionally, a significant decline in the live coral cover was noted from 45.09% in 2014 to 20.95% in 2019 in the study area. This decrease in coral abundance could have coincided with the bleaching induced coral mortality during the global bleaching event in 2015- 2016 (Raj *et al.,* 2018). During 2019 within four surveyed sites at MMS, ST#3 showed the highest sponge infestation rate (13.41%), followed by ST#1 (10.81%), ST#4 (10.74%) and ST#2 (9.84%, Fig. 7.2). The high abundance of the sponge at ST#1 and ST#4 may be attributed to the broad spatial coverage of hard bottom substrate (coral, Crustose Coralline Algae, rock) and low algal cover (macro- and turf-algae) compared to ST#2 (Hussain *et al.*, 2016). Similarly, the present findings provided a very similar result to that of *C. delitrix* in Florida, which showed high abundance in deeper outer reef areas (Halperin *et al.*, 2017) and the Great Barrier Reef. Ramsby *et al.*, (2017) reported that *C. orientalis* exhibited fastest-spreading in GBR when the macroalgal competition is low. It was observed that coral excavating/boring sponge abundance was higher in the deeper sites in response to higher availability of coral substratum and lower macroalgal coverage. Thus, the sponge could emerge as a superior competitor in the disturbed reef habitat, such as MMS. The resultant more coral destruction through bioerosion may ultimately lead to a community shift in this reef habitat.

These coral-killing Clinaid sponges are also known to hosts autotropic endosymbiont *Symbiodinium*, which provides additional nutritional support for sponge growth and bioerosion (Schönberg, 2006), which make them capable of overgrowing on live corals with the faster spreading rate (Schönberg *et al.*, 2017). Additionally, sponge zooxanthellae are more tolerant of thermal stress than corals (Vicente, 1990). Therefore, sponges enjoy a competitive advantage on bleached and weakened coral, further enhance sponge abundance accelerate coral degradation (Schönberg, 2000). Meanwhile, MMS witnessed ~14% coral bleaching during 2014 (De *et al.*, 2015) and 70.93% bleaching and 8.38% coral mortality during December 2015 (Raj *et al.,* 2018). Moreover, corals in MMS were subjected to severe stress due to other stressors like coral diseases (Hussain *et al.*, 2016) and increased pressure from the fishing and tourism (De *et al.,* 2017). The bleaching induced coral mortality, coral diseases, nutrients enrichment in reef support recruitment, and fast growth of coral-boring sponges (Carballo *et al.*, 2013; Chaves-Fonnegra *et al.,* 2018). Thus, the accelerated encrustation by *C. thomasi* and other boring sponges might be the repercussions of coral bleaching events and elevated stress to corals in the MMS.

In the MMS, dominant reef-forming coral genera comprised of massive coral *Porites* spp., *Favites* spp. and foliose coral *Turbinaria* sp. Thus, increasing sponge infestation on these coral species could lead to community shift and alter the structural properties of the reef (Loya *et al.,* 2001; Bell *et al.,* 2013). This study identified the rapid pace of coral invading sponge; therefore, further studies are required to understand the exact mechanisms of coral excavation and to develop a possible measure to restrict their rapid growth. Furthermore, local fisher populations depend on the MMS for their livelihood and recreational tourism like SCUBA diving, which is rapidly emerging in MMS because of easy accessibility (De *et al.,* 2015). Consequently, the degradation of the reef ecosystem could jeopardise the livelihood and local economy of this region (De *et al.,* 2015; Raj *et al.,* 2018). Additionally, considering increasing stress and earlier reports on the presence of Clionaid sponge in the Indian reefs *viz*; from Palk Bay (Thomas, 1972, 1979), Lakshadweep (Thomas, 1989), Nicobar Islands (Namboothri & Fernando, 2012) and, recent report on an outbreak of *Cliona viridis* complex species and *Clathria* (Microciona) *aceratoobtusa* from GoM (Ashok *et al.,* 2018; Ashok *et al.,* 2020) indicates the potential threat of Clionaid and other sponge invasions in different Indian reefs. Hence, it could be repercussive to coral community structure in these reefs. Therefore, the initiation of a long-term reef biomonitoring program is need of the hour.

Chapter 8

8. Climate change-induced coral bleaching at the Malvan Marine Sanctuary

8.1. Introduction

Coral reefs provide substantial ecological services, offer habitat for ~60000-9 million marine biotas, fish production, coastline protection, and generate billion-dollar economy through fishery and tourism (Costanza *et al.,* 2014; Reaka-Kudla, 1997; Plaisance *et al.,* 2011). Climate change coupled with chronic anthropogenic disturbances includes overfishing and herbivore loss, eutrophication, sedimentation have exacerbated coral reefs degradation over the past decades (Pörtner *et al.,* 2005; Burke *et al.,* 2011; Hoegh-Guldberg *et al.,* 2017; Weijerman *et al.,* 2018), deterioration of coral reef health globally become a severe conservation concern for the persistence of the world's coral reefs (Bellwood *et al.,* 2004; Braiard *et al.,* 2013; Birkeland, 2015). This damage reached a level that was described as a "coral reef crisis" (Bellwood *et al.,* 2004; Madin & Madin, 2015), with 75% of global coral reefs threatened in some way as of 2010 (Burke *et al.,* 2011; Hughes *et al.,* 2017).

The tropical coral reefs are formed where ocean temperature ranges from 18°C in winter to 28°C in summer months (Frieler *et al.,* 2013). Reef-building corals bleach when warmer than usual sea temperatures disrupt their mutualistic relationship with the algal symbionts, called zooxanthellae, that reside within their tissues, (*Symbiodinium* spp.), whose photosynthesis provides corals with up to 90% of their energy (Stanley, 2006). However, increasingly frequent thermal disturbance associated with climate change has induced severe coral bleaching globally, which has emerged as the greatest threats to the persistence of coral reefs (Hughes *et al.*, 2017, 2018; Eakin *et al.,* 2019; Sully *et al.,* 2019). Coral reefs across the world's oceans have witnessed the longest bleaching event (back to back since mid-2014 to 2017) triggered by unprecedented and extended ocean heatwave, exacerbated by one of the strongest El Niño events on record (Heron *et al.,* 2016b; Eakin *et al.,* 2019; Skirving *et al.,* 2019). Before 2015, the *El Niño*-Southern Oscillation (ENSO) events observed to impact coral reefs globally occurred in 1982–83 and 1997–98 (Heron *et al.,* 2016b; Lough *et al.,* 2018). Whereas the 2015–2016 *El Niño* emerged as the most extreme both in terms of ocean warming intensity and extent (Eakin *et al.,* 2016, 2019; Van Hooidonk *et al.,* 2016), causing unparalleled ecological, and economic consequences worldwide. The period between 2014-2016 was the record-breaking warmest year in history (NOAA, 2016). In fact, 2014 emerged as the warmest year in terms of global surface temperature. The year 2015 was 0.16 °C warmer than 2014, setting not only the record for the warmest year ever but also the record for the most substantial single-year increase (NOAA, 2016). The year 2016 was characterized by exceptional warming, and global temperatures were the highest on record, and mass coral bleaching happened globally. The back-to-back warming events left no window of recovery to corals from the previous bleaching events to the following year, leading to mass mortality (Eakin *et al.,* 2019; Skirving *et al.,* 2019), as coral mortality following a bleaching event depends on the extent of the heat stress and severity and duration of bleaching (Brown, 1997; Anthony *et al.,* 2008).

Coral bleaching events have long-term ecological, economic, and social impacts (Brown 1997). Mass coral bleaching and mortality events are driven by extreme *El Niño/La Niña* events that can jeopardize the structural and functional set up of the coral reef ecosystem (Graham *et al.,* 2015). Although corals can re-establish themselves after major bleaching events, sometimes it could take decades to return the coral ecosystem to the pre-bleaching state (Baker *et al.,* 2008). However, in recent times, increasing trends in the intensity and frequency of bleaching events could upset the capacity of corals to recover between the recurrent events. Coral bleaching can significantly affect the fishery population, drastically cut the tourism revenue as well as leads to changes in benthic habitat (Pratchett *et al.,* 2011; Munday *et al.,* 2008; Pratchett *et al.,* 2008; Wilson *et al.,* 2006). The year 1982/83 and 1997/98 had extreme *El Niño*-driven anomalous conditions caused widespread environmental disruptions, including the disappearance of marine life and decimation of the native bird population in the Galapagos Islands (Valle *et al.,* 1987) and severe bleaching of corals in the Pacific and beyond (Aronson *et al.,* 2002). The impacts extended to every continent, and the event of 1997-/98 alone caused US$35–45 billion in damage and claimed an estimated 23,000 human lives world-wide (Sponberg, 1999).

The magnitude of the *El Niño* Southern Oscillation (ENSO) event is implicated as the primary cause of coral mortality in reef ecosystems in the Indian Ocean (Wilkinson, 2009). Most of the

Indian coral reefs experienced mass bleaching during the *El Niño* years, viz. 1998, 2010 and 2016 (Arora *et al.*, 2019a; Arthur, 2002; De *et al.*, 2017; Krishnan *et al.*, 2018, 2011; Raghuraman *et al.*, 2013; Sarkar & Ghosh, 2013; Venkataraman *et al.*, 2003). The year 2010 recorded the highest thermal stress for Andaman, Nicobar, and Gulf of Kachchh regions, and the year 2016 was severe for Lakshadweep and Gulf of Mannar regions (Arora *et al.*, 2019a, b). However, there is still a knowledge gap on the long-term impact of coral bleaching event and coral recovery status in the Indian coral reefs (De *et al.*, 2017). Therefore, there is an urgent need for long-term monitoring (before-during-after bleaching) for a better understanding of the bleaching impacts for improved reef management practices. Thus, this study aimed to gain a synoptic view of the thermal-stress driven coral bleaching prevalence in the Malvan Marine Sanctuary (MMS) and provide insights on the impact of the bleaching events on the reef environment.

8.2. Materials & Methods

8.2.1. Study area

The Malvan Marine Sanctuary (MMS), located in the Central West coast of India along the Eastern Arabian Sea, spreads over 29.122 km^2 area. The sanctuary harbor several nearshore patch coral reefs mostly dominated by massive and encrusting *Porites* species and foliose and plate forming *Turbinaria mesenterina.* Other coral species include *Porites lichen, P. lutea, P. compressa, Pseudosiderastrea tayami, Siderastrea savignyana, Coscinaraea monile, Favites melicerum, Favites halicora, Cyphastrea serailia, Plesiastrea versipora, Goniopora stokesii* and *Tubastraea coccinea* (ICMAM 2001; De *et al.*, 2015). Although the MMS designated as a Marine Protected Area (MPA), the absence of robust management system and opposition of the MPA from the local people resulted in severe local perturbation includes fishing, waste-water drainage, and unchecked recreational activities along with climate change disturbance (Rajagopalan, 2008; De *et al.*, 2015).

8.2.2. Underwater survey design and method

Random visual surveys were conducted in four reef sites (ST#1, ST#2, ST#3, ST#4; Fig.8.1) in the MMS from October 2014, May 2015, May 2016, November 2016, May 2017, October 2018 to April 2019 with the aid of SCUBA to monitor coral health status and bleaching episode. Following this, a more intensive *in situ* examination was conducted to estimate the severity of coral bleaching. At each site, belt transects (20 m X 2m) were laid in triplicate (English *et al.*, 1997), separated by a distance of ~10m, along the depth contour of 2-10 m in the reef flat. All the coral colonies falling within the belt transect were identified up to the genus level, enumerated, photographed, and checked for bleaching sign underwater. The total numbers of unbleached, partially bleached, and fully bleached colonies were counted to estimate the bleaching prevalence. Additionally, the areal cover of the dominant benthic component (live coral cover, macro-algae, and algal turf cover) within the belt transects were quantified by the line-intercept method (20m), following the central transect tape of the belt transects (English *et al.*, 1997). Diving locations and the transect points were mapped with Garmin GPSMAP 78S, and the same coordinates were followed for consecutive surveys. The i*n-situ* temperature was noted using a YSI multiparameter probe from the study sites.

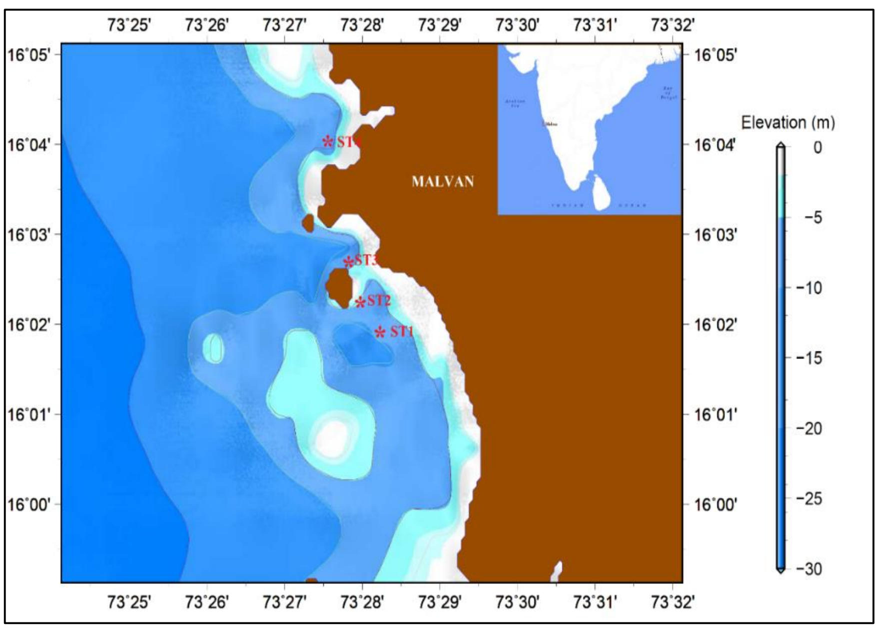

Fig. 8.1. Locations of patch coral reefs within the Malvan Marine Sanctuary, India surveyed during the present study

8.2.3. Bleaching prevalence

Bleaching prevalence was visually assessed by the external appearance of corals (colonies exhibited a pale colour or whitening of the colonies) during the underwater surveys. Bleaching prevalence was estimated by dividing the numbers of bleached colonies at each transect by the over-all colonies at that transect and expressed as a percentage of the total number of bleached colonies. Site-level mean bleaching prevalence or intensity and standard errors were estimated from the belt transects. Three prevalence values from replicate transects at each site further divided by the number of transects (n=3) to calculate a mean for each site.

8.2.4. Analysis of Sea Surface Temperature (SST) and calculation of bleaching indices

To understanding how the ENSO events related to heat stress affects coral reefs at a global scale requires quantification of the magnitude, intensity, and duration of thermal stress for each reef location. Analysis of daily SST trends and daily SST anomaly from January 2014 to May 2019 for the MPA was carried out using the National Oceanic and Atmospheric Administration (NOAA) Coral Reef Watch's (CRW) near-real-time daily global 5 km (0.05 x 0.05^{o} C exactly) satellite sea surface temperature (SST) monitoring high-resolution time-series data product known as 'CoralTemp' version 3.1 (available from NOAA Coral Reef Watch 2019 https://coralreefwatch.noaa.gov).

Bleaching Threshold (BT), Positive SST Anomaly (PA), and Degree Heating Weeks (DHW) are commonly used indices for calculating thermal stress on coral reefs (Liu *et al.,* 2014; Heron *et al.,* 2016a). Coral bleaching Threshold (BT) is based on the concept of Thermal Threshold (also known as long-term climatological mean). Thermal Threshold indicates the upper limit of SST, which corals can tolerate. If the SSTs exceed this threshold value, corals become stressed. The Thermal Threshold for the Malvan region was computed using the mean of warmest month SST from 1982 to 2019 (38 years period). The difference between the elevated SST to Thermal Threshold is known as Positive Anomaly. Positive SST anomaly depicts the extent of thermal stress conducive to coral bleaching (Liu *et al.,* 2003). Positive SST Anomaly was computed for the six years (from 2014 to 2019).

DHW provides information on the intensity and duration of thermal stress on coral reefs (Strong *et al.,* 1997). DHW can be calculated by the sum of Positive SST Anomaly (PA) at that location over the 12-week time period and is expressed in the unit °C-weeks (Venegas *et al.,* 2019). The HotSpot technique forms the basis of the "Degree Heating Week (DHW) " product, which is a cumulative measure of thermal stress over an area over the past three months (Strong *et al.,* 2006). HotSpots are positive anomalies, which derived by subtracting the thermal threshold from SST values (Liu *et al.,* 2014). One DHW is equivalent to one week of HotSpot staying at 1° C or 0.5 weeks of HotSpot at 2°C. This DHW product indicates the risk of bleaching of coral reefs around the world. Generally, the onset of coral bleaching has been observed when corals exposed to the DHW value of 0.5 or more (Done *et al.,* 2003). Arora *et al.,* (2019b) describe the five categories of DHW, which are used to describe the severity of bleaching, viz. no stress, bleach watch, warning, alert level-1, and alert level-2. No stress: 0°C < DHW < 2°C; Bleach watch: 2°C < DHW < 4°C; Warning: 4°C < DHW < 6°C; Alert level-1: 6°C < DHW < 8°C; Alert level-2: DHW > 8°C.

8.3. Results

8.3.1. Coral exposure to thermal stress

The SST and SST anomaly trends were derived from NOAA Coral Reef Watch during the period from Jan 2014 to May 2019. The SST patterns for the six periods provide information on magnitude, intensity, and duration of thermal stress over MMS (Figure 2). The climatologically warmest month for the Malvan region was May and recorded a maximum thermal threshold of 29.39° C (± 0.49). This study shows that the years 2015 and 2016 were the warmest among the analysis period from the year 2014 to 2019. The year 2015 recorded the highest warmest SST, warmest HotSpot, and continuous duration of thermal stress while the year 2016 was recorded the highest total duration of thermal stress and DHW (Table 1). The SST and HotSpot were recorded at 30.85° C and 1.46° C in the year 2015, which was 0.06° C higher than the year 2016. The highest value of DHW recorded in the year 2016 was 6.92° C and found 'alert level-1' warning status according to the coral bleaching alert protocol developed at Space Applications Centre, ISRO.

Table 8.1. Coral bleaching thermal stress indices during the period from the year 2014 to 2019 over the Malvan region.

Year	Warmest SST (◦C)	Warmest HotSpot (◦C)	Duration of Thermal Stress		Degree Heating Weeks (◦C)
			Continuous	Discrete	
2014	30.60	1.21	34	69	4.80
2015	30.85	1.46	58	58	5.09

2016	30.79	1.40	47	82	6.92
2017	30.37	0.98	21	51	4.00
2018	30.53	1.14	31	59	3.23
2019	29.84	0.45	06	13	0.25

The 2015-16 ENSO event had resulted in elevated the SST across the tropical oceans and unleashed unprecedented thermal stress on corals, led in mass coral bleaching caused by the (Arora *et al.*, 2019a). The first mass coral bleaching was experienced in 1997-98, and the second was experienced in 2010. This analysis also showed that October in 2014 and 2017, October-November month in the year 2015 and 2018 had experienced high thermal stress, which also crossed the thermal threshold and bleaching threshold value. The year 2019 has shown the least thermal stress indices due to the cyclone ‘Vayu’ in the Arabian Sea, which led in heavy to extreme rainfall and subsequently drawdown the SST. The satellite-derived SST and *in-situ* data indicated that corals at MMS experienced significant heat stress during a prolonged period from 2014 to 2019, which caused back-to-back coral bleaching events.

During our *in situ* survey period, observation of SST data derived from NOAA Coral Reef Watch showed 1.38^{o}C monthly anomaly in October 2014, whereas monthly mean SST was calculated to be 29.16^{o}C. In October 2015, monthly mean SST was 29.56^{o}C with an anomaly of 1.82^{o}C, positive SST anomaly persisted until November 2015, measured up to 1.59^{O}C, while monthly mean SST estimated was 29.62^{O}C. Average monthly SST rose to 30.11°C in May 2016, and all the days in May showed positive SST anomaly with a monthly mean positive anomaly of 0.90°C. During May 2017, the mean monthly SST was hit to 29.70°C, with a mean anomaly of 0.48°C. The mean monthly SST was recorded to be 29.21°C in October 2018, with continuous positive thermal stress, and the mean monthly SST anomaly calculated was 1.43°C. Further, in April 2019, the mean SST was recorded at 29.25°C, with an anomaly of -0.26°C. The recorded satellite derive SST data indicated that corals at MMS experienced significant heat stress during a prolonged period in 2014-2019, which lead to back-to-back coral bleaching events (Fig. 8.2-8.5).

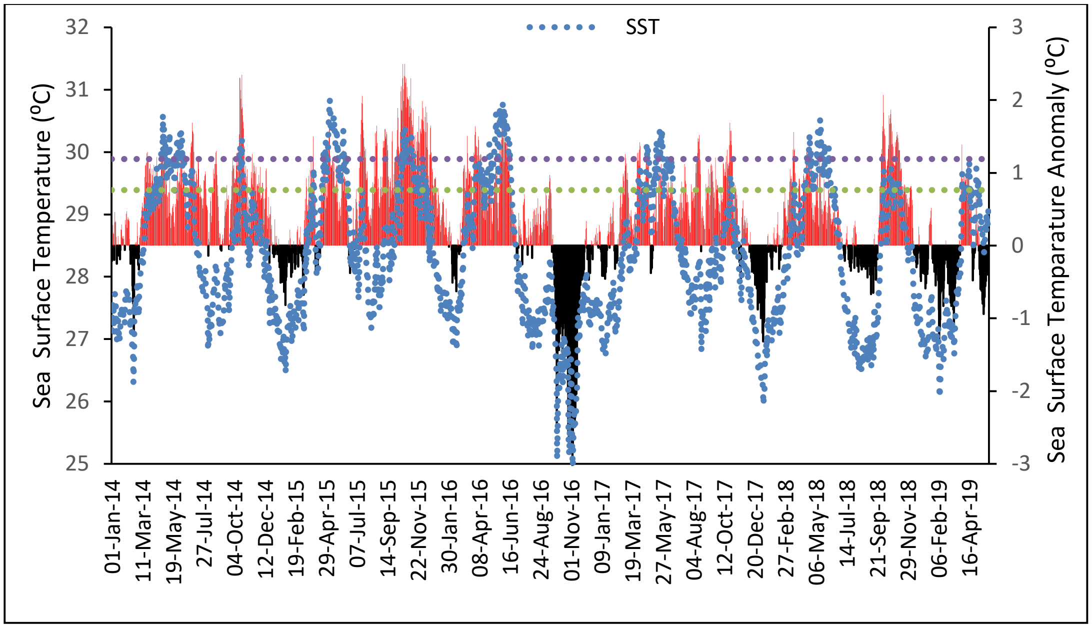

Fig. 8.2. Daily trends of mean SST and SST anomaly during the period from January 2014 to May 2019 (data derived from NOAA CRW v 3.1 datasets), the lower line showing the thermal threshold and above line showing the bleaching threshold.

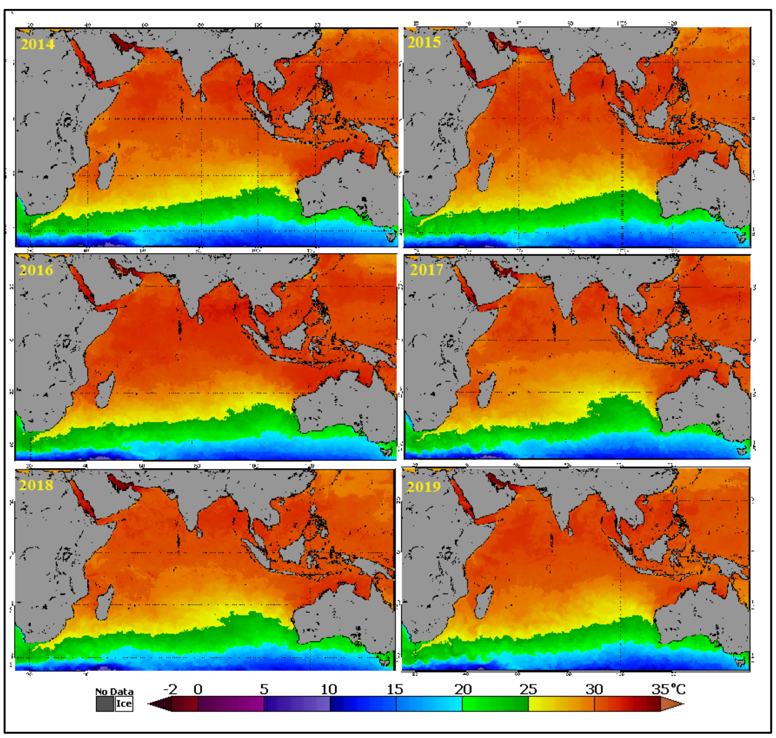

Fig. 8.3. Mean Sea Surface Temperature (SST) during 2014 to 2019. (Plots were drawn using NOAA-Coral Reef Watch web platform, http://coralreefwatch.noaa.gov/)

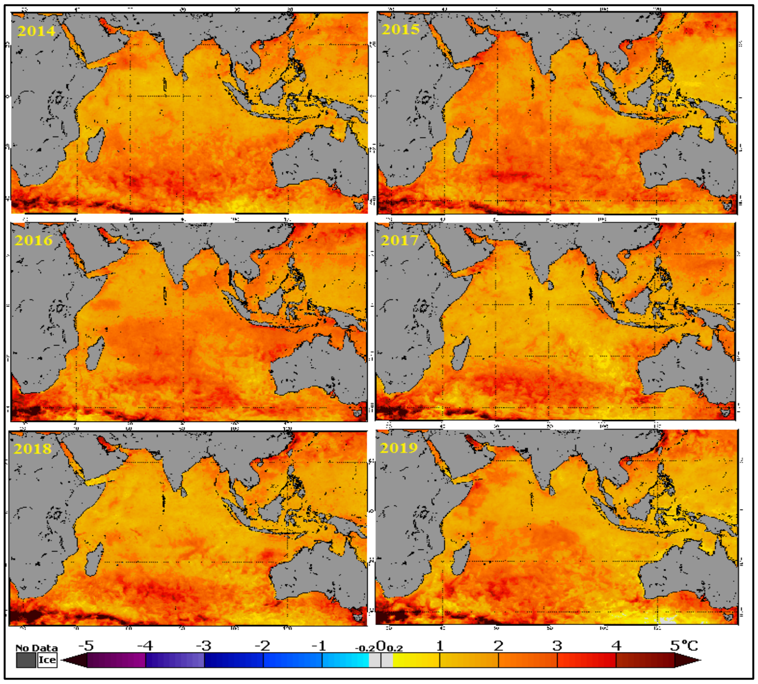

Fig. 8.4. Mean Sea Surface Temperature (SST) anomaly during 2014 to 2019. (Plots were drawn using NOAA-Coral Reef Watch web platform, http://coralreefwatch.noaa.gov/)

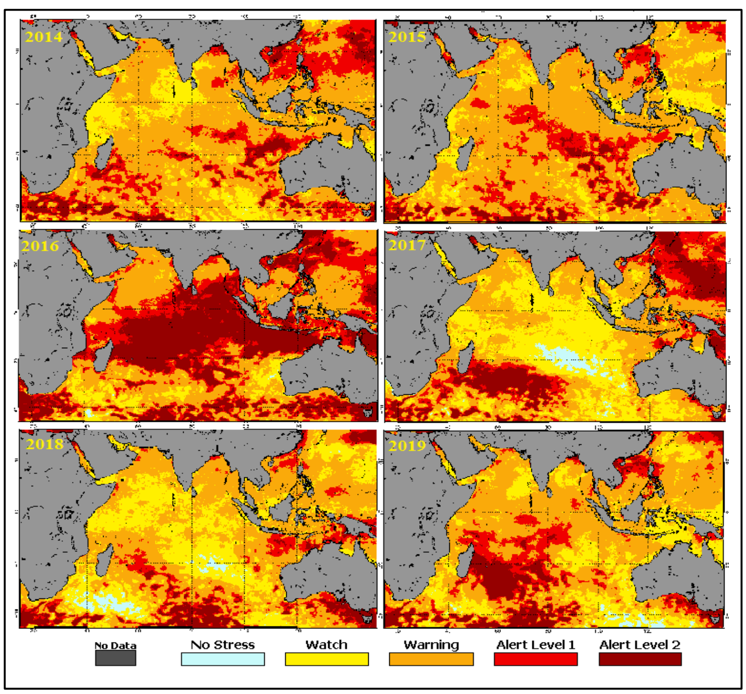

Fig. 8.5. NOAA-Coral Reef Watch coral bleaching alert during 2014 to 2019 (Plots were drawn using NOAA-Coral Reef Watch web platform, http://coralreefwatch.noaa.gov/)

8.3.2. Spatio-temporal variation in bleaching prevalence

Recurrent coral bleaching events and subsequent coral mortality were recorded in the MMS from 2014 to 2019. During the coral bleaching survey in October 2014, the estimated mean bleaching prevalence (MBP) was 14.58%±1.75SE. Underwater survey reveals partial and whole colony bleaching of *Porites lichen*, *Porites compressa*, *Favites melicerum*, *Turbinaria mesenterina* (Fig. 8.8), *Pseudosiderastrea tayami, Cyphastrea serailia, Plesiastrea versipora, Goniopora* spp, *Siderastrea savignyana.* MBP rose to 54.20%±2.58SE in May 2015, and a subsequent survey by Raj *et al.*, (2018) found 70.93% bleaching prevalence in December 2015. Our survey in May 2016 recorded a significant MBP of 46.76%±2.26SE and as well as 20.22%±0.73SE during May 2017. MBP had come down to 5.07%±0.61SE in October 2018, whereas the next survey in April 2019 revealed MBP of 8.37%±1.09SE.

Significant annual variation was observed in mean bleaching prevalence across the study sites (One-way Anova $p>0.001$). However, MBP did not show any significant differences at site depth (One-way Anova, $p=0.72$). During our first survey in 2014, MBP at ST#1 was estimated to be 12.95%±2.39SE. At ST#2, 18.24%±3.14SE coral colonies were found bleached. At ST#3 and ST#4, MBP was calculated to be 16.61%±2.07SE and 10.53%±1.64SE, respectively. In May 2015, the MBP of corals increased in all the reefs. The highest bleaching was estimated at 58.81%±4.41SE at ST#3, followed by 58.50%±5.02SE at ST#2. At ST#1, MBP was estimated to be 49.98%±4.88SE, and at ST#4, 49.51%±5.74SE bleaching was recorded. Furthermore, high MBP persisted during May 2016. Again, the highest MBP was recorded at the ST#2 (52.07%±4.87SE) and ST#3 (48.11%±3.43SE), respectively. Whereas at ST#1, MBP was measured to be 45.56%±2.97SE and was 41.31%±3.67SE at ST#4. Although, in May 2017, MBP had declined compared to the previous year (May 2016). However, significant bleaching was observed at all the sites. Highest MBP was recorded at the ST#1 (21.96%±5.59SE), followed by 20.46%±6.15SE at ST#2, 20.05%±1.01SE and 18.41%±1.93SE at the ST#4. In 2018 bleaching was reduced significantly. Estimated MBP was 6.34%±0.09SE at ST#2, followed by 5.88%±1.90SE at ST#3, 4.05%±1.08SE at ST#4 and 3.99%±0.21SE at ST#1. During 2019, a slight increase in MBP was recorded. Highest MBP was estimated at ST#3 (10.14%±2.08SE), followed by 9.88%±3.91SE at ST#2, 8.06%±0.14SE and 5.41%±0.64SE at ST#1 (Fig.8.6).

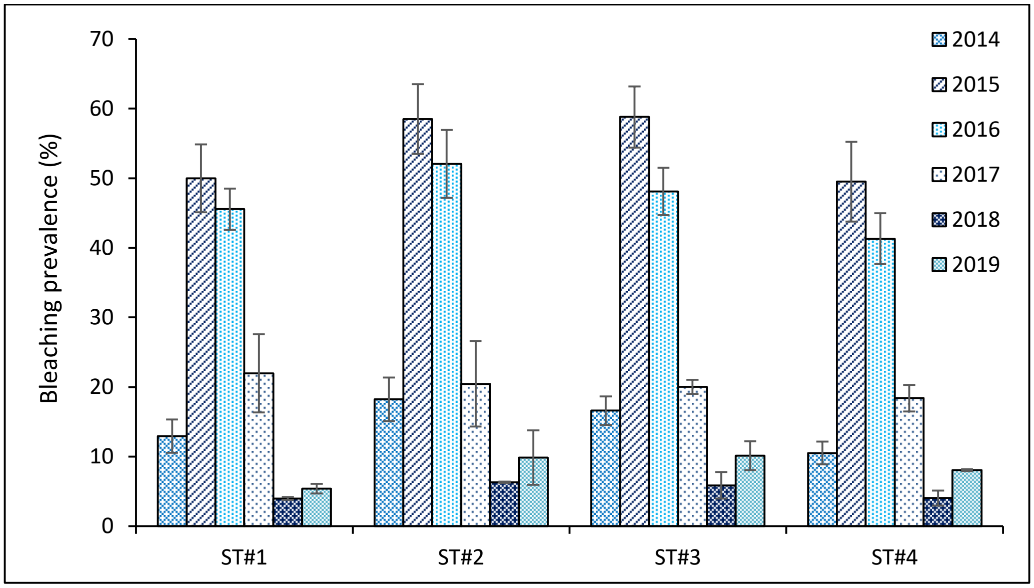

Fig. 8.6. Annual coral bleaching prevalence (Mean±SE) at different reefs from 2014 to 2019.

8.3.3. Decline in coral abundance and changes in live coral cover

The transect study revealed that the live coral cover declined significantly after each bleaching event (Fig. 8.7). Mean abundance of coral colonies reduced from 2.54 colonies/m^2 in 2014 to 1.04 colonies/m^{2}in 2019, which indicates a 59.06% decline in colony abundance. Furthermore, the mean live coral cover declined from 45.09% in 2014 to 19.72% in 2019, indicates dramatic coral loss due to the recurrent mass bleaching events.

Although the survey method and replicate were consistent throughout the survey period, however, a gradual decline in the live coral colonies was noticed in the reefs. During 2014, a total of 1210 coral colonies accounted for this study. In 2015, live coral colony numbers lowered to 1149. Further, 1010 live coral colonies were accounted at all the surveyed reefs in 2016. The number of live colonies was sharply downed to 626 in 2017. Coral colonies further declined to 551 and 501 in 2018 and 2019, respectively. The decrease in the live coral colonies was attributed to significant mortality events due to coral bleaching during 2014-2016 (Raj *et al.*, 2018), coral breakage by tourist activities, disease outbreak, and macro and turf algal overgrowth on coral (De *et al.*, 2020).

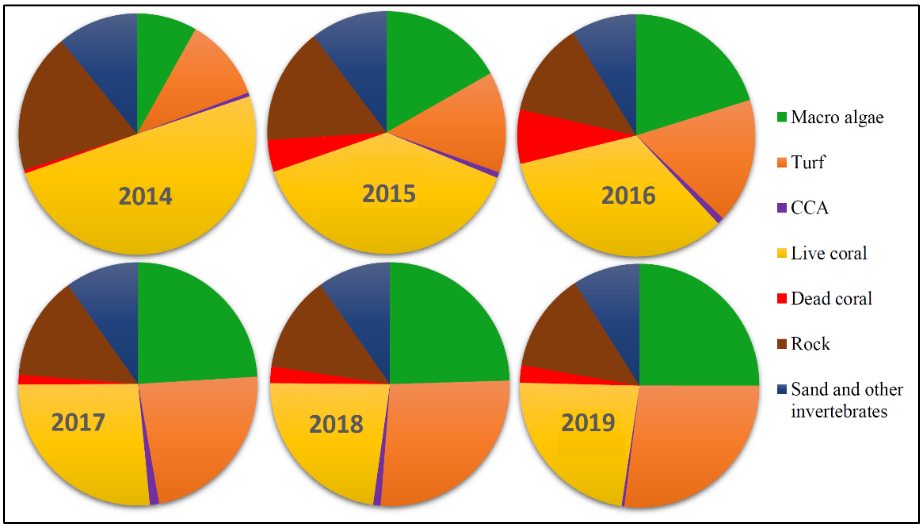

Fig. 8.7. Mean changes in prominent benthic category cover (%) in the MMS

8.4. Discussion

El Niño Southern Oscillation (ENSO) events emerged in tropical oceans during 1982–83, 1997–98, 2010, and 2015–16 (Eakin *et al.*, 2016; Muñiz-Castillo *et al.*, 2019). Ocean scale warming, driven by the ENSO event has caused the most prolonged and devastating global coral bleaching event on the history (Eakin *et al.*, 2019; Skirving *et al.*, 2019). Since mid-2014, observations of coral bleaching reports surfaced across all the tropical ocean (Eakin *et al.*, 2016; Heron *et al.*, 2016b). Eventually, it is realized that coral bleaching in 2014–2017 was the most severe, widespread, and deadly bleaching event globally (Vargas-Ángel *et al.*, 2019). Largescale coral mortality was reported in 2016 from the world's largest reef, the Great Barrier Reef (GBR), in Australia. The GBR has been affected by approximately 93% coral bleaching and 35% of the coral mortality (Cressey 2016). In the Indian reef, 23.92% ± 10.55% bleaching was recorded with 16.17± 8.46% coral mortality from March to June 2016 in the Gulf of Mannar (Patterson Edward *et al.*, 2018). Also, in the reef of the Gulf of Kachchh, 3.9% of coral bleaching observed (Arora *et al.*, 2019a). In the MMS, the temperature threshold for coral bleaching measured to be 29.8^{0}C caused coral bleaching and subsequent mortality in December 2015 (Raj *et al.*, 2018).

Fig. 8.8. Bleached corals at Malvan Marine Sanctuary. A-b, bleached and dead coral colonies covered by turf algae during the mass bleaching event; c, bleached massive *Porites compressa*; d, partially bleached colonies of *P. compressa*; e, bleached *Turbinaria mesenterina;* f, bleached *Favites* sp.

Mass coral bleaching caused by the ENSO events fuelled by climate change can trigger cascading ecosystem-level changes by reducing coral species heterogeneity, weakening the reef carbonate framework, loss of reef functional complexity, and negatively impacting on the reef-associated biodiversity (Brown 1997; Hughes *et al.*, 2018). Apart from causing coral bleaching, elevated thermal stress is also increasing abundance and virulence of coral disease-causing pathogens in the reef system, resulting in a further coral loss due to disease prevalence (Bruno *et al.*, 2007). Additionally, rates of coral calcification on the GBR and many other reef systems around the world have declined by 15–20% since ∼1990 due to increasing thermal stress (De'ath *et al.,* 2009). Bleaching mediated coral loss has transitioned or caused 'phase shift' in several coral ecosystems towards cyanobacteria, algae, and sponge-dominated ecosystem (Bell *et al.,* 2013; Madin & Madin, 2015; Selig *et al.*, 2012). Bleaching related 'phase-shift' is also reported even in the Indian reef systems (Machendiranathan *et al.,* 2016; Mote *et al.,* 2019).

Recent studies have shown that the intensity and the frequency of the El Niño increased in the past three decades (McPhaden *et al.,* 2011). Extreme El Niño events are predicted to be more frequent, severe, and spatially widespread in the future due to greenhouse warming (Eakin *et al.,* 2016; Ainsworth *et al.,* 2016; Heron *et al.,* 2016b), and the frequency of severe coral bleaching events is expected to increase even under moderate warming scenarios (Van Hooidonk *et al.,* 2016). Recovery potential of coral reef after mass bleaching events depends on the local pressure (fishing, sedimentation, pollution, exploitation, etc.) and the frequency of thermal stress (Arthur *et al.,* 2006), which can be achievable by strict reef management practice (Weijerman *et al.,* 2018). Most worryingly, studies projected that if coral bleaching events continue unabatedly due to the ongoing climate change, rates of disturbance will overwhelm the adaptive capacity and recovery potential of reef-building corals (Hughes *et al.,* 2018b; Lough *et al.,* 2018; Madin & Madin, 2015).

Fishing and eco-tourism related to the reef are the primary sources of livelihood for the local population at Malvan (De *et al.,* 2015). Apart from being the first long-term study on coral bleaching from this region, the present study also suggests that if this event continues, it will ruinously affect the marine biodiversity as well as the socio-economy of the region in the upcoming years. The emphasis of the present study is on the urgent implementation of long-term robust reef management plans to reduce the local stressors on the reefs for the bleaching recovery and to prevent further coral loss.

Chapter 9

9. Status of coral disease in Malvan Marine Sanctuary

9.1. Introduction

Facing extinction risk due to ongoing climate change and various human impacts, coral reefs globally are diminishing rapidly (Hughes *et al.,* 2014; Madin & Madin, 2015; Moberg & Folke, 1999). Since the last three decades, coral reefs in the Caribbean declined by an average of 50% and 80% (Bruno & Selig, 2007; Gardner *et al.,* 2003). Coral disease is one of the imminent threats, causing alterations in species diversity and ecosystem function, while reducing coral cover worldwide (Vega Thurber *et al.,* 2014; Lamb *et al.,* 2018).

Additionally, direct human activities including land-based pollution, nutrient influx, overfishing, sedimentation, tourism mediated coral physical damage enable the introduction of new pathogens in the reefs and increasing virulence of existing pathogens in the reef environments (Bruno *et al.,* 2003; Lamb *et al.,* 2014, 2018; Vega Thurber *et al.,* 2014). Coral disease cause significant mortality and decline of coral cover, followed by major ecological regime shifts from coral to algal proliferation (Bruno *et al.,* 2009). These changes often cascade into biodiversity loss, impaired ecosystem function & services with severe socio-economic consequences (Kennedy *et al.,* 2013).

Though, sporadic reports on the occurrence of the coral disease from the major Indian reefs are available (e.g. Ravindran *et al.,* 1999; Ravindran & Raghukumar, 2002; Thinesh *et al.,* 2009, 2014, 2017; Thangaradjou *et al.,* 2016); however, detailed investigation of cause and consequence of disease outbreaks in the reef environment is limited. Most of the accounts are based on only a single time point observation and long-term studies on coral disease dynamics and ecological implications are missing. Admittedly, coral disease research is still in a nascent state in Indian coral reefs compared to the magnitude of its impact. The present study aimed to determine the temporal variation in coral disease prevalence around the Malvan Marine Sanctuary (MMS), a Marine Protected Area (MPA) along the west coast of India in the eastern Arabian Sea.

Although, the study area was designated as an MPA in 1987 and known as a 'biodiversity hotspot', it is one of the understudied coral reef areas in India with few recent reports on coral bleaching and coral mortality in the MPA during 2014 and 2016 (De *et al.,* 2015; Raj *et al.,* 2018). The only report (Hussain *et al.,* 2016) on coral disease from this reef highlights the presence of skeletal tissue growth anomalies (STA) for a major reef builder *Turbinaria mesenterina*. Lack of long-term systematic observations make it difficult to build a cause-effect phenology for the onset of multiple stressors, thus, our study is an account of comprehensive 5-year seasonal observation that justify robust findings. The present study reveals the occurrence of several coral diseases, white syndrome, tissue necrosis and infestations of coral boring mollusc and trematodiasis for the first time from this MPA. Hypothesized that most of the coral health was damaged due to diseases that was triggered by the bleaching event and significantly mediated by macroalgal phase shift.

9.2.Material and methods

9.2.1. Survey design and data collection

Underwater surveys were undertaken at four patch reefs (ST#1-ST#4) around the MPA during the fair-weather months of October 2014, November 2015, May 2016, May 2017, October 2018 and April 2019 for monitoring coral health and bleaching events. Coral disease rapid assessment was carried out November 2015 onwards following the monitoring protocol described by Raymundo *et al.,* (2008). During each survey, belt transects (20m x 2m) (n = 3 per reef) were placed at the reef sites, and transects were laid approximately 10m interval at 3-10m depth. GPS coordinates of the sites and the start and endpoint of the transect were recorded using handheld Garmin GPS. The subsequent annual surveys were conducted following the same GPS coordinates to monitor the same patch reefs. The areal cover of the dominant benthic component within the belt transects was quantified by the line-intercept method (20m), following the central transect tape of the belt transects (English *et al.,* 1997). Coral colonies within each transect were identified up to the genera level, enumerated, photographed, and examined for symptom of bleaching, disease, and compromised health, and underwater photographs of corals and diseased colonies were used for confirmation of in situ identification. The total number of uninfected and disease-infected colonies were counted to estimate the disease prevalence (percent corals with disease symptom in a transect) (Raymundo *et al.,* 2008).

Observed diseases based on the external appearance of corals are described in Table 9.1 (Raymundo *et al.,* 2008; Aeby *et al.,* 2017). Coral mortality was estimated by assessing the reduction in percent live coral and as well as a headcount of live coral colonies in each belt transects in each year. The density of the juvenile coral colonies (>5 cm max diameter that can be detected in the field visually) were documented along each transect.

9.2.2. Analysis of water parameters

Analysis of Sea Surface Temperature(SST) trends and daily SST anomaly from January 2014 to April 2019 for the MMS was carried out using the National Oceanic and Atmospheric Administration (NOAA) Coral Reef Watch's (CRW) near real time daily global 5km (0.05 degree exactly) satellite sea surface temperature (SST) monitoring high-resolution time series data product known as 'CoralTemp' Version 3.1 (available from NOAA Coral Reef Watch 2019 https://coralreefwatch.noaa.gov).

Physical water parameters, *viz*. temperature, salinity, pH was measured in situ using a YSI multiparameter probe (model: YSI Professional plus, make YSI, USA). Water samples were collected and refrigerated onboard for analysis of nutrient and suspended particulate matter (SPM). One litter of water sample was filtered through a 0.22µm polycarbonate filter (pre-weighed), and the SPM was estimated by measuring the weight of matter retained on the filter. The essential dissolved nutrients like Nitrate, Nitrite, and Phosphate were analysed using a Skalar SAN++ Continuous Flow Analyzer.

Table 9.1. Descriptions of different coral disease observed in the Malvan Marine Sanctuary

Observed disease	Description/morphologic diagnosis	Figure
Skeletal tissue growth anomalies (STA)	Irregular growth formations on the coral skeleton, overgrown corallites, tumour formation.	Fig. 9.4 (A, B, C)
Boring mollusc infestation (VER)	Presence of Vermetid gastropods and boring bivalves on massive, sub-massive, and encrusting corals. Presence of multiple boreholes or burrows on the coral colony and secretion of mucus net.	Fig. 9.4D
White syndrome or tissue loss diseases (WS)	Presence of lesions, tissue loss, the appearance of white patches, or discoloured spots on the coral colonies.	Fig.9. 4 (E, F)

Tissue necrosis (TN)	The occurrence of tissue sloughing, tissue loss, and presence of necrotic patches on the colonies.	Fig. 9.4G
Trematodiasis (TRM)	Pink colour spots on the colony, pink swelling of polyps.	Fig. 9.4H

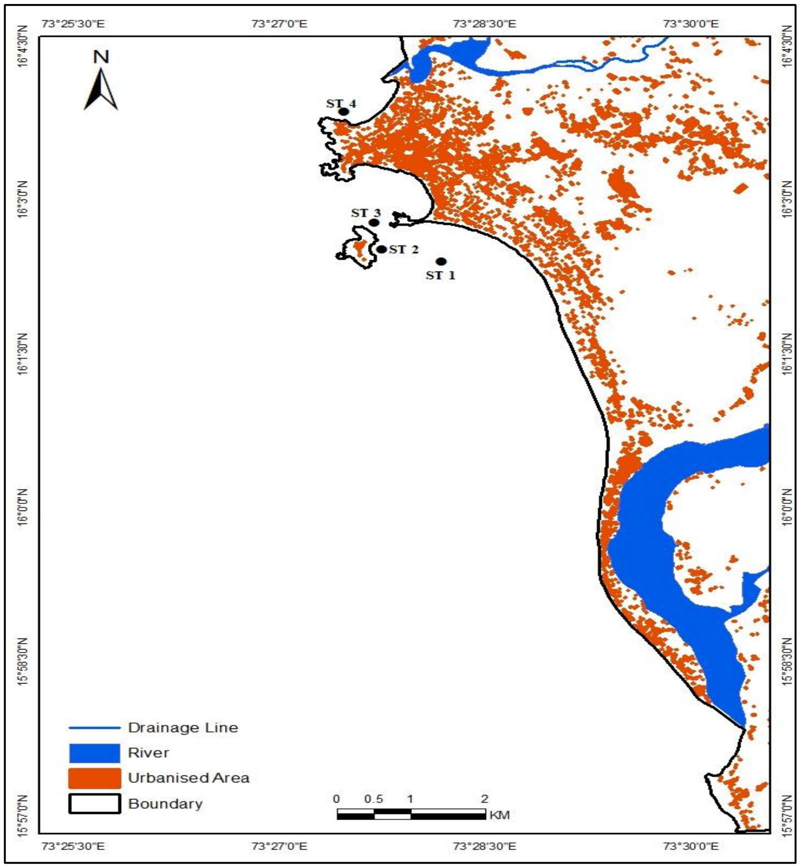

Fig. 9.1. Location of the study sites in the Malvan Marine Sanctuary

9.2.3. Data analysis

Disease prevalence calculated by dividing the number of infected colonies at each transect by the total number of colonies at that transect and expressed as a percentage (Raymundo *et al.*, 2008). Site-level mean disease prevalence or intensity and standard errors were estimated from the belt transects. Three prevalence values from replicate transects at each site further divided by the number of transects (n=3) to calculate a mean for each site.

9.2.4. Statistical analysis

Abundance data were normalized by square root transformation, and environmental data were log (X+1) transformed to meet the normality assumptions. Significance of spatial and temporal variation of pooled disease prevalence data was tested using two-factor analyses of variance (ANOVA) at the $p < 0.05$ level of significance. Further, the Pearson's correlation analysis was performed in order to find out the strength of association between the prominent benthic component, viz. coral cover, benthic algal cover, and disease prevalence. Non-Multidimensional Scaling (nMDS) and Bray–Curtis dissimilarity matrices were constructed based on disease prevalence data after square root transformation. To compare Spatio-temporal variation in disease prevalence, a two-way Crossed analysis of similarities (ANOSIM) was used (999 permutations). The analysis was carried out using Excel data package and PRIMER-e v6 (Clarke & Warwick, 1994). All data are stated as means ± standard error.

9.3. Results

9.3.1. Heat stress exposure

The 2015-2016 El Niño Southern Oscillation (ENSO) was elevated the sea surface temperature (SST) across the tropical oceans and unleashed unprecedented thermal stress on corals that resulted in mass coral bleaching (Arora *et al.,* 2019a). SST data derived from NOAA Coral Reef Watch showed 1.38^{o}C monthly anomaly in October 2014, whereas monthly mean SST calculated to be 29.16^{o}C. In October 2015, monthly mean SST was 29.56^{o}C with an anomaly of 1.82^{o}C, positive SST anomaly persisted in November 2015 measured to 1.59^{O}C, while monthly mean SST estimated 29.62 ^{O}C. Average monthly SST rose to 30.11°C in May 2016, and all the days in May showed positive SST anomaly with a monthly mean positive anomaly of 0.90°C, whereas in-situ bottom water temperature was ranged between 29.97-30.16°C. During May 2017, the mean monthly SST was 29.70°C, with a mean anomaly of 0.48°C, and our in-situ observation recorded 29.19-29.77°C. In October 2018, the mean monthly SST was 29.21°C with continuous positive thermal stress. The, mean monthly SST anomaly calculated was 1.43°C and the mean bottom water temperature was 29.72-30.22°C. In April 2019, the mean SST was 29.25°C with an anomaly of -0.26°C, and bottom water temperature recorded was 29.82-30.06°C. The recorded satellite derive SST data and in-situ temperature data indicated that corals at MMS experienced significant heat stress during a prolonged period in 2014-2019, which causes in back-to-back coral bleaching events.

Recurrent coral bleaching events and subsequent coral mortality were recorded in the MMS since October 2014 to April 2019 (Fig.9.2). Coral bleaching survey in October 2014, estimated mean bleaching of 14.58%±1.75SE, comprised bleached colonies of *Porites lichen, Porites compressa, Favites melicerum, Turbinaria mesenterina, Pseudosiderastrea tayami, Cyphastrea serailia, Plesiastrea versipora, Goniopora* spp,. *Siderastrea savignyana.* Bleaching intensity rose to 54.20%±2.58SE in November 2015. Further, major bleaching recorded in May 2016 to be 46.76%±2.26SE and 20.22%±0.73SE during May 2017. Bleaching intensity recorded 5.07%±0.61SE in October 2018 and 8.37%±1.09SE.

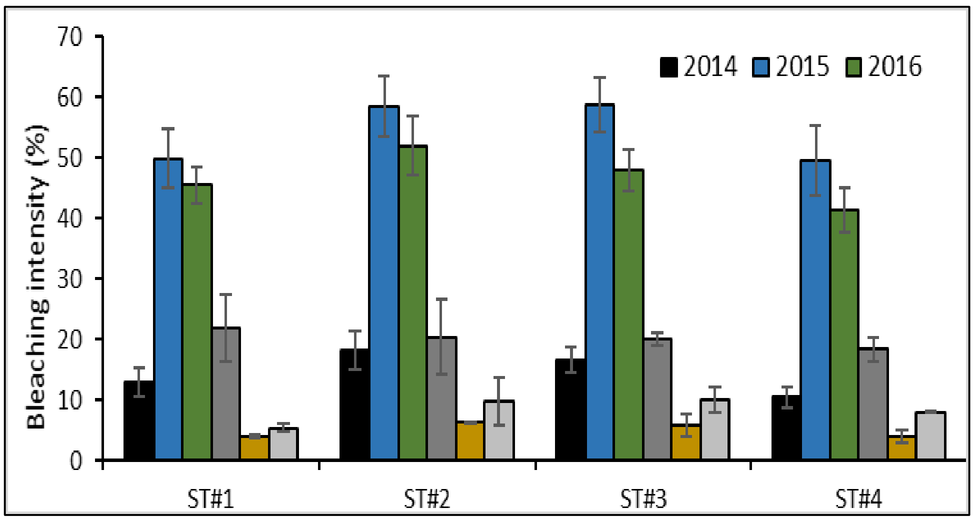

Fig. 9.2. Mean (±SE) coral bleaching intensity at different reefs along the MMS during October 2014, November 2015, May 2016, May 2017, October 2018 and April 2019, respectively.

9.3.2. Changes in coral and algae cover

During our first survey in the MPA in 2014, live coral cover was estimated to be 45.09% and was reduced to 19.72% in 2019. During November 2015, mean coral cover measured 38.65± 1.06SE and mean algae cover was estimated 31.50± 1.10SE. In May 2016, mean coral cover was recorded to 33.32% ± 1.12SE with a mean algae cover was 36.37%±1.28SE. In May 2017, coral cover was 26.31%± 1.07 SE, and algae cover increased to 48.20%±1.31SE. During October 2018, the measured live coral cover was reduced to 22.25%±1.59SE, and the algal cover was rose to 54.16%±1.67SE. While, in April 2019, coral cover further declined to 19.72%± 1.47SE and algae cover spiked to 59.47%± 1.56SE with a spreading of macroalgae and turf algae over the recent dead coral colonies (Fig. 9.3A). The alteration between coral cover and macro-algae dominance were statistically analysed by conducting paired two samples t-test, and the Pearson correlations showed a strong negative correlation (r = -0.94212, n=74, P>0.0005, Fig.6.5) between coral and benthic algal cover, which confirms the coral cover decrease with increasing algal cover.

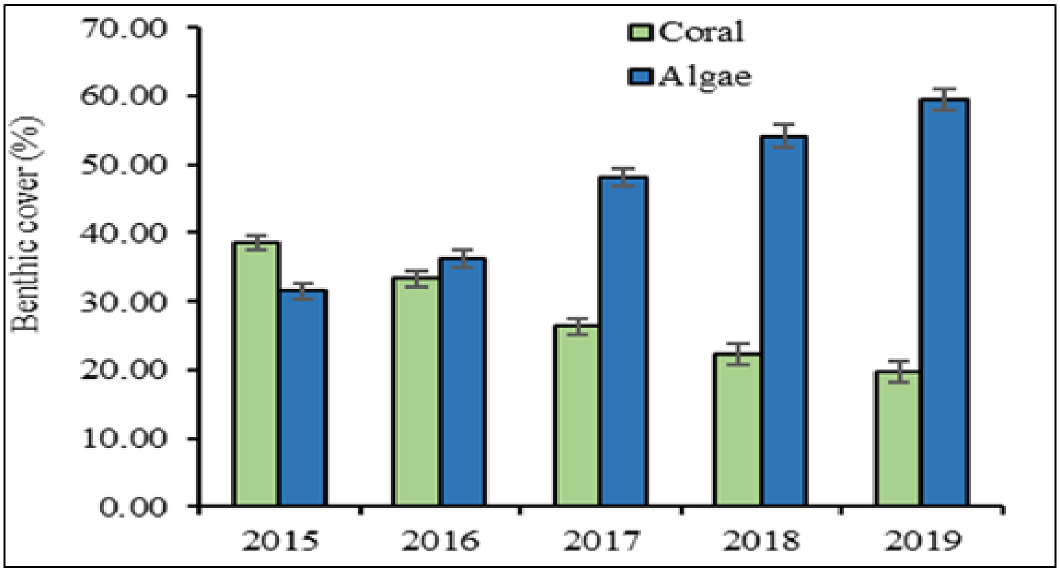

Fig. 9.3. The mean (±SE) benthic percent cover of live hard coral and algae (macro and algal turf) during the study period.

9.3.3. Abundance and distribution of diseased corals

The 12 replicate transects on the four reefs covered 240m^2 of the reef during each year, containing a total of 2688 coral colonies during the study periods (total 48 transects and 960 m^2 during the four years of study). A steady increase in coral disease prevalence was observed at the study site during the four years study period. A total of five types of coral diseases were found in all the study sites, viz. skeletal tissue anomaly (STA), infestation of boring mollusc (VER), white syndrome (WS), tissue necrosis, or necrotic patches on colonies (TN), and trematodiasis or pink spot (TRM) (Table 9.1 and Fig.9.4). The incidence of infected colonies during the entire study period (pooled disease prevalence data averaged across 2016-19) averaged 22.35% of the total number of colonies present (n=601), with a range of 10.89 to

47.83% per transect. The percent of diseased colonies (pooled disease colony data) was significantly different between the four-reef site. Variation of coral diseases among survey sites showed that the prevalence of coral diseases varied significantly among sites (two-way Anova, *df*=3, F= 17.9155, $p<0.001$) and as well as between the years (two-way Anova, *df*=3, F= 14.1084, $p<0.001$). The ANSOIM result showed that disease prevalence differs significantly among the years (R= 0.89; P= 0.1%), while no significant differences were observed between the sites (R= 0.59; P= 4.9%).

Fig. 9.4. Underwater photographs illustrate different type of coral disease observed during the present study, A. Skeletal tissue growth anomaly in *Turbinaria mesenterina*; B. Growth anomaly

in *T. mesenterina;* C. Growth anomaly on *Favites* sp.; D. Structural deformation of coral skeleton due to infestation of coral boring vermetids in *Plesiastrea versipora* colony; E-F. White syndrome on *Porites* sp. and *T. mesenterina*; G. Tissue necrosis of *Goniopora* sp.; H. Trematodiasis on *Porites* sp.**

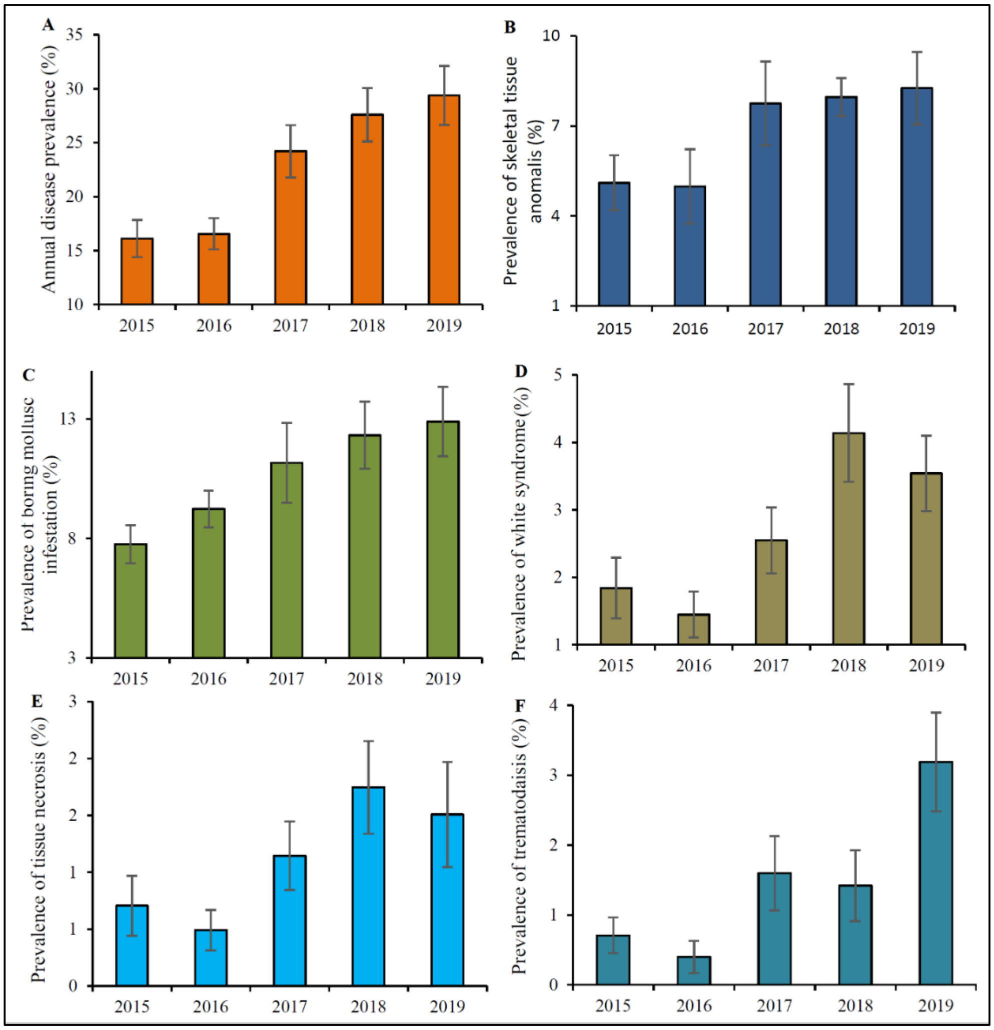

Fig. 9.5. A. Annual prevalence of major coral diseases (Mean±SE) in the Malvan Marine Sanctuary; B. Prevalence of skeletal tissue anomalies; C. Prevalence of boring mollusc infestation; D. Prevalence of white syndrome; E. Prevalence of tissue necrosis; F. Prevalence of Trematodiasis all data are pooled mean percentage value)

9.3.4. Spatio-temporal variation in coral disease prevalence

Coral disease prevalence was found to have amplified from the initial observations at all study sites during the monitoring period. Overall, an increase of 12.85% in the disease was observed.

Yearly disease prevalence recorded 16.55%±1.44SE when the study was initiated in 2016, whereas in 2019, prevalence reached to 29.39%±2.73SE. The highest number of diseases colonies were found at ST#2 near the Sindhudurg Island. The percentage of infected coral colonies was 33.39% in 2019, 35.39% in 2018, 34.37% in 2017, 23.33% in 2016. This was followed by ST#3 with 33.39% infected colonies in 2019, 24.78% in 2018, 22.91% in 2017, 14.76% in 2016. The trends of disease prevalence for each disease category are presented in in Fig. 9.5 A-F. The disease prevalence data were subjected to Bray-Curtis cluster analysis which detected three major groups (Fig. 9.6A). Group I, consisted of all sites sampled during 2016 with 91.67% similarity. Site1 (ST#1) and site four (ST#4) of 2017 formed a separate group II with 95.52% similarity. The rest of the sites formed the group III clusters. MDS ordination plot further confirmed the results of cluster analysis and detected the same three groups (Fig. 9.6B).

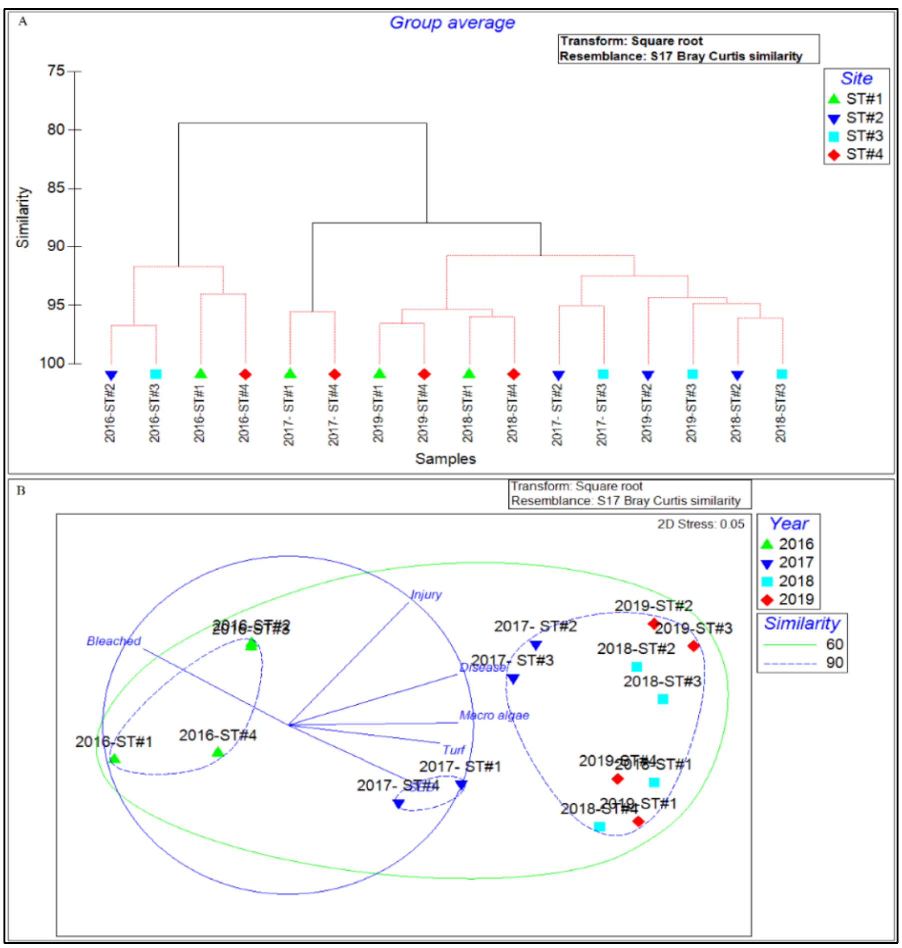

Fig. 9.6. A. Cluster-based on hierarchical cluster analysis based on disease prevalence data illustrating the dissimilarity between survey sites and years; B. Non-metric multidimensional

scaling plot illustrating the multivariate Bray–Curtis dissimilarity between survey sites and years based on disease prevalence data.

In 2016, mean disease prevalence accounted to be 16.55±1.44SE (n=166; among total 1010 colony accounted). STA was estimated to be 4.98±1.24SE (n=50), coral boring mollusc infestation was 9.23±0.77 SE (n=92), WS was 1.45± 0.34SE (n=15), TN was 0.49±0.18SE (n=5), and TRM was 0.40±0.23SE (n=4). Mean disease prevalence at ST#1 was estimated to be 13.91±1.53SE (n=36). Coral boring mollusc infestation was highest at this site (9.38±1.00SE, n=25), followed by STA (2.04±1.45SE; n=5). At ST#2, the mean disease prevalence estimated to be 23.33±2.97SE (n=58). Coral boring mollusc infestation was found to be the highest (10.92±1.51SE; n=27), followed by STA (10.42±2.83 SE, n=26). The mean disease prevalence at ST#3 recorded to be 14.76±1.95SE (n=32). At this site, coral boring mollusc caused the major stress (10.10±1.53SE, n=22) followed by STA (3.70±0.49, n=8). At ST#4, mean disease intensity was noted 14.21±0.27 (n=40). The infestation of coral boring mollusc accounted to be a major concern for coral health (6.51±1.35SE, n=18), the next prevalent disease at ST#4 was STA (3.76±167, n=11), whereas WS contributed to 2.5% of the disease (0.38SE, n=7).

During 2017, the mean disease prevalence accounted to be 24.21±2.42SE (n=143; among the total of 667 colonies accounted). Mean percent of STA was estimated to be 7.75±1.40SE (n=44), the coral boring mollusc infestation was 11.17±1.67 SE (n=66), WS was 2.55± 0.49SE (n=16), TN was 1.15±0.30SE (n=7), and TRM was 1.60±0.53SE (n=10). At ST#1, the mean disease prevalence estimated to be 19.66±0.79 (n=35). Coral boring mollusc infestation was measured to be 9.15±2.67SE (n=17), followed by STA (4.02±0.87SE; n=7), TRM (2.97±1.40SE, n=5), and WS (2.36±1.19SE, n=4). At ST#2, the mean disease prevalence was 34.37%±3.18SE (n=45). STA was found to be dominant (14.57±1.55SE, n=19), next was Coral boring mollusc (12.35±4.43, n=16), followed by WS (2.97±0.56, n=4), TN (2.29±0.13SE, n=3) and TRM (2.19±1.19SE, n=3). At ST#3, disease intensity was recorded to be 22.91±5.40SE (n=29). VER was highest (14.08±5.06SE, n=17), followed by STA (6.44±1.82SE, n=8), WS (1.26±0.64SE, n=2). TN and TRM both accounted to be 0.56±0.56SE (n=1). Mean disease prevalence was 19.88±4.13SE, n=34 at the ST#4. VER formed major coral disease (9.08±0.37SE, n=16), followed by STA (5.97±2.02SE, n=10), WS (3.60±1.28SE, n=6), TRM (0.65±0.65SE, n=1).

During 2018, mean disease prevalence accounted to be 27.59±2.47SE (n=151; among the total 551 colonies accounted). STA was estimated to be 7.96±1.33SE (n=43), VER infestation was 12.32±1.40SE (n=67), WS was 4.14± 0.73SE (n=23), TN was 1.75±0.41SE (n=10), and TRM was 1.42±0.51SE (n=8). The mean disease prevalence at ST#1 recorded to be 24.58±1.82SE (n=37). VER was 10.78±4.84SE (n=16), followed by WS (4.54±1.19SE, n=7), STA (4.11±2.52SE n=3), TN (2.60±0.52SE, n=4), TRM (2.54±1.59SE, n=4). At ST#2, the mean disease prevalence was recorded to be 35.39±7.86SE (n=44). Among the different type of diseases, coral boring mollusc infestation was dominant (13.47±3.28SE, n=17), followed by STA (12.42±3.58SE, n=16), WS (6.69±1.51SE, n=8), TN (1.77±0.92SE, n=2). At ST#3, overall disease prevalence was recorded 24.78±3.05SE (n=26). The infestation of coral boring mollusc was accounted to be 13.34±2.24SE (n= 14), followed by STA (7.73±1.07SE, n=8) and WS (1.85±0.93SE, n=2). TN and TRM were recorded to be minimum 0.90±0.90SE and 0.95±0.95SE. At ST#4, the mean coral disease prevalence was recorded to be 25.61±4.36SE (n=44). Coral boring mollusc infestation was calculated 11.70±1.26SE (n=20), followed by STA (7.59±0.96SE, n=13), and WS (3.47±0.92SE, n=6). Other diseases prevalence includes TN (1.71±1SE) and TRM (1.14±0.57SE).

During 2019, the mean disease prevalence accounted to be 29.39±2.73SE (n=141; among the total 501 colonies) in the MMS. STA was estimated to be 8.26±1.21SE (n=38), Coral boring mollusc infestation was 12.89±1.45SE (n=63), WS was 3.54± 0.56SE (n=18), TN was 1.17±0.21SE (n=18), and TRM 3.19±0.71SE (n=14). At ST#1, disease prevalence recorded to be 23.75±1.99SE (n=35). Coral boring mollusc infestation was highest (10.75±1.53SE, n=16), followed by WS (4.74±0.70SE, n=7), STA (4.13±1.29SE, n=6), TRM (2.10±1.23SE, n=3) and TN (2.03±.0.06SE, n=3). At ST#2, disease prevalence was estimated to be 33.93±7.65SE (n=40). Coral boring mollusc infestation recorded to be 15.08±4.16SE (n=18), followed by STA (7.48±1.59SE, n=9), TRM (4.06±0.73, n=5), WS (3.42±1.71SE, n=4) and TN (3.35±0.96, n=4). At ST#3, the mean disease intensity was recorded to be 34.25±7.76SE (n=27). Among the diseases, coral boring mollusc infestation was recorded highest (13.59±4.11SE, n=11), followed by STA (12.73±2.73SE, n=10), TRM (5.25±192SE, n=4), WS (2.66±1.37SE, n=2). Mean disease prevalence at ST#4 accounted to be 26.16±0.93 (n=39). The infestation of boring mollusc was 12.14±2.16SE (n=18), followed by STA (8.68±1.22SE, n=13), WS (3.34±0.62SE, n=5), TRM (1.35±0.67SE, n=2) and TN (0.65±0.65SE, n=1).

9.3.5. Correlation between the benthic component and coral disease prevalence

A significant negative correlation was detected between mean disease prevalence and high coral cover (Single-factor Anova, $p<0.005$, $R=-0.89$, $R^2=0.81$) (Fig.9.7A). Contrastingly, a positive correlation was established between the disease prevalence and benthic algal cover (single factor Anova, $p<0.001$, $R=0.91$, $R^2=0.83$; Fig. 9.7B). The decline in coral cover showed an increasing trend in STA (single factor Anova, $p<0.001$, $R=0.67$, $R^2=0.45$; Fig. 9.7C) and boring mollusc infestation rate (single factor Anova, $p<0.001$, $R=0.86$, $R^2=0.74$; Fig. 9.7D). Mean disease prevalence and coral physical damage showed a positive correlation (single factor Anova, $p<0.001$, $R=0.71$, $R^2=0.51$; Fig. 9.7E). However, surveys revealed that the density of juvenile coral recruits decreases with the higher disease prevalence (single factor Anova, $p<0.001$, $R=0.52$, $R^2=0.27$; Fig. 9.7F).

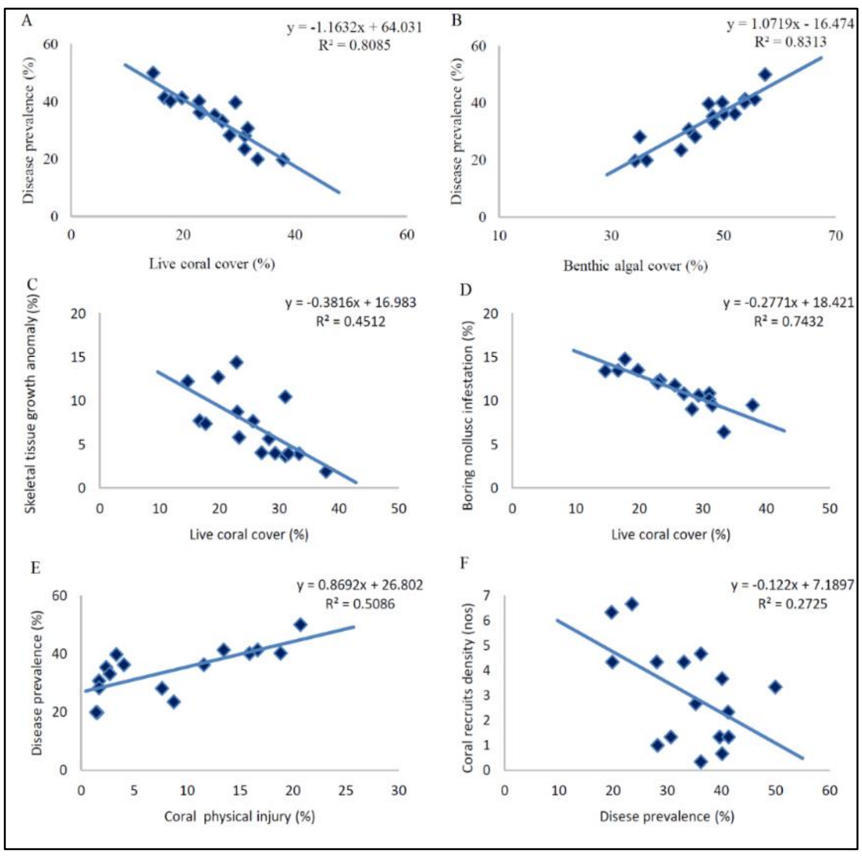

Fig. 9.7. Relationship between the mean disease prevalence- A. live coral cover; .B. benthic algal cover; relationship between live coral cover- C. the intensity of skeletal tissue growth anomaly

and; D. Boring mollusc infestation; E. relationship between mechanical injury in coral colony and disease prevalence; F. relationship between coral disease prevalence and density of juvenile coral recruits.

Further, a significant positive correlation was found between the coral disease prevalence and the intensity of macroalgal contacts with the live corals (single factor Anova, $p<0.005$, $R=0.78$, $R^2=0.61$) (Fig. 9.8A) and the intensity of turf algal intrusion on live corals (single factor Anova, $p<0.13$, $R=0.84$, $R^2=0.70$) (Fig. 9.8B). Furthermore, a positive correlation between the boring mollusc infestation and the benthic algal cover was detected (single factor Anova, $p<0.001$, $R=0.81$, $R^2=0.66$; Fig. 9.8C). A weak correlation was also found between STA and benthic algal cover (single factor Anova, $p<0.001$, $R=0.52$, $R^2=0.27$; Fig. 9.8D). Likewise, weakly positive correlation was detected between the trematodiasis and algal cover (single factor Anova, $p<0.001$, $R=0.54$, $R^2=0.30$), and among WS and algal cover (single factor Anova, $p<0.001$, $R=0.43$, $R^2=0.18$).

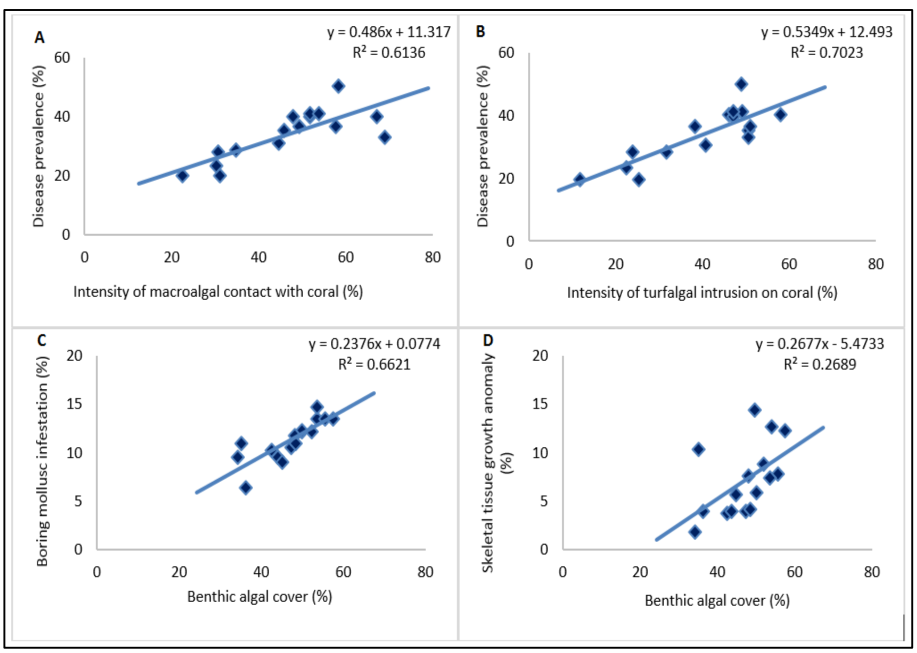

Fig. 9.8. Relationship between the mean disease prevalence- A. intensity of macroalgal contacts with live corals; B. intensity of turf algal intrusion on live corals; C. relation between benthic algal cover and intensity of boring mollusc infestation; D. relationship between benthic algal cover and prevalence of skeletal tissue growth anomaly.

9.4. Discussion

Many historic events shape the fate of coral reefs, from volcanic eruption, temperature spikes, bio-invasion, bleaching to diseases (De'ath, 2012). However, most of the studies are yearlong studies fail to detect the set of successive events that cascade in the present status of a habitat. This study has revealed a significant decline in hard coral cover from 45.09 in 2014 to 19.72% in 2019 over a five years period with an average annual decline of 5.07% y^{-1}, based on methodologically consistent surveys. The rate of coral decline in much higher than the reported coral cover decline in Great Barrier Reef (0.53%y^{-1}) (De'ath *et al.,* 2012), Indo-Pacific (1%) (Bruno & Selig, 2007). This comprehensive survey gives insights into the dwindling status of corals from a marine protected area on the central west coast of India. Malvan Marine Sanctuary is the only coral reef patchily distributed on fringing rocky coastline having basalting and lateritic outcrops. Land locked towards east (Fig. 9.1) the nearest coral habitat is 200km south (Grande Island), 200km west on a Seamount (Angriya Bank) and towards north at around 700km (Gulf of Kachchh). Thus, considering the isolated patchiness of this important habitat, it is imperative to conserve and protect this habitat and apparently has been under protection since last 4 decades. But, being a coastal habitat, in the recent years has experienced concoction of adversities ranging from local stresses to global events instigated by the 2015-16 *El Nino*.

9.4.1. *El Nino* triggered bleaching exacerbated coral disease

The 2015-16 *El Nino* event spiked the temperature in this coastal water, negatively impacted the coral community from the MPA causing significant bleaching (Fig. 9.2). The temperature anomaly that crossed the threshold of coral tolerance caused mean bleaching of 14.58% in 2014, followed by 54.20% in 2015, 46.76% in 2016, 20.22% in 2017, 5.07% in 2018, and 8.37% in 2019. This complete and partial beaching greatly impaired coral species in recovery while making them vulnerable to diseases being at their weakest (Brandt & McManus, 2009; Brodnicke *et al.,* 2019; Miller & Richardson, 2015). Although coral communities at these locations are presumably resilient, bearing the adverse coastal settings (Nair & Qasim, 1978), this study reports a series of undesirable impacts over the course of 5 years triggered by temperature anomaly due to *El Nino* effect in 2015-16. First disease survey of 2015 notes the dominance of molluscan infected coral colonies (Fig.9.3 and 9.4C), presumably implying that infestation was an ongoing phenomenon and its onset is not the consequence of *El Nino*. As the

MPA must be experiencing urban organic load and high turbidity due to localized hydrodynamic setting (De *et al.,* 2015; Hussain *et al.,* 2016; Raj *et al.,* 2018), it provides an opportunistic environment favouring mollusc infestation (Shima *et al.,* 2013, 2015).

9.4.2. Macroalgal phase shift

The temperature anomaly cause bleaching in significant coral colony (54.20% in 2015, 46.76% in 2016) while opening a niche for others to occupy (Bellwood *et al.,* 2006; Mcmanus & Polsenberg, 2004). This was an opportunistic chance for seasonally proliferating macroalgae, as they find vacant non-allelopathic substratum to establish. Seasonal seaweed proliferation brought the second wave of deleterious effects in post-monsoon of 2015/16 which covered sizable area (Fig.9.3) predominantly by *Caulerapa* sp., *Sargassum* sp. Algal phase shifts are known to have significant negative impact on coral communities which includes dominance of macro-algae, and turf algae (Bruno *et al.,* 2009; Fung *et al.,* 2011; Wild *et al.,* 2014). Seaweed proliferation triggered spread of pathogens that infected corals with potential pathogenic bacteria known to act as a causative agent of white syndrome and other coral diseases (Nugues *et al.,* 2004; Sweet *et al.,* 2013). Macroalgae or seaweeds are also produce secondary metabolites those are allelopathic to corals and have potency to cause coral diseases (Longo & Hay, 2017; Rasher and Hay, 2010; Sweet *et al.,* 2013). Our study also shows that a total 16.55% of the colonies were infected during 2016 with multiple diseases (Fig.9.6), in the subsequent years, the average disease spread increased by 3.21% annually with increasing algal proliferation in the reef environment. Macroalgae requires nutrients which was delivered in the MPA by the constant organic drainage from landward urban increased usage. This drainage has remained steady (nutrient data did not show an increasing trend) over the years and could help the macroalgal bloom only due to unoccupied space created by bleaching event. In 2019 survey, most of the areas (59.47%) in the MPA were noted with algae having 6.14% rate of increase per year. Subsequent ENSO bleaching events gave scope to seasonal space competitors (seaweeds), which proliferated coral diseases. All these deleterious entities overwhelmed the corals because the habitat had increasing localized anthropogenic stresses (Fig. 9.9) over the duration of the study.

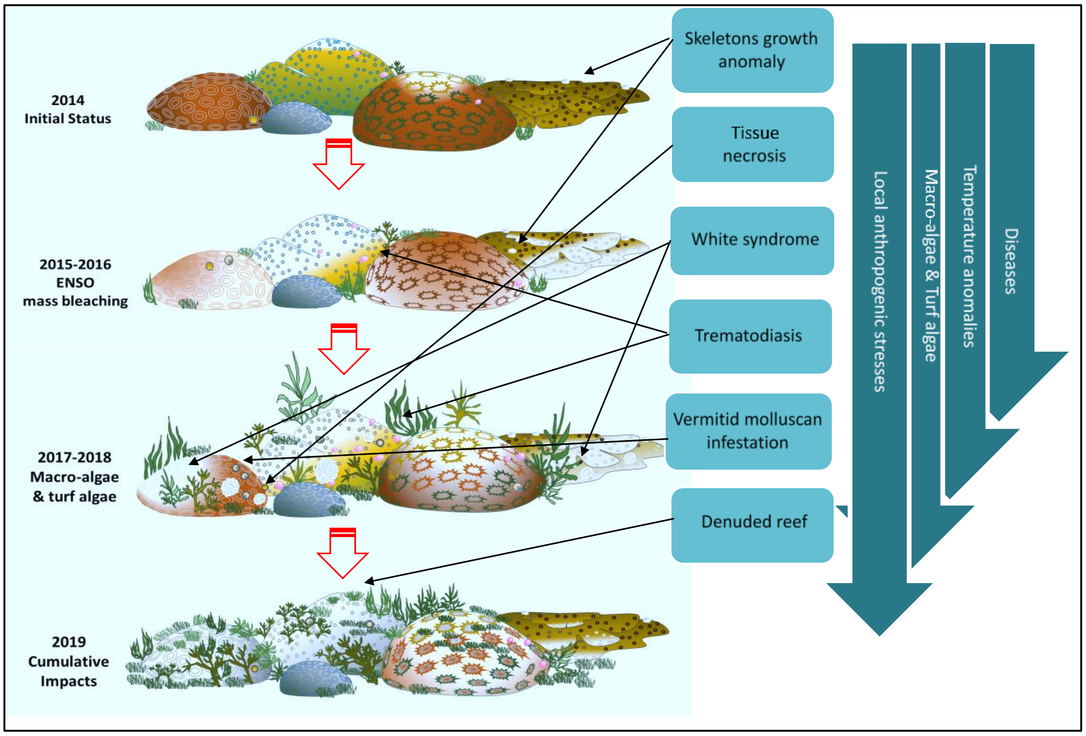

Fig. 9.9. Graphical representation of the chronology of the events observed in the studied reef

9.4.3. Disease proliferation

Globally, diseases have proved to be major threat (Harvell *et al.*, 2007; Raymundo *et al.*, 2008) impairing coral physiology, survival, recruitment, development, population structure and diversity (Miller & Richardson, 2015; Pollock *et al.*, 2011). In the recent past, emergence of new coral diseases and syndromes have been reported with limited knowledge about their causes, pathology and aetiology (Pollock *et al.*, 2011). Reportedly, this MPA has been exposed to intense anthropogenic stress with concurrent temperature anomalies (De *et al.*, 2015; Raj *et al.*, 2018). Significant positive correlation occurred between algal physical contact with corals and disease prevalence (Fig.9.8A-D) and subsequent mortality which is a noted trend (Nugues *et al.*, 2004; Sweet *et al.*, 2013).

At both ends of the MPA (Fig. 9.1) two small rivers (Kolamb and Karli river) add sediment runoff, responsible for high suspended particulate matter (SPM), nutrients and turbidity in to the coral reef habitat. Apart from the rivers, several domestic drains directly discharge sewage, debris and plastic waste in to the Bay thus increasing the organic load and associated pathogenicity. Concurrently, this region experienced heavy rainfall (Fig. 9.10) during the southwest monsoon during 2016 to 2019 (IMD 2019) bringing freshwater in the reef and with it large amount of terrigenous sediment. Runoff brings organic matter enriched with nutrients in the reef area that increases microbial load while elevating disease prevalence and virulence (Kim & Harvell, 2001; Bruno *et al.*, 2003; Haapkylä *et al.*, 2011). Poor water quality with high pathogen load negatively impacts the coral colonies by interfering with crucial physiological mechanism, reducing calcification rates, fecundity, fertilization, and larval development (Fabricius 2005; Redding *et al.*, 2013) and also increases white syndrome disease (Pollock *et al.*, 2014).

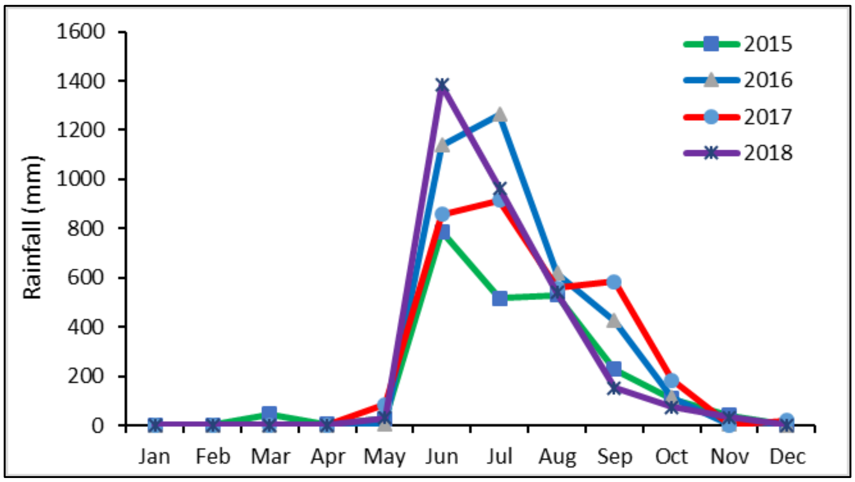

Fig. 9.10. Annual average rainfall in the study area (data derived from the IMD http://hydro.imd.gov.in/hydrometweb/(S(0ut4ru45epptbn55jkkkfy2m))/landing.aspx#)

Analysis of seawater from the MPA revealed the presence of known coral pathogenic bacteria viz; *Vibrio splendidus,* and *Pseudomonas putida* known to cause white plague syndrome (Portillo *et al.*, 2018, Roder *et al.*, 2014, Cárdenas *et al.*, 2011). *Pseudomonas fluorescens, Pseudomonas syringae*, *Pseudomonas moraviensis,* and *Aeromonas hydrophila* cause white syndrome (Roder *et al.*, 2014, Kooperman *et al.*, 2007*). Rickettsia prowazekii* responsible for White band disease in coral (Cases *et al.*, 2004, Roder *et al.*, 2014). These pathogens could be responsible for observed white syndromes, however, further studies are required to validate the microbial origin of this pathology. Cumulative impacts of local and global stress showed cascading effects on the coral health with positive cause-effects relationship.

STA prevalence has been attributed to acute sedimentation and turbidity (Riegl *et al.,* 1996), elevated water temperature (Peters *et al.,* 1986), nutrient enrichment (Bruno *et al.,* 2003), bacterial load (Breitbart *et al.,* 2005) and coral's physical contact with macroalgae (Morse *et al.,* 1981). STAs are characterized by atypical skeletal forms, overgrown corallites, low number of polyps, reduced zooxanthellae and abnormal pigmentation (Bak 1983; Breitbart *et al.,* 2005; Domart-Coulon *et al.,* 2006; Burns *et al.,* 2011; Stimson 2011) and known to reduce growth rate, fecundity and increased bleaching (Stimson 2011; Yamashiro *et al.,* 2011).

Coral tissue loss diseases collectively known as white syndromes (WS), a virulent group of coral diseases results in significant mortality decreasing the coral cover on reefs throughout the Indian Pacific Oceans and GBR (Work *et al.,* 2012; Pollock *et al.,* 2017; Heron *et al.,* 2010). Yet, aetiologies for WS remains elusive. Elevated heat stress have been linked to increased populations and virulence of pathogens associated with WS outbreak (Heron *et al.,* 2010; Brodnicke *et al.,* 2019) wherein the MPA, about 14% was noted in the MPA (De *et al.,* 2015) and during December 2015 revealed high bleaching intensity of 71 % with mortality of 8% (Raj *et al.,* 2018).

Present study demonstrates a high infestation rate of coral boring molluscs includes vermetids (*Dendropoma* spp.) and other boring bivalves (*Lithophaga* spp.) across the reefs, characterized by the presence of a mucus net on the coral surface for suspension-feeding from the water column (Zvuloni *et al.,* 2008). Common throughout the Indo-Pacific (Shima *et al.,* 2010) high infestation rate of coral boring vermetids gastropods (*Dendropoma* spp.) and bivalves (*Lithophaga* spp.) was noted on *Porites* and *Plesiastrea,* with characteristic mucus net (Fig. 9.4D) for suspension-feeding from the water column (Zvuloni *et al.,* 2008). This infestation is known to reduce photochemical efficiency, growth rates, affect colony morphology, decrease skeletal growth by up to 81% and survival by up to 52% of corals and alter species composition within a short time-scales (Shima *et al.,* 2010, 2013; Wong *et al.,* 2016). They drill through corals by secreting chemicals (Glynn, 1997; Glynn & Manzello, 2015) ultimately making corals susceptible to diseases. In Hong Kong reef, rapid progression of tissue loss, white syndrome disease with subsequent coral mortality were reported around the vermetid boreholes (Wong *et al.,* 2016). Precursors for their proliferation is nutrient enrichment and high sedimentation rate in the MPA helping the filter feeders. Additionally, fishing activities can increase gastropod densities in coral reef by reducing their predators (McClanahan 1990), which is prevalent in the present study area (personal observation).

Multifocal, distinct small pink swollen tissue nodules observed on massive genus Porites subjected to infection by the digenetic trematode (parasitic flatworm) larvae, occur chronically on reefs year-round (Raymundo *et al.,* 2008; Aeby *et al.,* 2017). The trematode undergoes complex life cycle involving different intermediate host- a molluscan first, Porites as the second host, and coral-feeding fish as the final host (Aeby 2003). Trematodiasis can significantly decrease in colony growth of up to 50% further could weaken coral's ability for space competition on the reef (Aeby 2003; Aeby *et al.,* 2017). Trematodiasis in Porites colonies have been found reported from the Guam, Papua New Guinea, Australia, French Polynesia, Hawaiian Islands (Aeby 2003; Raymundo *et al.,* 2008).

9.4.4. Indian reefs coral diseases

Diseases have gripped the corals of the world ocean, sparing few habitats or species. Generalized major reason is deteriorating environmental health. But at higher resolution, the peril of impacts depicts each habitat characterized by a distinctive set of diseases. As discussed earlier the nearest habitats with respect to the study area (MPA) are Grande Island, Angriya Bank, Gulf of Kachchh and Lakshadweep with a range of 100-800km. The coral beta/gamma diversity and community composition at the other locations is distinct due to substratum, local hydrodynamics and water chemistry. The differences as well reflect in the degree of environmental degradation and aligning coral aliments. Disease prevalence in the present study and the comparison with the other Indian habitats reveal stark differences suggesting local settings to play crucial role and evading ubiquitous disease prevalence (Table 2). Other speculation is that each habitat does not share most of the diseases due to isolated current regimes and non-mixing of water masses. However, the threat of disease spread from the present study area will always be looming on other habitats, while there is hazard vice-a-versa. With the onset of global climatic adversities reducing each coral reef with a percentile of coral cover dead (Hughes *et al.,* 2017; Bellwood *et al.,* 2019), the other Indian habitats are no exception (Raghuraman *et al.,* 2013; De *et al.,* 2017; Thinesh *et al.,* 2017). In a rapidly changing global environment, the consequences of increasing coral disease may be severe, leading to elevated extinction risk and loss of critical reef habitat (Sokolow 2009). Therefore, with our limited knowledge about the diseases aetiology and limited ability to restrict bleaching, great deal of science needs to emerge to aid conservation. In the meanwhile, the only plausible solution to save the corals is restoration of habitats with all possible methods available today (Epstein *et al.*, 2003; Nanajkar *et al.*, 2019).

Table 9.2. Reports on the occurrence of coral disease in different coral reefs in Indian water

Reef	Coral disease	Infected coral species	Reference
Lakshadweep Islands	Pink line syndrome	*Porites lutea*, *Porites solida*	Raghukumar & Raghukumar 1991; Ravindran *et al.*, 2001; Ravindran & Raghukumar, 2002; Jeyabaskaran, 2006; Ravindran & Raghukumar, 2006, Idrees Babu & Suresh Kumar 2016;
	Pink spot	*Porites* sp.	Thangaradjou *et al.*, 2016
	Red plague syndrome	*Porites lutea*, *Montipora informis, Acropora sp., Favia, Favites, Turbinaria, Goniastrea, Goniopora, Diploastrea, Platygyra lamellina*	Jeyabaskaran, 2006
	White band	*Porites lutea*, *Acropora* sp., *Pocillopora* sp., *Goniopora stokesi*, *Goniopora* sp., *Montipora divarigata*, *Montipora informis*	Ravindran *et al.*, 1998; Muley *et al.*, 2000; Jeyabaskaran, 2006; Thangaradjou *et al.*, 2016
	White plague	*Porites* sp.	Jeyabaskaran, 2006; Thangaradjou *et al.*, 2016
	White pox	*Porites* sp.	Thangaradjou *et al.*, 2016
	Black band	*Porites* sp.	Ravindran *et al.*, 1998; Muley *et al.*, 2000;
	Dark band	*Acropora* sp., *Pocillopora damicornis*	Jeyabaskaran, 2006
	Necrosis	*Porites lutea*	Ravindran *et al.*, 1998; Thangaradjou *et al.*, 2016
	Fungal blotch	*Acropora* sp.	Thangaradjou *et al.*, 2016
Andaman and Nicobar Islands	Black band	*Montastrea curta*, *Porites* sp.	Chakkravarthy & Raghunathan, 2011; Ramesh *et al.*, 2014
	Brown band	*Leptoria phrygia*	Chakkravarthy & Raghunathan, 2011
	Pink spot	*Porites* sp.	Chakkravarthy & Raghunathan, 2011

	Pink-line syndrome	*Porites solida*, *Porites* sp.	Chakkravarthy & Raghunathan, 2011; Ramesh *et al.*, 2014
	Yellow band/blotch	*Porites mayeri*	Chakkravarthy & Raghunathan, 2011
	White syndrome	*Acropora hyacynthus*	Chakkravarthy & Raghunathan, 2011
	White band	x	Muley *et al.*, 2000
	White pox	*Porites annae*	Chakkravarthy & Raghunathan, 2011
	White spot	*Porites lutea*, *Montipora* sp., *Echinopora* sp., *Favia* sp., *Heliopora* sp.	
	White line syndrome	*Porites* sp.	Ramesh *et al.*, 2014
	White plague	*Porites* sp.	Ramesh *et al.*, 2014
	Growth anomaly	*Porites* sp.	Ramesh *et al.*, 2014
	Blue spots		Ramesh *et al.*, 2014
	Yellow spot	*Porites* sp.	Ramesh *et al.*, 2014
	Polychaete infestation		Dam Roy *et al.*, 2009
	Fungal infection	*Porites* sp.	Dam Roy *et al.*, 2009
	Necrotic patches	*Porites lutea*, *Porites lichen*, *Porites sp.*, *Montipora tuberculosa*, *Goniopora* sp., *Goniastrea* sp.	Raghukumar & Raghukumar 1991; Ravindran *et al.*, 1999; Jeyabaskaran & Rao, 2007
Gulf of Mannar and Palk Bay	Pink line syndrome	*Porites* sp., *Acropora* sp., *Fungia* sp.	Kumaraguru *et al.*, 2005, Thangaradjou *et al.*, 2016; Ramesh *et al.*, 2019
	Pink Spot	*Porites* sp.	Thinesh *et al.*, 2009; Thinesh *et al.*, 2011; Thangaradjou *et al.*, 2016; Ramesh *et al.*, 2019
	Black band	*Acropora cytherea*, *Acropora* sp., *Favites abdita, F. halicora*, *Pavona decussatus, platygyra* sp., *Montipora digitata*, *Porites* sp., *Pocillopora* sp., *Goniastrea* sp.	Thinesh *et al.*, 2009; Thinesh *et al.*, 2011; Thinesh *et al.*, 2014; Ramesh *et al.*, 2019
	Black spot		Thinesh *et al.*, 2009; Thangaradjou *et al.*, 2016
	White band	*Acropora cytherea*, *Acropora* sp., *Pocillopora* sp., *Montipora digitata*, *Pocillopora* sp., *Favia stelligera*	Thinesh *et al.*, 2009; Thinesh *et al.*, 2011; Thinesh *et al.*, 2014; Thangaradjou *et al.*, 2016; Ramesh *et al.*, 2019
	White plague	*Porites* sp., *Porites solida*	Thinesh *et al.*, 2009; Thinesh *et al.*, 2011;

			Thangaradjou *et al.*, 2016
	White spot/White patch	*Porites* sp., *Porites solida*	Thinesh *et al.*, 2009; Thinesh *et al.*, 2011; Ramesh *et al.*, 2019
	White pox	*Porites* sp.	Thangaradjou *et al.*, 2016
	White line	*Porites lutea*	Ramesh *et al.*, 2019
	Yellow spot		Thinesh *et al.*, 2014
	Yellow band	*Acropora* sp.	Thinesh *et al.*, 2009; Thinesh *et al.*, 2011; Thangaradjou *et al.*, 2016
	Red band	*Turbinaria mesenterina, T. peltata*	Chellaram *et al.*, 2006
	Coral tumour		Thinesh *et al.*, 2009
	Necrosis	*Acropora* sp., *Montipora* sp., *Porites* sp.	Thangaradjou *et al.*, 2016
	Fungal blotch	*Acropora* sp., *Favia* sp., *Favites* sp., *Goniastrea* sp.	Thangaradjou *et al.*, 2016
	Coralline lethal orange disease	Crustose coralline algae	Abey *et al.*, 2015; Ramesh *et al.*, 2019
	Acropora white syndrome	*Acropora* sp.	Thinesh *et al.*, 2017
Gulf of Kachchh	White plague	*Porites* sp.	Kumar *et al.*, 2014
	Ulcerative white spots	*Porites* sp., *Favites* sp.	Kumar *et al.*, 2014
	Yellow spot	*Porites* sp.	Kumar *et al.*, 2014
	Black band	*Porites* sp., *Favia* sp.	Kumar *et al.*, 2014
	Pink spot	*Porites* sp.	Kumar *et al.*, 2014
Malvan Marine Sanctuary	Skeletal tissue growth anomalies	*Turbinaria mesenterina*, *Favites* sp.	Hussain *et al.*, 2016; Present study
	White syndrome	*Porites* sp., *Favites* sp., *Turbinaria mesenterina, Plesiastrea sp.*	Present study
	Tissue necrosis	*Porites* sp., *Favites* sp., *Turbinaria mesenterina, Goniopora sp.*	Present study
	Coral boring mollusc infestation	*Porites* sp., *Plesiastrea sp.*	Present study
	Trematodiasis	*Porites* sp.	Present study

Chapter 10

10. Destruction of corals by recreational diving tourism in the Malvan Marine Sanctuary

10.1. Introduction

The sheer diversity, complexity, and flamboyant nature of organisms in coral reef habitat fascinate and attract millions of tourists and explorers to visit this fragile ecosystem. Coral reef tourism is one of the rapidly expanding business domains, worth US$35.8 billion per year, attracting around 70 million tourists globally, shaping and transforming the local economy, even an entire nation (Lamb *et al.*, 2014; Spalding *et al.*, 2017). In Southeast Asia and the Caribbean region, coral reef ecotourism has gained popularity while supporting the local economy and generating employment (Brander *et al.*, 2007; De Groot *et al.*, 2012; Sarkis *et al.*, 2013).

Nonetheless, there is growing recognition that unmanaged and intensified recreational use of coral reef often results in environmental degradation, coral skeletal breakage, and tissue abrasion due to physical contact with tourists and boats (Hawkins *et al.*, 1999; Guzner *et al.*, 2010; Terrón-Sigler *et al.*, 2016; Flynn and Forrester 2019; Giglio *et al.*, 2020). It has been noted that the physically damaged corals are more vulnerable to diseases, increased predation, poses disadvantage in space competition, and eventually leading to coral mortality (Guzner *et al.*, 2010; Lamb *et al.*, 2014; Giglio *et al.*, 2020), illustrated in Fig. 10.1. Since the last four decades, numerous studies revealed that worldwide recreational activities mediated reef damage poses a significant concern to coral reef managers and conservators (Woodland and Hooper, 1977; Kay and Liddle, 1989; Hawkins and Roberts, 1993; Medio *et al.*, 1997; Hawkins *et al.*, 1999; Jameson *et al.*, 1999; Hasler and Ott, 2008; Chung *et al.*, 2013; Lamb *et al.*, 2014; Roche *et al.*, 2016). Most often, the pressure of unrestricted dive tourism in poorly managed reefs in the tropical developing nations hinders conservation priorities (de Groot and Bush 2010; Lamb *et al.*, 2014). Further, hampering the coral growth rate, competitive success, susceptibility to thermal anomalies, and bleaching recovery (Carilli *et al.*, 2010; Lamb *et al.*, 2014; Roche *et al.*, 2016).

It is a known fact that most of the Indian coral reefs are under severe stress due to anthropogenic pressure and climate change (De *et al.*, 2017; Majumdar *et al.*, 2018; Nanajkar *et al.*, 2019). Massive tourist inflow and recreational diving induced coral damage have been reported from Andaman and Nicobar Island, Lakshadweep, and Gulf of Mannar (George and Jasmin, 2015; Turner *et al.*, 2009; Venkataraman *et al.*, 2012; Majumdar *et al.*, 2018). Therefore, it is imperative to assess the impact of dive tourism on coral reef health to assist in reef management strategies and conservation policies in developing countries like India.

Densely populated coastal regions in the developing country often ensue a sense of intensified social interaction where natural resource supply is short, and demand is more. In such cases, resource over-exploitation takes place due to competition rather than resource utility. We analyzed this socio-environmental interface in a marine protected area on the West coast of India, where a newly discovered profession of recreational dive tourism has triggered a situation that preliminarily depicts the classical case of '*Tragedy of the commons*' (Hardin, 1968), but desolately reveals more deleterious consequence when we investigate qualitative socio-economy. The present study aimed to investigate the diving impact on coral health at the MPA while considering the socio-behavioral approach of tourists and local stakeholders pertaining to recreational diving. For the first time, this study evaluates video logs in the public domain providing visual proof for our inferences. These aspects were analyzed further from the perspective of the 'public goods' game and resource sustainability perspective.

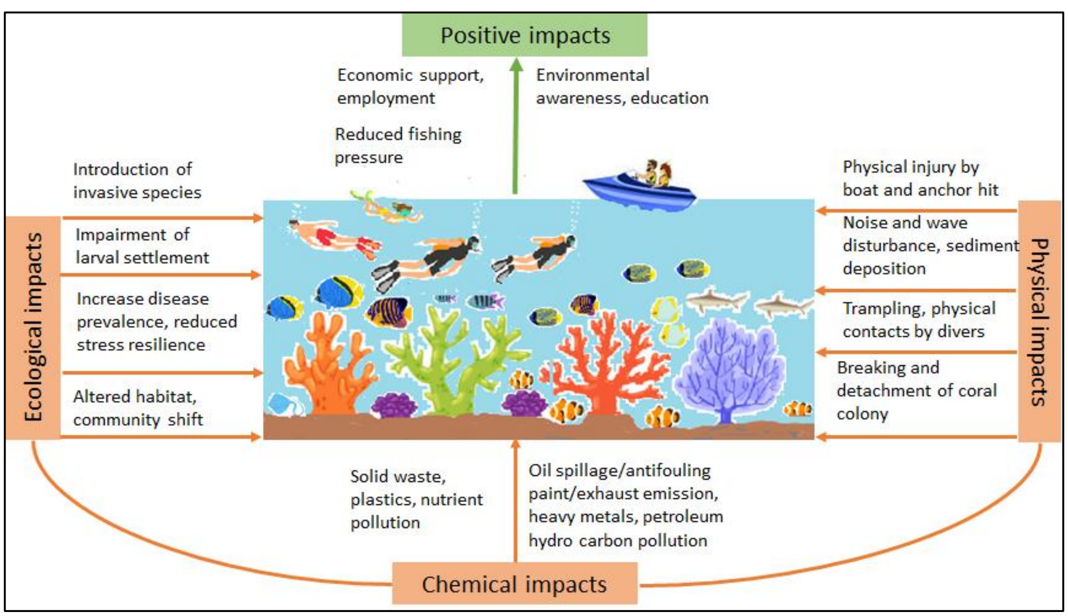

Fig. 10.1. Graphical representation of potential impacts of recreational activities includes SCUBA diving, snorkeling, boating on the coral reef habitat.

10.2. Material and Method

10.2.1. Study area

The Malvan Marine Sanctuary, located on the Central West Coast of India, spreads over 29.122 km^2 area. Recreational and diving activities were restricted from October to May each year due to fair weather conditions. According to month-wise tourist arrivals in Malvan, December and January are the peak months for foreign and domestic tourists' influx. Tourist footfall in the Sindhudurg district is estimated to be 200,000 in 2001-2002 and projected to be 641,427 in 2021-22, mostly visiting Malvan, with an overall annual increase of 6.00% (Munjal, 2019). Local coastal communities, who were earlier involved in fishing, are now involved in tourism business (around 3000), the estimated annual revenue from tourism is about $2.5 million (UNDP 2011).

Four patch reef sites (T1, T2, T3, and T4) were selected for the present study, based on daily recreational diving and snorkeling activities. Two reef sites (T2 and T3) were chosen near the famous Sindhudurg Island (Sindhudurg fort), where most of the tourists were taken for SCUBA activity by the local diving operators and designated as the high diving intensity (HDI) sites. Another two patch reef sites (T1 and T4) were selected about one Kilometer away from the intensive diving and snorkeling sites, where the diving activity is low due to wave action and more depth, designated as low diving intensity (LDI) sites. The study site T2 and T3 (depth 2-4m) are located on the leeward side of the Sindhudurg Island (50-100m away from the shore) and subjected to massive tourist influx. Whereas T1 is located on the Southern side of the Island (~1km away) offshore, T4 is situated ~2km north of the Island (Fig. 10.2).

Coral reef survey design

During a pilot study on coral bleaching in 2016, we observed numerous physically damaged corals. To investigate the reason for the coral physical damage in the MPA, we conducted subsequent underwater surveys at four reef sites (T1-T4, Fig. 10.2) during May 2016 aftermath of mass coral bleaching events. The following field surveys were carried out in the fair-weather months of May 2017, October 2018, and April 2019, consecutively. During each survey, three replicate belt transects (20m X 2m) were placed at each site of the sub-tidal reef by two divers (total 12 transects at four sites each year), and transects were laid approximately 10m away from each other. Each belt transects covered 40m^2 area and a total of 120m^2 area per survey site. GPS locations of the sites and coordinates of the transect endpoint were recorded using a handheld GPS device. The next consecutive annual surveys were conducted following the same GPS coordinates to monitor the same patch reefs. Assessment of coral damage by tourism activity was designed based on Lamb *et al.*, (2014). Headcount of all the live coral colonies and physically damaged colonies were carried out along each transect. The number of physically damaged colonies (breakage, abrasion, tissue damage, and up-rooted coral colony) were counted to estimate the prevalence of impact by tourist activities (percent coral colonies per transect with visible physical damage).

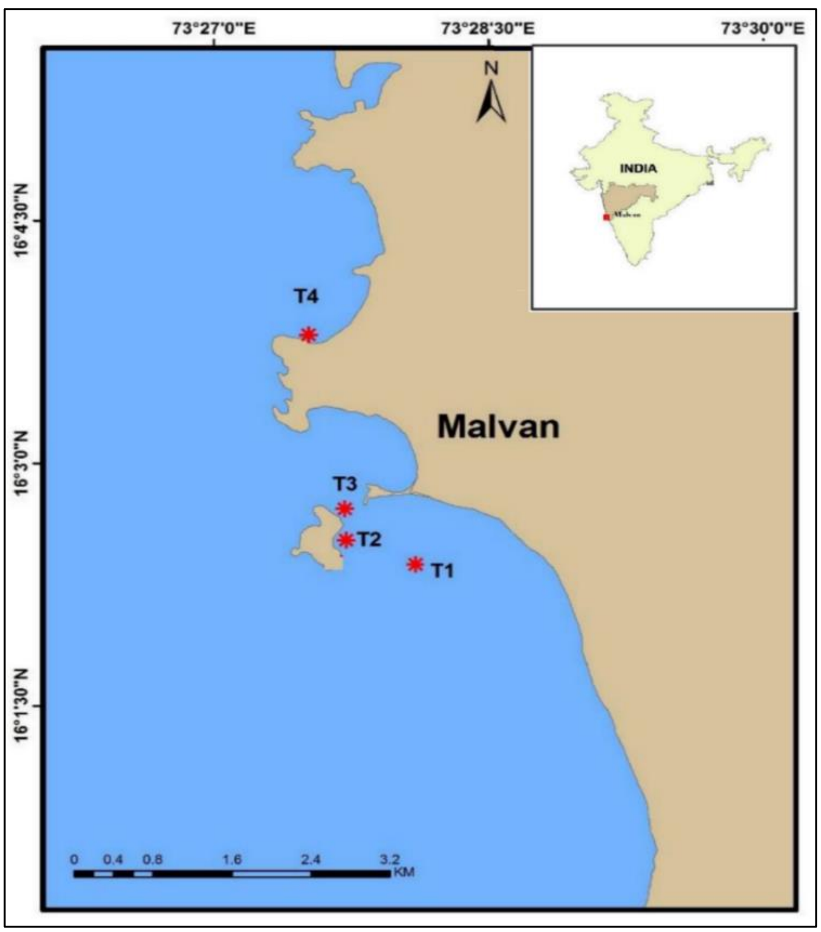

Fig. 10.2: Location of the study sites at the Malvan Marine Sanctuary

Observation of diver's behaviour

In the absence of long-term ecological datasets on the impact of SCUBA diving in the MPA, we considered unconventional data sources such as underwater video logs uploaded on the open-access video-sharing platform www.youtube.com by dive operators and individual tourists for analysis of the divers' and dive operators behavior and attitude towards the marine life during diving. In recent years, the application of YouTube videos in terrestrial wildlife conservation studies is relatively new (Harrington *et al.*, 2019; Rebolo-Ifrán *et al.*, 2019). However, the use of YouTube videos to study the health status of coral reef and tourism impacts is novel. To navigate relevant videos of the MPA, preliminary search of videos was carried out by using English language search keywords, e.g., tourism+travel+SCUBA+diving+snorkel+coral+reef+underwater+Sindhudurg+Malvan+Tarkarli+Devbag+cheivla+sarjekote. Videos qualified the preliminary search categories, were further scrutinized for confirming the locations (in this case MPA) by carefully observing the surroundings like landmarks, Sindhudurg Island (Fort), bottom substrate, coral and fish species assemblage, boat name, number, and local language. Only original videos were considered for this study, and redundant postings were dropped. All the videos were downloaded and analyzed in October 2019.

10.2.2. Quantification of divers and snorkelers

An estimate of annual SCUBA divers and snorkelers' at each diving site in the MPA was carried out following Jameson *et al.*, (1999), wherein authors conducted 2-3 days long limited monitoring at each diving site. Whereas in the present study, an observation was conducted for four days employing a similar methodology (Table 2). To estimate the annual number of SCUBA divers or diver per year (DPY), and snorkelers per year (SPY) at each diving study site, a daily headcount of SCUBA divers and snorkelers were conducted during 6-7 October 2018, and 16-18 April 2019 between 09:00 AM to16:00 PM. DPY and SPY also include the number of diving crew underwater. The number of tourist boats and anchors used on the reef were also counted at the diving sites. To avoid the bias in tourist footfall numbers between working days and weekend holidays, in 2018, the survey was conducted on weekends, and in 2019, the survey was conducted during weekdays.

10.2.3. Data analysis

All the coral colonies were counted on each transect. The prevalence of physically damaged coral colonies was calculated by dividing the number of impacted colonies (damaged) at each transect with the total number of colonies at that transect and expressed as a percentage. At each location, three transects (n=3) were averaged, and mean values were considered for analysis. Further, the Coral Damage Index (CDI) was calculated at each diving site, following the methodology of Jameson *et al.*, (1999), i.e., a crucial cutoff value of 4% was considered for broken and damaged coral colonies. All data are reported as means ± standard error (SE).

For analysis of SCUBA drivers' underwater behavior, we found a total of 126 video logs on the broadcasting website- YouTube. Videos were analyzed for three categories; 1) divers' body contact (touching/holding corals to control bouncy and due to fin movement), 2) feeding of reef fishes, and 3) chasing of reef fishes.

Assessment of the number of tourist boats, anchors, divers, and snorkelers was calculated based on the daily observation data at the diving sites. DPY and SPY were estimated by multiplying the mean of the divers and snorkelers with the total number of SCUBA operation days (approximately 240 days) in a year.

The mean percentage data were normalized by arcsine square-root transformation. The impact of diving and snorkeling on coral physical damage was analyzed by one-factor and two-factor analysis of variance (ANOVA). Pearson correlation was calculated by conducting a paired two-sample Student's t-test with $\alpha = 0.05$ to detect a correlation between coral physical damage and the annual mean diver and snorkeler numbers.

10.3. Result

10.3.1. Prevalence of physical injury on corals

During The prevalence of physical injury includes breakage and abrasion of coral colonies were more prevalent at HDI sites compared to LDI sites throughout the study period. An increasing annual trend in mean coral physical injury was observed at the HDI sites. Mean physical injury in 2016 was recorded 8.19%±0.59SE at the HDI sites, which increased to 13.75%±2.16SE in 2017. Damaged colonies rose to 15.06%±1.6SE in 2018. During our survey in 2019, damaged coral colonies recorded to be 19.79%±0.94SE. The relative coral physical injuries at LDI sites were also found to be increased, estimated to be 1.47%±0.05SE during 2016. In 2017, physically injured colonies recorded to be 1.69%±0.01SE. Whereas in 2018 and 2019, coral damaged recorded to be 2.83%±0.48SE and 3.36±0.66SE, respectively.

Based on annual diver and snorkelers number estimated in 2018-2019, annual mean coral physical damage in all sites was significant with the annual mean diver number (ANOVA $p<0.05$; SS=81.94; df=1; F=8.5), and also with the mean snorkeler number (ANOVA $p<0.05$; SS=90.57; df=1; F=8.9),. Mean damaged coral colonies differed significantly between HDI and LDI sites (ANOVA $p<0.01$). The Pearson correlation coefficient of 0.99 indicates a positive relationship between the mean coral physical damage and the annual mean diver and snorkeler number.

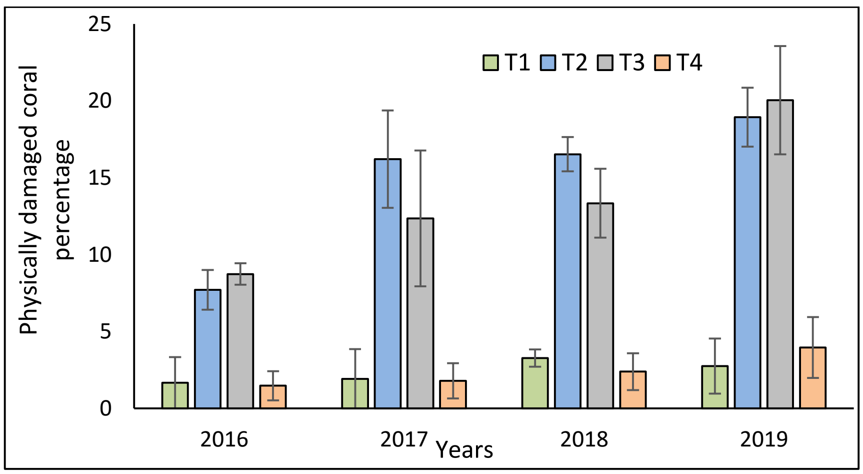

Fig. 10.3. Prevalence of coral physical injury (Mean±SE; broken and tissue damaged) during the study period.

At HDI sites T2 and T3, corals experienced severe physical damage, the prevalence of physical injury was recorded to be 7.63%, and 8.86% during 2016, 15.91%, and 11.59% during 2017, 16.67%, and 13.46% during 2018, and 18.85%, and 20.73% during 2019, respectively. Whereas at LDI sites (T4), physical injury was documented comparatively lower. The prevalence of physical injury was recorded to be 1.42% in 2016, 1.69% in 2017, 2.35% in 2018, and 4.03% in 2019. Whereas at T1 prevalence of physical injury was documented to be 1.52% in 2016, 1.68% in 2017, 3.31% during 2018, and 2.7% during 2019 (Fig. 10.3).

Coral damage index (CDI) (Jameson *et al.,* 1999) value exceeded the 4% ceiling value at both of the HDI sites throughout the four years of survey (at T2, CDI=7.63% in 2016; CDI=15.91% in 2017, CDI=16.67% in 2018 and at T3, CDI=8.76% in 2016; CDI=11.59% in 2017, CDI=13.46% in 2018) ; highest CDI value was observed in 2019 (CDI= 18.85% and 20.73% at T2 and T3 respectively). On the other hand, CDI value only crosses 4% cut off only during 2019 at the LDI T4 (CDI=4.03%), where the diving practice started in 2018.

10.3.2. Videos logs analysis

YouTube video analysis for the SCUBA divers' behavior showed that in 98.41% (n=124) of the video clips, the SCUBA divers had body contact and fin contact with corals. Other activities include touching, holding, sitting, standing, and walking on the coral colonies for maintaining underwater buoyancy and posing for photos/videos and eventually breaking corals. Nonetheless, another detrimental practice observed was reef fish feeding (generally bread or rice) by tourists and diving operators to attract fish around the tourists. Fish aggregations were mostly comprised of the sergeant fish (*Abudefduf sordidus* and *A. bengalensis),* Demiselles *(Neopomacentrus violascens, N. cyanomos),* Damsel*s (Cheiloprion labiatus),* and Moorish idol (*Zanclus cornutus*). Fish feeding practice was observed in 73.80% (n=93) of the videos. Further, in 19.04% (n=24) of the videos, divers were found to chase different species of reef fishes for photos/videos.

Additionally, almost 97.62% (n=123) of the videos showed visibly damaged coral, which includes skeletal breakage, abrasion, tissue damage, growth anomaly, bleaching, macroalgal, algal turf and boring sponge growth on coral. The most common behavior of divers was found to hold the *T. mesenterina* colony for balancing underwater buoyancy and capturing photos and videos. Video logs further showed that diving instructors encouraged tourists to hold the coral colonies to fix their position for photos and videos. Divers were also seen to walking, crawling, sitting, lying on coral colonies.

10.3.3. Quantification of boats, divers and snorkelers

Recreational activities in the MPA restricted to fair-weather months from October to May (n=240 days). At the HDI site T2, mean number of dive boat visiting per day estimated as 20.5±1.32SE, mean number of anchors were found to be 39.5±2.5SE, mean divers were estimated to be 174.75±15.59SE, mean number of snorkelers were calculated to be 181±12.02SE. At HDI site T3, mean number of boats, anchors, divers, snorkelers were found to be 17.5±1.04SE, 33.5±2.75SE, 154.75±24.19SE, and 172.25±13.60SE, respectively (Fig. 10.4). At T1, no tourist activity was observed. Whereas at T4, a newly started diving site, mean of boats, anchors, divers, snorkelers recorded 5.00±0.57SE, 10.00±1.15SE, 31.25±4.87SE, 33.75±5.35SE, respectively. This study revealed that weekend holidays experienced a higher number of tourist arrival compared to working days. Divers per year (DPY) and snorkelers per year (SPY) at T2 were calculated to be 41940 individuals, and 43440 individuals, respectively. At T3, DPY was accounted to be 37140 individuals, and SPY was estimated to be 41340 individuals. Whereas at T4, DPY was estimated to be 7500 individuals, and SPY was 8100 individuals (Table 10.2).

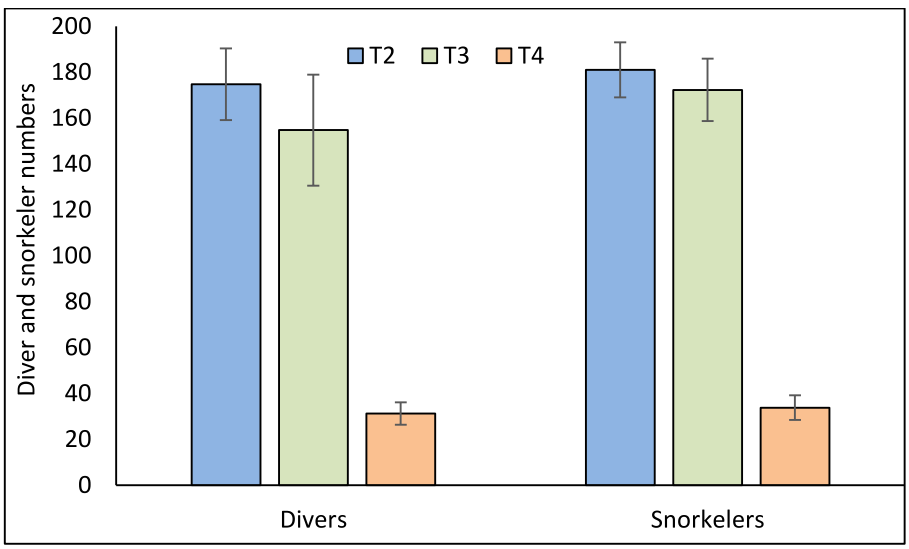

Fig. 10.4. Average number of divers and snorkelers per day at the HDI sites (T2 and T3) and LDI site (T4).

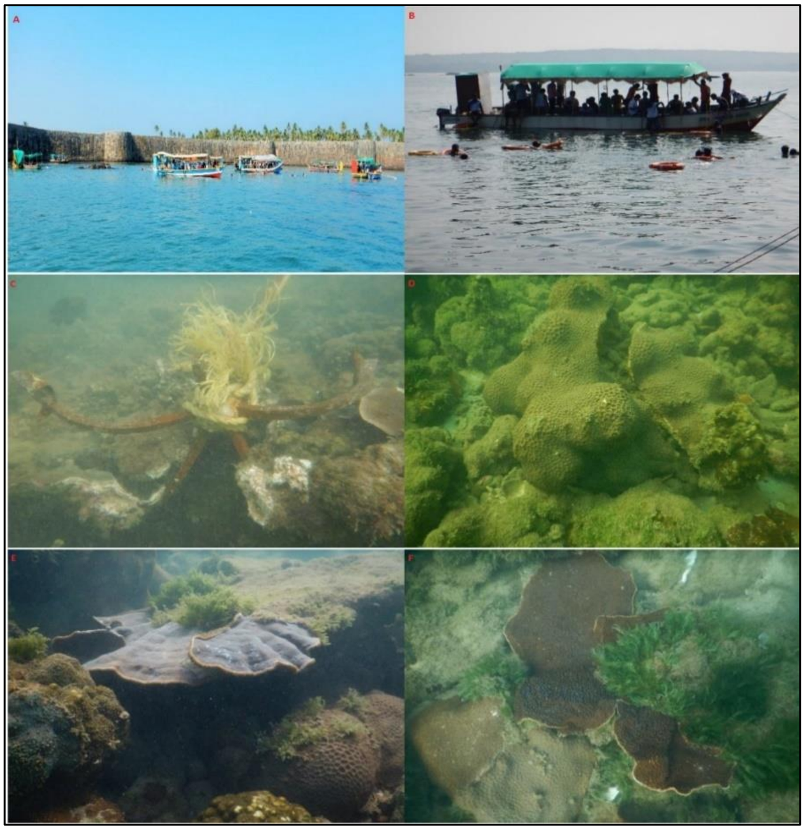

Fig. 10.5. Underwater photographic evidence of recreational diving and snorkeling caused coral damages in the MPA; A) assemblages of diving boats at shallow (2-4m) the diving site (11 boats in the frame); B) divers and snorkelers on a diving boat, total 35 people on the boat including crew member; C) boat anchor mediated coral damage; D) broken *Favites* sp. by boat anchor; E) up-rooted and overturned colony of *T. mesenterina*; F) settlement and growth of *Caulerpa* sp. on broken *T. mesenterina* colony.

Table 10.1. Estimation of daily number (mean±SE) of boats, anchors, divers/estimated dives per year (DPY), snorkelers/estimated snorkelers per year (SPY) at study sites following Jameson *et al.*, (1999).

Dive sites/Date	Boats	Anchors	Divers	Snorkelers/SPY
T2				
6-Oct-18	17	34	133	158
7-Oct-18	20	40	175	197
16-Apr-19	22	38	183	163
17-Apr-19	23	46	208	206
Mean±SE	20.5±1.32	39.5±2.5	174.75±15.59	181±12.02

DPY and SPY			41940	43440
T3				
6-Oct-18	18	36	131	146
7-Oct-18	15	30	107	153
16-Apr-19	17	28	162	187
17-Apr-19	20	40	219	203
Mean±SE	17.5±1.04	33.5±2.75	154.75±24.19	172.25±13.60
DPY and SPY			37140	41340
T4				
13-Oct-18	4	8	23	36
14-Oct-18	4	8	31	20
18-Apr-19	6	12	26	33
21-Apr-19	6	12	45	46
Mean±SE	5±0.57	10±1.15	31.25±4.87	33.75±5.35
DPY and SPY			7500	8100

Fig. 10.6 A-F) A few snapshots taken from the videos analysed for the diver behavior at the MPA from https://www.youtube.com (Please see Table 10.3 for the links)

10.4. Discussion

With the progress in developing nations and the elevated financial status, the populace of many countries like India indulged in adventure tourism. Much of the elite goes abroad to exotic locations while middle to lower-middle-class ventures into tourist places within the country. In the recent past, Malvan (study area) has emerged as a budget tourist destination and has experienced sudden bloom (UNDP 2011; De *et al.,* 2015, 2017) for its coasts, cuisine, and pilgrimage in the nearby areas, while recreational diving being a recent addition. The price of diving tourism at the MPA varies at different diving shops and ranges from Rs. 500-1000 or approx. 8-14USD per person, which is far cheaper than the other diving destination in India like Goa, Lakshadweep, and Andaman and Nicobar Islands. The last decade experienced flourishing of this adventurous activity with the locals realizing its potential as an alternative to hardship borne artisanal fishery (UNDP 2011). However, our quantitative monitoring transects at three diving and control sites, qualitative analysis of video log broadcasting website (www.youtube.com), and direct quantification of plying boats and number of diving and snorkeling tourists' presents evidential testimony of detrimental activities resulting in coral damage. The present study exclusively analyses the socially evident parameters alongside supported by ground-truthing surveys of corals, disproving the counter-intuitive hypothesis' ecotourism aids conservation.' The resulting analysis clearly implies that coral damage and associated mortality is the direct function of recreational diving also reported elsewhere (Giglio *et al.,* 2016, 2020). Behavioral analysis suggests that tourists participate in these adventures for several reasons other than the affection for marine life, and the most probable reason is social media. Since the video log behavior showed that most of them carry less admiration for oceanic organisms and instead are eventful with camera-centric activities. Hence, the ignorance towards this diving footprint on marine life reveals a significant conservation concern (reviewed in Giglio *et al.,* 2020).

Video log analysis showed that diving operations carried out in the MPA neither follow the basic safety norms or standard SCUBA diving practices, nor the ethics of responsible diving. This analysis also raises questions on the legitimacy of the destructive diving practice in the MPA, wherein corals are protected as Schedule-I species under the Wildlife Protection Act (1972) of India. While the forest department and state biodiversity board, along with the tourism department, are striving hard for conservation of the MPA, but the habitat has been victimized by overwhelming tourism resulting in implementation letdown.

Physical damage to corals (Fig. 10.5) is well documented globally due to direct anchorage on corals, tourist standing, walking, holding the coral for support, fin movement and jumping on shallow-water corals (Hawkins and Roberts 1993; Hawkins *et al.,* 1999; Barker and Roberts 2004; Hasler and Ott 2008; Juhasz *et al.,* 2010; Krieger and Chadwick 2013; Chung *et al.,* 2013; Lamb *et al.,* 2014; Roche *et al.,* 2016). The present study categorically documents all activities mentioned above at the MPA, wherein the most damaging activity was identified to be diver's physical contact with corals. The video log analysis has confirmed that the inexperienced tourists/divers hold coral colonies to counter underwater bouncy, which often leads to detachment and topples the entire coral colony. The prevalence of coral physical damage has increased from 4.83% in 2016 to 11.58% in 2019. Therefore, the cumulative physical damage estimated to be 33.08% in the last four years in the MPA.

Another noticeable detrimental behavior was reef fish feeding by dive operators in this MPA (Fig. 10.6F). Fish feeding practice by divers has reportedly increased worldwide, which could result in trophic alteration, malnutrition, and catalyze reef degradation (Hémery and McClanahan 2007; Brookhouse *et al.,* 2013; Paula *et al.,* 2018). Moreover, coral reef recreational activity associated damages include direct impacts by boat hull, anchorage (Fig. 10.5) and propeller collision with corals and fishes, (Rogers and Garrison 2001; Jones 2007) wherein the damage of reef bottom could take more than a decade for recovery (Rogers and Garrison 2001). Motorboat noise can also affect fish physiology, individual fitness, behavior, and population in the coral reef environment (Simpson *et al.,* 2016; McCormick *et al.,* 2018). Boat generated waves, chemical pollutants such as heavy metal, petroleum hydrocarbons, and antifouling paints further impact coral habitat negatively (Whitfield and Becker 2014). Furthermore, boats and divers increase sediment resuspension, wastewater, nutrient discharge, and shoreline erosion (Hawkins *et al.,* 1999; Lamb *et al.,* 2014).

Estimation of carrying capacity for reef diving tourism is an important measure to evaluate how much diving pressure one reef can withstand without compromising reef health and species diversity over time; therefore, setting the diving limit is necessary for long-term sustenance of coral reef (Wafar 1997; Jameson *et al.,* 1999). Annual carrying capacity has been measured, and diving limits have been implemented in different reefs across the world (Table 3). However, in none of the Indian reefs, estimates of diving carrying capacity have been determined so far, and at the present MPA, carrying capacity exceeds many folds of the global average. The cumulative effects of several parameters indicate high Coral Damage Index (CDI) value due to remorseless diving activities and bleaching events during 2014-2016 due to sea surface temperature (SST) anomaly (De *et al.,* 2015; Raj *et al.*, 2018). Noting the recent worldwide coral reef decline triggered by recurrent mass bleaching events (Hughes *et al.,* 2018; Eakin *et al.,* 2019), recreational tourism mediated reef degradation will severely hinder the reef recovery processes (Giglio *et al.,* 2020). Declining of reef ecosystem may have a wide range of possible effects on coastal human communities, including reduced food, income, and longer-term effects such as increased vulnerability to extreme weather events (Pascal *et al.,* 2016; Anthony 2016).

Table 10.2. Carrying capacity estimated for sustainable recreational diving

Estimated of scuba diver carrying capacity (individuals)	Reef location	References
4000-6000	Bonaire	Dixon *et al.,* 1993, 1994
500	US Virgin Islands	Chadwick-Furman, 1996
Up to 5,000	Eastern Australia	Harriott *et al.,* 1997
5000-6000	Egypt, Bonaire, and Saba	Hawkins & Roberts 1997, Hawkins *et al.,* 1999
5000-6000	Eilat, Israel	Zakai & Chadwick-Furman, 2002
Up to 7000	Sodwana Bay, South Africa	Schleyer & Tomalin, 2000

Considering the present cumulative coral damage trend (Fig. 10.3), it can be predicted that the entire habitat will be altered and converted to a *non*-coral ecosystem in the coming years. The only possibility that the corals may survive beyond the mortality trend will be due to following reasons; 1) Unaffected reminiscent polyps with the ability to revive the colony; 2) Remaining unknown remote patches of corals that will sustain the local gene pool; 3) Larval flux from nearby habitats on the west coast; 4) Halting the deleterious ongoing activities with immediate effects.

The ignorance of tourists is not the primary cause of concern (considering the socio-economic status of the tourist-they are presumed oblivious), but the governance is in question, and more importantly, the accountability of local stakeholders for carrying out this activity unprofessionally. The stakeholders completely fail to realize the long-term loss behind this short-term monitory gain. The long-term loss will affect local stakeholders the most rather than the tourists or administration. This is a characteristic case of nonexcludable common resource mismanaged, considering it to be '*private goods*' resulting in a '*tragedy*' (Dietz *et al.,* 2003). Marine resource mismanagement has resulted in collapse (McWhinnie 2009), with a significant change in phenology (Enquist *et al.,* 2014; Wabnitz *et al.,* 2018), and in many cases bringing local extinction (Wabnitz *et al.,* 2018). From commercial fisheries (McWhinnie 2009), ecosystem services (Lant *et al.,* 2008) to reef tourism (Hawkins and Roberts 1992, 1993) all have resulted in widescale negative impacts primarily driven by a game skewed by selfish short-term gains (Hardin 1968).

Hardin (1968) explained that common property could be used by society depending on the quality and quantity of property and funds received in that proportion sustainably. Nevertheless, in an unmanaged situation (e.g., present study), it has open access for exploitation and attains a critical point of collapse, resulting in a '*tragedy of the commons'* wherein deterministic escalation ensues between the selfish users in which they are both the villains and the victims (Berkes 1985). In the present case, the financial stability attained due to the recreational diving business will not keep the stakeholders underprivileged while allowing them to switch to new businesses or employments when the coral-based income ceases. Thus, it is not a classical '*tragedy' of the commons*' (Hardin 1968) for the stakeholders (commons) as they go unharmed after the resource is consumed (in the present case destruction of corals). The stakeholders are not the victims and can avert the '*tragedy'* as they will have alternative professions, making them '*not so commons*.' On the other hand, it turns out to be damaging for the coral community, bringing them to the brink of localized extinction.

In many cases, social media has been pivotal in the conservation of nature and wildlife (Wu *et al.,* 2018), but ironically in the present case, it has been the culprit. Finally, a coral habitat in an MPA will be sacrificed for the financial upbringing of this coastal community; however, a sense of accountability can still keep the habitat alive. Damaging activities have to be halted in the immediate future by management intervention; otherwise, it will inevitably cease due to the non-existential status of corals. The currently unsustainable model of resource damage rather than optimum utilization requires reframing of conservation priorities with structured regulations aiming at sustainability (Bellwood *et al.*, 2004; Mcleod *et al.*, 2019).

Several cases have shown successful self-organized management of community-based public goods (Ostrom *et al.*, 1999; Dietz *et al.*, 2003) with equitable benefit sharing with a '*win-win*' co-operative game (Luttinger 1997). Educating locals have shown promising results with a positive outcome of the reiterated public goods game (Kobori *et al.*, 2016). Similarly, proper briefings and close supervision by the diving operators on tourists could reduce diver's impact on the reefs (Barker and Roberts 2004; Krieger and Chadwick 2013; Giglio *et al.*, 2020). These results are alarming for the habitat and immediately demands estimation of carrying capacity for setting policies to improve tourism and enhance conservation; furthermore, it requires restoration measures (Horoszowski-Fridman *et al.*, 2015; Nanajkar *et al.*, 2019) to compensate the presently damaged coral community and to buffer them from other global stressors. India, as a developing country, is planning to declare new MPAs; however, pronouncement without legally binding implementations ultimately will not serve the purpose. This MPA declared 42 years back, might not remain a coral habitat if there are no immediate actions taken to counter the local stressors in an era of rapid climate change, and we hope that the present study serves positively directional.

Table 10.3. List of YouTube videos (URL links) analysed to document the diver's behaviour in the MMS ('✓' denotes observed behaviour and '- 'denotes absence)

Sr. no	YouTube URL (accessed on 01/11/2019)	Diver body contact	Chasing of fish	Feeding fish	Visible coral damage
1	https://www.youtube.com/watch?v=WRJ17EtzCC0	✓	-	✓	✓
2	https://www.youtube.com/watch?v=z7nMQaYZXVI	✓	-	✓	✓
3	https://www.youtube.com/watch?v=tLjfO5cmCK0	✓	-	✓	✓
4	https://www.youtube.com/watch?v=yrYY3d-VFb4	✓	✓	✓	✓
5	https://www.youtube.com/watch?v=ALkaIhyiTjA	✓	-	✓	✓
6	https://www.youtube.com/watch?v=goU2KBFbf2k	✓	✓	✓	✓

7	https://www.youtube.com/watch?v=iOiELSyI3QA	✓	✓	✓	✓
8	https://www.youtube.com/watch?v=i2tzLSJe6fI	✓	-	✓	✓
9	https://www.youtube.com/watch?v=6Z4Qg_fNo0Q	✓	-	✓	✓
10	https://www.youtube.com/watch?v=c8sfo42T4DI	✓	✓	✓	✓
11	https://www.youtube.com/watch?v=_fTzYUsL_DQ	✓	✓	✓	✓
12	https://www.youtube.com/watch?v=abBZQuZheCM	✓	✓	✓	✓
13	https://www.youtube.com/watch?v=zs9ZrsCGbuc	✓	✓	✓	✓
14	https://www.youtube.com/watch?v=dnhDjYwsQys	✓	-	✓	✓
15	https://www.youtube.com/watch?v=jK8oSQCwzzI	-	-	✓	✓
16	https://www.youtube.com/watch?v=57isitutHNg	✓	-	✓	✓
17	https://www.youtube.com/watch?v=57isitutHNg	✓	-	✓	✓
18	https://www.youtube.com/watch?v=mH-NxyE1LUE	✓	✓	✓	✓
19	https://www.youtube.com/watch?v=_bfoDTBLRE4	✓	-	✓	✓
20	https://www.youtube.com/watch?v=3YJeIfUgIfk	✓	✓	✓	✓
21	https://www.youtube.com/watch?v=Zy-1ism3Et4	✓	✓	✓	✓
22	https://www.youtube.com/watch?v=Kelyqy--Kvg	✓	-	✓	✓
23	https://www.youtube.com/watch?v=RgZB8i2oSZY	✓	-	✓	✓
24	https://www.youtube.com/watch?v=9TGIZUClfYM	✓	-	✓	✓
25	https://www.youtube.com/watch?v=Nq_3jYe2xFM	✓	-	✓	✓
26	https://www.youtube.com/watch?v=UFOE2nKTCzg	✓	-	✓	✓
27	https://www.youtube.com/watch?v=_-szA63L44c	✓	-	✓	✓
28	https://www.youtube.com/watch?v=cCVQrJk46Ls	✓	-	✓	✓
29	https://www.youtube.com/watch?v=KKo2A_VXYDE	✓	-	✓	✓
30	https://www.youtube.com/watch?v=BPidHQShKKg	✓	-	✓	✓
31	https://www.youtube.com/watch?v=Zy-1ism3Et4	✓	-	✓	✓
32	https://www.youtube.com/watch?v=HiXenhDH7HU	✓	-	-	✓
33	https://www.youtube.com/watch?v=VU6I1iGk7So	✓	-	-	✓

34	https://www.youtube.com/watch?v=JfGyOxm_ags	✓	-	✓	✓
35	https://www.youtube.com/watch?v=CpDQ2zYbBFU	✓	-	✓	✓
36	https://www.youtube.com/watch?v=wSM606FAOAo	✓	-	✓	✓
37	https://www.youtube.com/watch?v=fwWhNnszauI	✓	-	-	✓
38	https://www.youtube.com/watch?v=ciTMKViruPw	✓	-	-	✓
39	https://www.youtube.com/watch?v=2e3k7f9WaEs	✓	-	✓	✓
40	https://www.youtube.com/watch?v=XMSy3rnBhGI	✓	-	✓	✓
41	https://www.youtube.com/watch?v=YDlGDcUmAN0	-	-	-	✓
42	https://www.youtube.com/watch?v=68m4XkFEIos	✓	-	-	✓
43	https://www.youtube.com/watch?v=XCLRhjb28Z0	✓	-	✓	✓
44	https://www.youtube.com/watch?v=G8IYpxTozO8	✓	-	✓	✓
45	https://www.youtube.com/watch?v=dVc98R7IkHk	✓	-	-	✓
46	https://www.youtube.com/watch?v=VFQjJ4Ve3Ms	✓	✓	✓	✓
47	https://www.youtube.com/watch?v=VFQjJ4Ve3Ms	✓	-	✓	✓
48	https://www.youtube.com/watch?v=7kZKtoivFdg	✓	-	✓	✓
49	https://www.youtube.com/watch?v=JjJibW7IUw4	✓	-	-	✓
50	https://www.youtube.com/watch?v=XwpoVT26KpE	✓	-	✓	✓
51	https://www.youtube.com/watch?v=9iXeYMaRLyw	✓	-	✓	✓
52	https://www.youtube.com/watch?v=V1SzAQpjYKU	✓	-	-	-
53	https://www.youtube.com/watch?v=g-nskJivkjk	✓	-	-	-
54	https://www.youtube.com/watch?v=wULGcdiJND4	✓	-	✓	✓
55	https://www.youtube.com/watch?v=DKvW8EjHiRk	✓	-	✓	✓
56	https://www.youtube.com/watch?v=_AFpd2_P8Y8	✓	-	-	✓
57	https://www.youtube.com/watch?v=2wMbuyUM4nU	✓	-	✓	✓
58	https://www.youtube.com/watch?v=boP6Q2o6ntw	✓	-	✓	✓
59	https://www.youtube.com/watch?v=Ic6oKMmX_x8	✓	-	✓	✓
60	https://www.youtube.com/watch?v=_FjXXiBp7Hk	✓	✓	✓	✓

61	https://www.youtube.com/watch?v=fEPDZc8Tzgw	✓	✓	✓	✓
62	https://www.youtube.com/watch?v=grIGgCirUA0	✓	✓	✓	✓
63	https://www.youtube.com/watch?v=G_3bReANRlQ	✓	-	✓	✓
64	https://www.youtube.com/watch?v=vGsraHwCvZA	✓	-	✓	✓
65	https://www.youtube.com/watch?v=CdvjscmElU4	✓	-	✓	✓
66	https://www.youtube.com/watch?v=c8sfo42T4DI	✓	-	✓	✓
67	https://www.youtube.com/watch?v=IvCHY5jlyYY	✓	✓	-	✓
68	https://www.youtube.com/watch?v=MLGl4g5F0qc	✓	-	✓	✓
69	https://www.youtube.com/watch?v=RmoxsSZGxlc	✓	-	-	✓
70	https://www.youtube.com/watch?v=-ubU2DQm5Jk	✓	-	✓	✓
71	https://www.youtube.com/watch?v=e9WBjKO8-Bg	✓	-	-	✓
72	https://www.youtube.com/watch?v=HJapWW8nYt8	✓	-	✓	✓
73	https://www.youtube.com/watch?v=YygdUnXdhHw	✓	-	✓	✓
74	https://www.youtube.com/watch?v=3PLFFkGR0EM	✓	-	-	✓
75	https://www.youtube.com/watch?v=qpDRI3Jqp-0	✓	✓	✓	✓
76	https://www.youtube.com/watch?v=JUKNVXp1LWs	✓	✓	✓	✓
77	https://www.youtube.com/watch?v=eA9pfPoJZmk	✓	✓	✓	✓
78	https://www.youtube.com/watch?v=oWS0T91gAU4	✓	-	✓	✓
79	https://www.youtube.com/watch?v=ch4NMxzmbEs	✓	✓	-	✓
80	https://www.youtube.com/watch?v=OVyilmJzj1g	✓	✓	✓	✓
81	https://www.youtube.com/watch?v=xDr9jpo-wUY	✓	✓	-	✓
82	https://www.youtube.com/watch?v=cfP-jjXljNc	✓	-	-	✓
83	https://www.youtube.com/watch?v=4aiZ_b32wDc	✓	-	-	✓
84	https://www.youtube.com/watch?v=0g9LEOvQbwI	✓	-	✓	✓
85	https://www.youtube.com/watch?v=tMy0EA8MeO8	✓	-	-	✓
86	https://www.youtube.com/watch?v=X8rAmjF8HLI	✓	-	-	✓
87	https://www.youtube.com/watch?v=dkr2x95aP9Y	✓	-	✓	✓

88	https://www.youtube.com/watch?v=luuBG3Lev-E	✓	-	✓	✓
89	https://www.youtube.com/watch?v=8980okE_grw	✓	-	✓	✓
90	https://www.youtube.com/watch?v=QipPsewSAyA	✓	✓	✓	✓
91	https://www.youtube.com/watch?v=vFZfdvM17YE	✓	-	✓	✓
92	https://www.youtube.com/watch?v=OY9Vz5_ivRA	✓	-	✓	✓
93	https://www.youtube.com/watch?v=_1ZZDG5cgWo	✓	-	-	✓
94	https://www.youtube.com/watch?v=wfE8kKHDUJ8	✓	-	✓	✓
95	https://www.youtube.com/watch?v=pbr7cKr9AlE	✓	-	-	✓
96	https://www.youtube.com/watch?v=Td9ECnFjecA	✓	-	✓	✓
97	https://www.youtube.com/watch?v=II5Z3bLH9XU	✓	✓	✓	✓
98	https://www.youtube.com/watch?v=WFP4RIgWJUo	✓	-	✓	✓
99	https://www.youtube.com/watch?v=aEzoEvqPaLU	✓	-	✓	✓
100	https://www.youtube.com/watch?v=o8bLL8BnBtk	✓	-	-	✓
101	https://www.youtube.com/watch?v=C-GkVWOC1kM	✓	-	-	✓
102	https://www.youtube.com/watch?v=oz2Al0ctSsQ	✓	-	✓	✓
103	https://www.youtube.com/watch?v=-qPaYMaj_gY	✓	-	✓	✓
104	https://www.youtube.com/watch?v=FrVKs6KgvAI	✓	✓	✓	✓
105	https://www.youtube.com/watch?v=ovg4-8Z1rnQ	✓	-	✓	✓
106	https://www.youtube.com/watch?v=SZa7UwFZKwE	✓	-	-	✓
107	https://www.youtube.com/watch?v=xrOdbcGq4Jo	✓	-	-	✓
108	https://www.youtube.com/watch?v=a-II-5_plOo	✓	-	✓	✓
109	https://www.youtube.com/watch?v=bqmT6uQR3w0	✓	-	-	✓
110	https://www.youtube.com/watch?v=MFFBOiUCph4	✓	-	✓	✓
111	https://www.youtube.com/watch?v=Dt1sPqeIzNg	✓	-	-	✓
112	https://www.youtube.com/watch?v=piwqgo4JG90	✓	-	✓	✓
113	https://www.youtube.com/watch?v=_daGsG_iEuY	✓	-	✓	✓
114	https://www.youtube.com/watch?v=Uyf87pmFaUg	✓	-	✓	-

115	https://www.youtube.com/watch?v=QP7U9irfVRs	✓	-	✓	✓
116	https://www.youtube.com/watch?v=EHD3ow4tqBE	✓	-	✓	✓
117	https://www.youtube.com/watch?v=gAvzszajqo4	✓	-	✓	✓
118	https://www.youtube.com/watch?v=jzY-9j6oyoo	✓	-	-	✓
119	https://www.youtube.com/watch?v=2cLei6M157o	✓	-	-	✓
120	https://www.youtube.com/watch?v=mHD2dtUsg6c	✓	-	✓	✓
121	https://www.youtube.com/watch?v=8S2rDT_MLyE	✓	-	-	✓
122	https://www.youtube.com/watch?v=u8sdZgx5ync	✓	-	✓	✓
123	https://www.youtube.com/watch?v=FrnVH3kcDrI	✓	-	-	✓
124	https://www.youtube.com/watch?v=W-l8Bx3g2Jk	✓	-	✓	✓
125	https://www.youtube.com/watch?v=ACxrmy8nRvw	✓	-	✓	✓
126	https://www.youtube.com/watch?v=cj28V8MPp0I	✓	-	✓	✓

Chapter 11

11. Summary and future research scope

11.1. Inferences

The Malvan Marine Sanctuary (MMS) is the only Marine Protected Area (MPA) in the Central West Coast of India with rich biodiversity with patch coral reefs. Despite the presence of coral reefs, very little is known so far about the coral reef ecology, health status, i.e., coral disease, bleaching, and influence of different environmental drivers on the reef ecosystem. Therefore, a detailed study of reef biodiversity, the extent of the reef formation, the health status of reef-forming corals, and the impact of coastal pressure and changing climatic condition was planned and executed. The present investigation, therefore, forms a comprehensive study on the biodiversity and ecology of scleractinian corals and associated biota in the MMS.

An account of biodiversity is necessary for the management and the proper functioning of MPA in a region. In the present study, a total of nineteen species of scleractinian coral belonging to fourteen genera and eight families recorded from the MMS. Wherein, thirteen coral species are reported for the first time from this region, of which *Siderastrea savignyana* and *Favites melicerum* are considered to be rare globally, and *Pseudosiderastrea tayami*, *Favites halicora*, *Goniopora stokesi*, *Goniopora stutchburyi are* uncommon species (Veron, 2000). According to the IUCN red list of threatened species, five species (*Pseudosiderastrea tayami*, *Favites melicerum*, *Favites halicora*, *Goniopora stokesi*, *Goniopora pedunculata*) are under the 'near threatened' category, and the *Turbinaria mesenterina* is under the 'vulnerable' category.

Very little was known on the occurrence and diversity of reef-associated fishes from the MMS. The present study reported, 47 species of reef-associated fishes belonging to 35 genera and 26 families. Additionally, during the surveys, artisanal fishing activity was observed in the core coral reef area wishing gill net and cast net, which could lead to the declination of critical functional groups of reef species with cascading impacts on coral reef habitats and associated species in reef ecosystems health degradation.

This study also reports increasing trends in benthic algal cover and intense coral-algal interaction. Twenty-six species of fleshy macroalgae were identified at the four surveyed reef sites. In 2014, the mean algal cover was recorded to be 22.63%±0.72SE. While, in 2019, algal cover spiked to 52.77%± 1.34SE with a spreading of macroalgae and turf algae over the recent dead coral colonies. In 2015, 30.56%±1.97SE of the surveyed coral colonies found to have direct physical contacts of macroalgae, and 26.70%±2.48SE colonies were subjected to turf algal colonization. In 2019, macroalgal contacts with corals surged to 62.75%±3.96SE, and turf-algal spreading on live coral colonies spiked to 52.75%±4.30SE. The consequences of increasing algal competition to coral may be severe, leading to elevated extinction risk and loss of critical reef habitat. However, further work is required to elucidate the impact of increasing algal abundance on the coral resilience and recovery process.

The present study first revealed that the coral-killing bio-eroding Clionaid sponges are spreading in the reef rapidly, which is critical for the management and conservation of the vital reef ecosystem. In 2014, sponge infestation was estimated to be 3.14%, which increased to 11.39% during 2019, indicating the potential threat of Clionaid and other sponge invasions in the MMS reefs, which could be repercussive to coral community structure.

Recurrent annual coral bleaching events and subsequent coral mortality were recorded in the MMS from 2014 to 2019. Major bleaching event recorded in October 2014, showed 14.58%±1.75SE coral bleaching. Bleaching rose in May 2015 and estimated to be 54.20%±2.58SE. Further mass bleaching events documented during May 2016, which caused 46.76%±2.26SE coral bleaching. Coral bleaching in May 2017, October 2018, and April 2019 caused 20.22%±0.73SE, 5.07%±0.61SE and 8.37%±1.09SE coral bleaching, respectively. Therefore, there is an urgent need for long-term monitoring (before-during-after bleaching) to understand the bleaching impacts of improved reef management practices. Thus, this study aimed to gain a synoptic view of the thermal-stress driven coral bleaching prevalence in the Malvan Marine Sanctuary (MMS) and provide insights on the impact of the bleaching events on the reef environment.

Recurring stresses led in a surge of coral diseases prevalence (16.13% in 2015 to 29.39% in 2019) such as white syndrome, tissue necrosis, skeletal growth anomaly, trematodiasis, and boring mollusks infestation. A total of five types of coral diseases were found predominant in the study sites, viz. skeletal tissue anomaly (STA), an infestation of boring mollusks (VER) mostly by vermetid, white syndrome (WS), tissue necrosis, or necrotic patches on colonies

(TN), and trematodiasis (TRM). The relative disease prevalence estimated to be 20.38% during the entire study period from 2015 to 2019 (n=782 out of the n=3837 colonies accounted). In a rapidly changing global environment, the consequences of the increasing coral disease may be severe, leading to elevated extinction risk and loss of critical reef habitat (Sokolow, 2009). Therefore, with our limited knowledge about the disease etiology and limited ability to restrict bleaching, a great deal of science needs to emerge to aid conservation. In the meanwhile, the only plausible solution to save the corals is the restoration of habitats with all possible methods available today (Epstein *et al.*, 2003; Nanajkar *et al.,* 2019).

Physical damage to corals is well documented due to direct anchorage on corals, tourist standing, walking, holding the coral for support, fin movement, jumping on shallow-water corals. The assessment on SCUBA divers' behaviour showed that in 98.41% (n=124) of the video clips, the diver had body contact with corals with their fins and hand. Other activities include touching, holding, sitting, standing, and walking on the coral colonies for maintaining underwater buoyancy and posing for photos/videos and eventually breaking corals. Additionally, almost 97.62% (n=123) of the videos showed visibly damaged coral, which includes skeletal breakage, abrasion, tissue damage, growth anomaly, bleaching, macroalgal, algal turf and boring sponge growth on coral.

Estimation of carrying capacity for reef diving tourism is an important measure to evaluate how much diving pressure one reef can withstand without compromising reef health and species diversity over time; therefore, setting the diving limit is necessary for long-term sustenance of coral reef (Wafar 1997; Jameson *et al.,* 1999). Annual carrying capacity has been measured, and diving limits have been implemented in different reefs across the world. However, in none of the Indian reefs, estimates of diving carrying capacity have been determined so far, and at the present MPA, it exceeds many folds of the global average.

The study has revealed that the abundance of live coral colonies declined dramatically in MMS after each bleaching event. Mean coral abundance drop from 2.54 individual colonies/m^2 in 2014 to 1.04 colonies/m^2 in 2019. Furthermore, live coral cover was declined from 45.09% in 2014 to 20.95% in 2019, which indicates dramatic coral loss due to the recurrent mass bleaching events, algal, and coral boring sponge competition with coral, and due to occurrence of different coral diseases. Extrapolation, based on the present cumulative coral damage trend, predicts the entire habitat to be altered and converted to a *non*-coral ecosystem within a decade if the current magnitude of stressors persists unabated.

11.2. Future research scope

Despite being an ecological hotspot and economic importance, scientific documentation of the coral reef biodiversity and mapping of the threats to marine life yet to be complete in the MMS, which could assist for regulatory interventions from a conservation point of view. Hence, habitat restoration and management practice should be focused on improving the reef environment with limiting reef resource exploitation will help the MMS coral reef to achieve resilience to the local and climate-induced stressors. Further, there is a scope for establishing new species with molecular taxonomic tools.

The only possibility that the corals may survive beyond the predicted trend by halting the deleterious ongoing activities with immediate effects. Besides, to the conventional management practices, an active reef restoration program could restrict further reef degradation, which will ensure the persistence of coral reefs at the MMS, and the goods and services they provide.

"দাও ফিরে সে অরণ্য, লও এ নগর,
লও যত লৌহ লোষ্ট্র কাষ্ঠ ও প্রস্তর
হে নবসভ্যতা!"

রবীন্দ্রনাথ ঠাকুর

"Give us back the sylvan past, take away today's cities

Take as much iron as you can, wood and stone

O neo-civilization"

[Tagore,1895.Chaitali :poem Sabhyatar Prati, p.18]

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Annexure-I

List of Publications

1. **De, K.,** Nanajkar, M., Mote, S., Ingole, B., Coral damage by recreational diving activities in a Marine Protected Area of India: Unaccountability leading to 'tragedy of the not so commons'. ***Marine Pollution Bulletin***, 155. Doi: 10.1016/j.marpolbul.2020.111190

2. **De**, **K.**, Venkataraman, K., Ingole, B., (2020). The hard corals (Scleractinia) of India: a revised checklist. ***Indian Journal of Geo-Marine Science*** (**accepted**).

3. **De, K.,** Venkataraman, K., Ingole, B., (2017). Current status and scope of coral reef research in India: A bio-ecological perspective. ***Indian Journal of Geo-Marine Science***, 46, 647-662.

4. **De**, **K.**, Sautya, S., Mote, S., Ingole, B., (2015). Is climate change triggering coral bleaching in a tropical patchy fringing reef at Malvan Marine Sanctuary, west coast of India? ***Current Science***, 109(8), 1379-1380.

5. Nanajkar,M., **De, K.,** Ingole, B., (2019). Coral reef restoration - A way forward to offset the coastal development impacts on Indian coral reefs. ***Marine Pollution Bulletin***, 149. doi:10.1016/j.marpolbul.2019.110504

6. Hussain, A., **De**, **K.,** Thomas, L., Nagesh, R., Mote, S., Ingole, B., (2016). Prevalence of skeletal tissue growth anomalies in Scleractinian coral: Turbinaria mesenterina of Malvan Marine Sanctuary, Eastern Arabian Sea. ***Disease in Aquatic Organisms***, doi: 10.3354/dao03038

7. **De, K.,** Nanajkar, M., Mote, S., Ingole, B., Coral disease outbreak after an El Nino induced coral bleaching in an anthropized Marine Protected Area of India. ***Marine Environmental Research*** (under review)

8. **De, K.**, Sushant V. S., Mote, S., Nanajkar, M., Ingole, B. Reef-associated ichthyofauna from a marginal coral reef habitat along the West coast of India: implication for management strategies. ***CBM-Cahiers de Biologie Marine*** (under review)

9. Mote, S., Gupta, V., **De, K.**, Hussain, A., More, K., Nanajkar, M., Ingole, B. Host-specific symbiont selection by the coral *Turbinaria mesenterina* and coral bio-eroding sponge Cliona thomasi in an extreme marginal reef. ***Scientific Report*** (under review)

Popular article/ science outreach article:

1. **कल्याण डे, संभाजी मोटे, मंदार नानजकर (2019) मालवण प्रवाळ बेटांची गाथा व व्यथा.** https://www.agrowon.com/agriculture-news-marathi-article-regarding-coral-conservation-21887

Presentation in conference and symposium:

1. Coral reef degradation in the Malvan Marine Sanctuary: Consequence of climate change and increasing human disturbances. Presented in Aquatic Ecosystems: Sustainability and Conservation, 20-21 December 2019, Indian Institute of Science Education and Research (IISER), Pune (Oral).
2. Ecological effect of multiple stressors on a tropical coral reef. Presented in International Conference on Benthos 06-08 March 2019, Cochin University, Kerala (Oral).
3. Ecological impacts of multiple stressors on a tropical coral reef-a case study from India. Presented in WESTPAC Ocean Acidification Symposium, 5-7 November 2018, Xiamen, China (Oral).
4. Coral reef ecosystem in the central west coast of India. Presented in the International conference on climate change and sustainability, 21-23 December 2015, Thakur College, Mumbai (Oral).
5. Biogeography of Scleractinian coral from Indian reefs. Presented in the International Symposium on the Indian Ocean- Dynamics of Indian Ocean: Perspective and Retrospective, 30 November-4 December 2015, Goa, India (Poster).
6. Biodiversity and status of coral reef in Malvan Marine Sanctuary, Central West coast of India. Presented in the Ocean Science conference (OSICON 2015), 22-24 March 2015, Goa, India (Poster)