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GVI Seychelles – Mahé

Marine Conservation Expedition

January - December 2018

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Seychelles National Parks Authority (SNPA)

Produced by

Marta Andrea Ezeta Watts – Science Coordinator

Nico Fassbender – Science Officer

And

Jim Clark – Program Manager

Stewart Porter – Dive Officer

Bella Mangroo – Research Assistant

Luke Searle – Evans – Scholar

Katie Searle – Evans – Scholar

Lydia Ropel – Scholar

Christophe Mason-Parker – Regional Director

Thank you also to our hardworking volunteers for the collection of all data.

GVI Seychelles – Mahé/Marine Conservation Expedition

Address: GVI c/o SNPA, PO Box 1240, Victoria, Mahé, Seychelles

Email: capternay.base@gviworld.com Web page: http://www.gvi.co.uk and http://www.gviusa.com

Summary

This report summarises the findings of data collected from January to December 2018, covering all monitoring activities carried out by GVI Seychelles on the northwest coast of Mahé. This included surveys to determine benthic cover, hard coral genera diversity and lifeforms, coral recruitment, reef and commercially important fish density, as well as abundance of invertebrates that are of commercial importance or ecosystem indicator species.

Mean percentage hard coral cover across all surveyed sites has increased by 12% (17.93 ± 1.20 %) in comparison to the previous survey period (2017: 15.93 ± 0.90%). It is still below 2010 levels 34.66 ±1.47 %), but 50% higher than 2005 mean hard coral cover (2005: 11.95 ±0.79 %). All granitic sites combined remain at a higher percentage coral cover than carbonate sites combined, (19.60 ± 1.74 %, and 16.53 ± 1.61 % respectively). Additionally, the most dominant lifeform on both carbonate and granitic reefs is encrusting coral. Highest coral cover was found at the granitic site Bay Ternay North East with 32.91 (±8.12) % and the granitic site Therese South with 26.78 (±4.94) %. *Porites* was the most abundant coral genus found on all sites (comprising 32.20 % of mean hard coral cover). Mean density of coral recruits was found at 7.81 (±0.13) individuals per m^2, a decrease of 22% in mean coral recruits per m^2 from the previous survey period. Highest coral recruit density was recorded at the granitic site Whale Rock with 7.54 (±2.25) coral recruits per m^2, the lowest at Baie Ternay North West with 2.84 (±0.38). Highest recruitment was found for the genus *Porites* with a mean of 1.78 (±0.06) recruits per m^2.

The mean fish density in 2018 for all survey sites was 0.395 (± 0.02) individuals per m^2 (Fig.18). Compared to 2017, density declined by 5.5% (2017: 0,417 ± 0.02). Since 2016, mean fish density has now declined by 12.7% compared to fish stocks prior to the bleaching event. While the severity of the reduction in density following the 2016 bleaching event is not comparable to that observed post-2008, a lag effect is still clearly visible with fish densities continuing to decrease in 2018. Reef fish density decreased by 5.2% compared to the previous year (0.166 ± 0.01) and by a total of 15.3% since 2016 (0.186 ± 0.01). Commercial fish density decreased by 16,7% compared to 2017 (0.252 ± 0.01) and by a total of 10.9% since 2016 (0.265 ± 0.01). Commercial fish continue to show higher densities than reef fish. Except for Scaridae, all commercial fish species show a decline in abundance over the past year after showing a slight increase for 2017. Reef fish display varying changes in density throughout the past year after the bleaching event in 2016. The most noticeable and prominent change occurred in the Chaetodontidae family, where a significant decline can be observed for each year with densities now 57.7% lower than 2016

levels. This can be attributed to a substantial loss of hard coral cover amongst all survey sites. Protected sites continue to display a higher overall fish density than unprotected sites and show early signs of recovery with fish densities for 2018 increasing by 3.35% (0.489 ± 0.017) compared to 2017 (0.474 ± 0.032). Baie Ternay Centre once more supports the highest density of fish in 2018 (0.766 fish m^2), an increase of 23.9% compared to 2017 (0.618 fish m^2). For the other survey sites, a clear pattern can be observed with the most exposed / semi-remote sites located around the islands of Therese and Conception continuing to support higher fish densities than sites situated along the Mahé coast. The biggest decrease was observed at Rays Point, with fish densities dropping by 41.5% compared to 2017. Overall fish abundance of smaller juvenile fish in-between 0-20 cm is higher at unprotected sites while adult fish sized between 21-50 cm display a higher abundance at protected sites. Serranidae fish density is higher within the marine protected areas for all size classes bigger than 10 cm (Figure. 28).

For the 10 m Line Intercept Transects, all invertebrate taxa decreased in mean density in comparison to 2017 surveys. Consistent with previous years, long-spined (*Diadema* sp.) and short-spined (*Echinothrix* spp.) urchins were the most abundant invertebrates recorded on the 50 m belt surveys during both survey periods of January-June and July-December 2018. The overall mean density of *Drupella* spp. continues its decreasing trend since 2015, while still maintaining levels greater than 2009; 0.006 individuals per m^2 (± 0.09) compared to 0.013 (± 0.0025) in 2018. Sea cucumbers increased from 23.31 (± 2.4) in 2017 to 29.36 (±0.36) in 2018. *Pearsonothuria graeffei* and *Stichopus* spp. populations were the most commonly observed taxa with 0.02 (±0.003) individuals m^{-2} and 0.01 (±0.007) individuals m^2 respectively. Densities of *P. graeffei* remained at a similar level as observed in 2017.

Results show that the global bleaching event, impacting the Seychelles in early 2016, continues to affect the coral reef communities of northwest Mahé. Although live hard coral cover increased, diversity and recruitment are decreasing. Fish densities continue to decrease at unprotected sites yet start showing early signs of recovery at protected sites. The continuation of this monitoring programme will be valuable in the coming years when aiming to understand the impact of predicted future anthropogenic climatic disturbances, such as sea temperature rise induced bleaching events. Continuing these efforts will prove crucial in guiding best management practices to aid future reef resilience in the Seychelles and Western Indian Ocean.

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1. Introduction

Global Vision International (GVI) is a globally operating volunteering organisation, which has two expedition bases within the inner granitic islands of Seychelles. One expedition base is situated on Curieuse Island within the Curieuse Marine National Park to the north of Praslin. The other expedition base is located within the Baie Ternay Marine National Park at Cap Ternay in the northwest of Mahé Island. All of GVI's scientific activities in Seychelles are carried out on behalf and under the methodological directory of the Seychelles National Parks Authority (SNPA), which manages all of Seychelles' national parks. GVI provides experienced staff, volunteers and supplies equipment to support the research section of the SNPA in their monitoring activities by the collection of long-term data sets.

1.1 Coral reef monitoring

The 1997/98 El Niño Southern Oscillation (ENSO) and the subsequent coral bleaching event caused severe coral mortality worldwide (Spencer et al. 2000; Engelhardt 2002). Scleractinian coral mortality in the inner granitic islands of the Seychelles exceeded 90% due to the combined effects of bleaching and an *Acanthaster planci* outbreak (Engelhardt 2002), with dominantly branching genera *Acropora* and *Pocillopora* suffering high rates of mortality (Spencer et al. 2000). Monitoring of recovery of the reefs surrounding the northwest coast of Mahé was initiated in 1998 by the Shoals of Capricorn, a three-year programme funded by the Royal Geographic Society in conjunction with the Royal Society. Reef states and development were further assessed between 2001 and 2004 as part of the Seychelles Marine Ecosystem Management Project (SEYMEMP), which was the most comprehensive assessment of the coral reefs within the inner islands of the Seychelles to date. Eighty-one carbonate and granitic reef sites throughout the inner islands were monitored using fine scale monitoring techniques. Monitoring efforts were continued by Reefcare International, a non-governmental organisation based in Australia. The protocols established by Reefcare International provided a foundation for those adopted by GVI Seychelles, which continued reef monitoring along the northwest coast of Mahé at sites selected by SNPA.

This continuing long-term data set, of over a decade, shows a unique trajectory of reef development, allowing the assessment of ecosystem recovery after the 1998 and 2016 bleaching events. Evaluation of ecosystem recovery after two major bleaching events will provide crucial data and implications for future coral reef and fisheries management.

1.2 Aims

The aim of the continuous survey activities is to monitor hard coral cover, recruitment and diversity, fish density and diversity as well as the density of invertebrates. Specifically, the aims of GVI's survey activities along the northwest coast of Mahé between January 2018 and December 2018 were to:

- Assess diversity and density of reef and commercially important fish species
- Assess sizes of commercially important fish species
- Assess benthic assemblage, including evaluation of hard coral, soft coral, sessile organisms' coverage and substrate composition
- Assess diversity of hard coral genera
- Evaluate coral juvenile recruitment rates
- Assess density of invertebrate hard coral predators and sea urchins
- Assess abundance and diversity of commercially targeted invertebrate species including sea cucumbers, lobster and octopus
- Monitor and manage the abundance of crown-of-thorns sea stars (*Acanthaster planci*)

2. Methodology

2.1 Survey sites

Surveys are conducted at 13 granitic and 11 carbonate reefs around the northwest coast of Mahé (Figure 1). Each survey site is divided into 'shallow' and 'deep' zones, with the shallow zone being defined at 1.5 – 5.0 m depth and the deep zone being between 5.1 – 15.0 m depth. Each site has a central point, marked by a distinctive landmark on the coastline, and is further divided into left, centre and right sections. These areas are loosely defined as such by their position with respect to the centre marker of the site (left and right are reached by a 25 m swim from the centre point). All depths are standardised with respect to tidal chart datum as to eliminate tidal influence. See Table 1 for further site details.

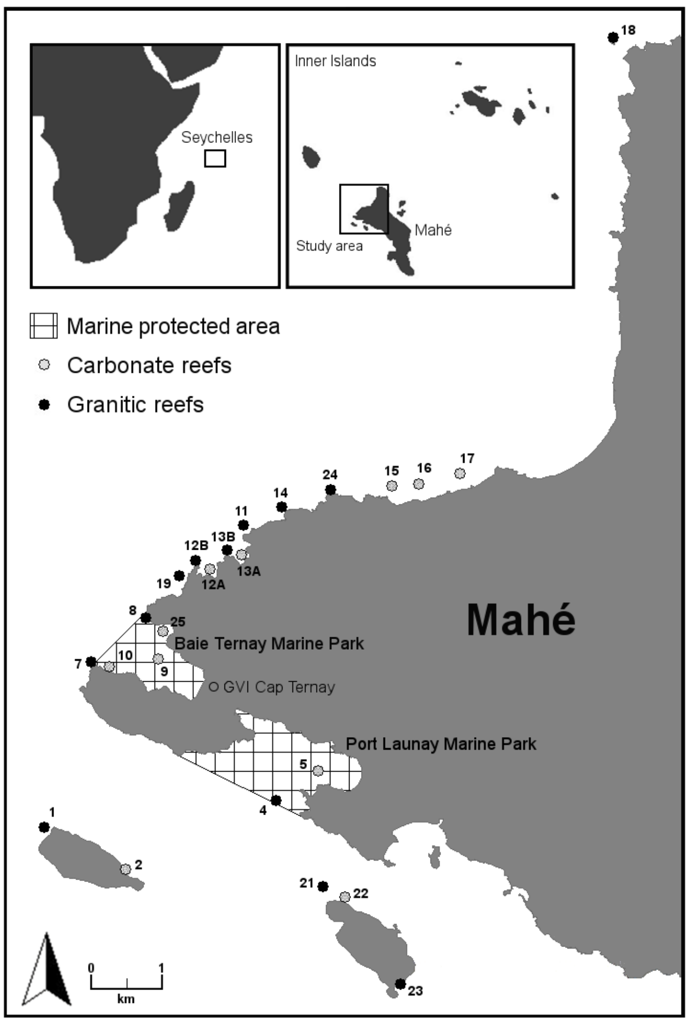

Figure 1: Location and substrate type of survey sites.

Table 1: Survey sites information

Site No	Site Name	GPS	Reef type
1	Conception North Point	S 04°39.583, E 055°21.654	Granitic
2	Conception Central East Face	S 04°39.891, E 055° 22.258	Carbonate
4	**Port Launay West Rocks**	**S 04º39.416, E 055º23.382**	**Granitic**
5	**Port Launay South Reef**	**S 04º39.158, E 055º23.695’**	**Carbonate**
7	Baie Ternay Lighthouse	S 04°38.373, E 055°21.993	Granitic
8	**Baie Ternay Reef North East**	**S 04°38.013, E 055°22.405**	**Granitic**
9	**Baie Ternay Reef Centre**	**S 04°38.321, E 055°22.504**	**Carbonate**
10	**Baie Ternay Reef North West**	**S 04°38.382, E 055°22.133**	**Carbonate**
11	Ray’s Point	S 04°37.347, E 055°23.145	Granitic
12 A	Willie’s Bay Reef	S 04°37.650, E 055°22.889	Carbonate
12 B	Willie’s Bay Point	S 04°37.589, E 055°22.776	Granitic
13 A	Anse Major Reef	S 04°37.546, E 055°23.121	Carbonate
13 B	Anse Major Point	S 04°37.509, E 055°23.010	Granitic
14	Whale Rock	S 04°37.184, E 055°23.424	Granitic
15	Auberge Reef	S 04°37.024, E 055°24.243	Carbonate
16	Corsaire Reef	S 04°37.016, E 055°24.447	Carbonate
17	White Villa Reef	S 04º36.935, E 055º24.749	Carbonate
18	L’ilot North Face	S 04°38.652, E 055°25.932	Granitic
19	Site Y	S 04°37.771, E 055°22.660	Granitic
21	Therese North End	S 04°40.101, E 055°23.737	Granitic
22	Therese North East	S 04°40.099, E 055°23.891	Carbonate
23	Therese South	S 04°40.764, E 055°24.310	Granitic
24	Site X	S 04°37.059, E 055°23.783	Granitic
25	**Anse du Riz ****	S 04°38.065, E 055°22.310	**Carbonate**

* Sites listed in bold are located within marine protected areas

** Formerly named “Secret Beach Reef”

2.2 Expedition Practice and General Methodology

Expedition and survey periods: The GVI expedition comprises of volunteering programs that are four, six, eight or twelve weeks long, running continuously throughout the year. Within one year, each site is aimed to be surveyed for fish and invertebrates twice, with the first set of surveys being conducted from January – June and the second set conducted from July – December. Line Intercept Transects (LITs) and coral diversity transects are undertaken from January – June to evaluate coral coverage and diversity. Coral recruitment quadrats are used from July – December to survey newly recruited colonies.

Health and Safety: The safety of all volunteers is paramount. All volunteers are given a health and safety induction on base upon arrival and conservative diving guidelines are adhered to for the duration of the expedition. In addition, volunteers complete the PADI Emergency First Response course, and are taught how to administer oxygen in the event of a diving related incident.

Dive Training: All volunteers must be at least PADI Open Water qualified to join the expedition. Volunteers then receive the PADI Advanced Open Water course covering Boat, Peak Performance Buoyancy, Navigation, Underwater Naturalist, and Deep Dive. Volunteers also complete the PADI Coral Reef Research Diver (CRRD) course, which is specifically developed for GVI. All volunteers are trained in the use of surface marker buoys, delayed surface marker buoys and tape reels, plus any other survey equipment specific to the surveys they will be conducting. Volunteers gain sufficient dive experience during the training period prior to conducting surveys. attention is given to the training of good buoyancy skills as surveys are conducted in water as shallow as two metres and over delicate reef ecosystems.

Species Identification and survey methodology training: Volunteers are required to learn identification of fish, coral or invertebrates. Training is provided in the form of presentations, workshops and informal discussion with the expedition staff. Self-study materials are also available in the form of electronic and hard copy flashcards, as well as Indian Ocean identification publications. Volunteers are taken on identification dives with staff members for in-water testing; their responses are recorded, and the dives continue until the volunteer has demonstrated accurate identification of all necessary species/genera. Volunteers need to pass a final classroom exam with at least 95% before they can proceed with the training in survey methodology. To learn GVI's survey methodology for the respective surveys, volunteers receive initial on land training and subsequent in water training during which they conduct practice surveys together with a staff member. This training continues until volunteers are deemed confident and reliable to conduct actual surveys.

2.3 Survey methodologies

Coral and Benthic cover surveys

Genera surveyed

During all benthic surveys, hard corals are surveyed to genus level, including 50 genera from 14 families. Genera were first introduced for the LIT surveys in 2009, prior to which only *Acropora* and *Pocillopora* were surveyed to this level, with all other genera categorised as ‘other coral’, and broken down into growth forms. See Appendix A for the full list of coral genera and benthic categories used during the LIT surveys.

Line Intercept Transects (LIT)

Benthic cover and substrate composition around northwest Mahé was assessed between January and June 2018 with six 10 metre LITs at each site. Each transect was placed parallel to the shore, with three transects placed within the shallow depth range (1.5 m – 5.0 m) and three transects placed within the deep depth range (5.1 m – 15.0 m). All survey depths were standardised to the respective chart datum at the time of the survey. Transects were haphazardly spread amongst the left, centre and right of the site with at least 15.0 m distance between them to avoid overlap (Figure 2). The benthic assemblage encountered directly under the tape as well as the respective substratum was identified and recorded at each transition point to the nearest centimetre. Coral was identified to genus level and majority growth form of the colony recorded.

Coral Diversity Belt Transects

Two 50 m belt transects were conducted at each site to assess the diversity of coral genera between January and June 2018. The transect tapes were laid out from the shallow centre towards the deep left (Belt A) and the deep right (Belt B) of the site at a 45˚ angle from shore where possible (Figure 2). Due to the topography of some sites, various transects had to follow the reef instead of a 45˚ angle. Each diver in a buddy pair surveyed 2.5 m in a tight S-shape pattern to the left or the right of the transect tape, recording coral genus presence or absence.

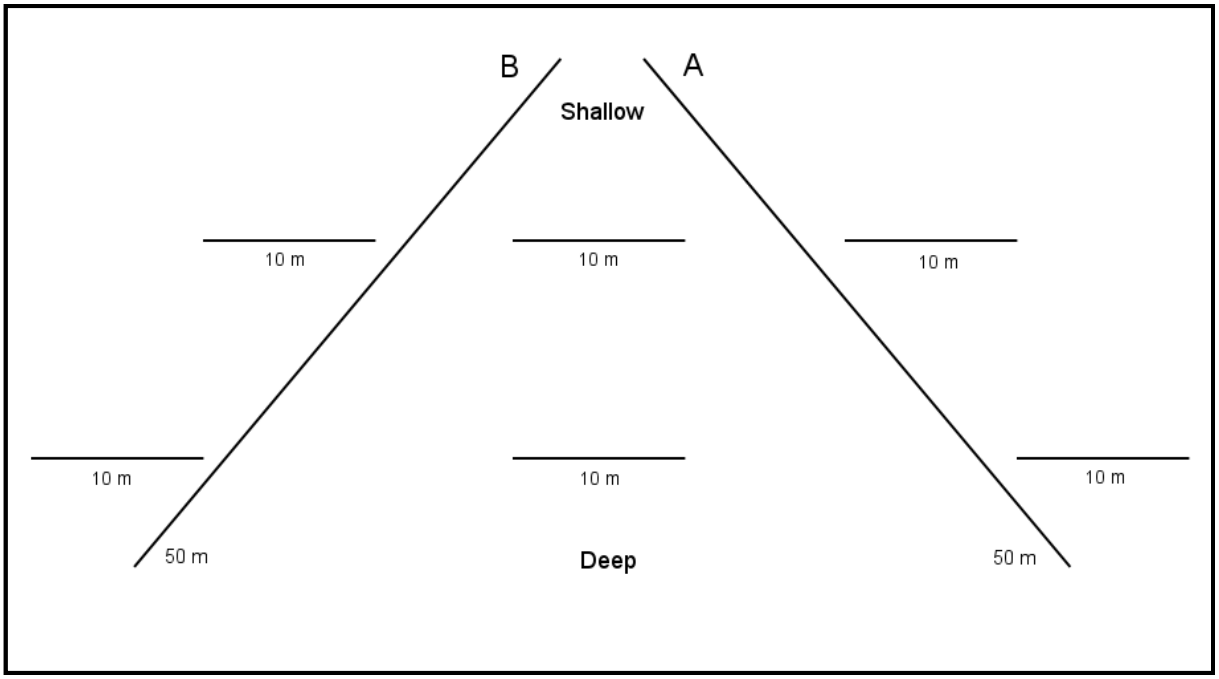

Figure 2: Survey site schematic: Layout of benthos line intercept and invertebrate belt transects (10 m each) and coral diversity as well as invertebrate belt transects (50 m each)

Coral recruitment quadrats

Reef regeneration around northwest Mahé was investigated using haphazardly placed 1m^2 benthic quadrats (methodology based on (Engelhardt 2002)). Placement of quadrats was done across a specified depth range (1.5 m – 5 m for shallow, and 5.1 m – 15 m for deep surveys) (Figure 3). Quadrats were placed over reef substratum, not on large patches of sand or silt, and were held to a height of 1m above the area to be sampled, carefully dropped then allowed to settle before examining the area contained. To ensure safe diving practices, surveys were conducted in a buddy pair with each diver working on any one quadrat and quadrats were placed 2m apart to maintain buddy contact. The percentage of substrate cover (rock, rubble and sand) was described for each quadrat together with percentage algal cover and the depth. Individual coral recruits located within the quadrats were assigned to one of two size classes (0-2 or 2.1-5 cm size class), identified to genus level and counted. All recruits with distinct grazing marks or any other damage (e.g. bleaching signs) were recorded separately. Ideally 36 quadrats should be completed at each site; 18 for each depth range, although a minimum of 30 quadrats per site was required.

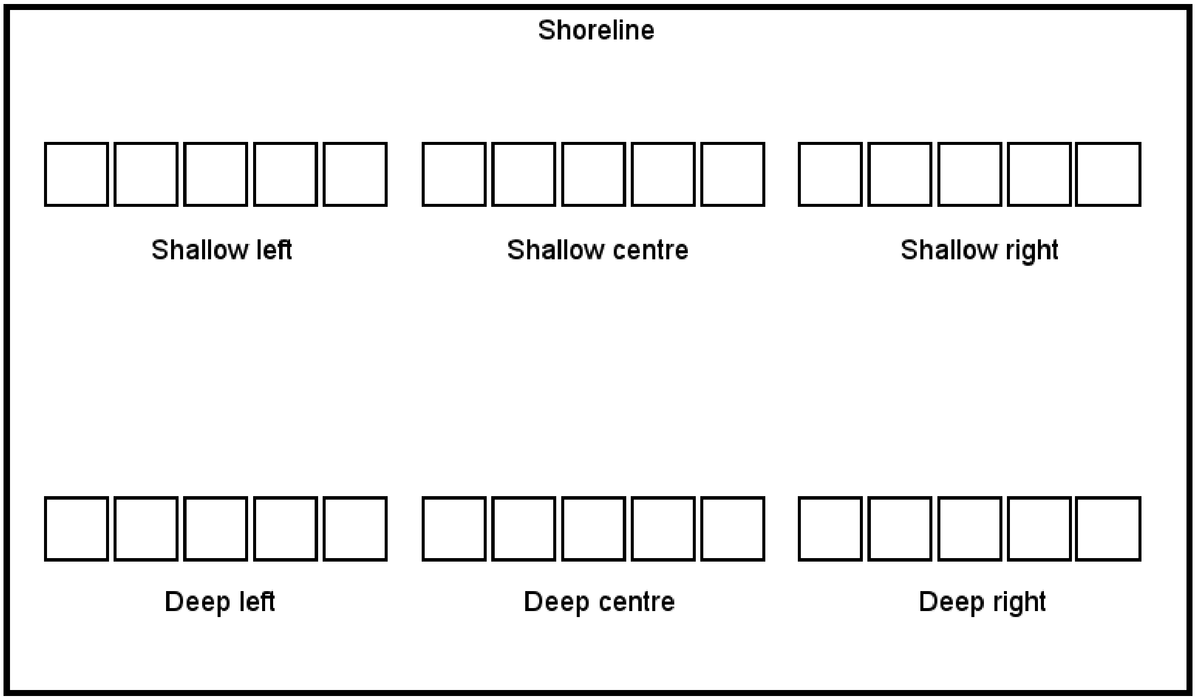

Figure 3: Layout of coral recruitment quadrats at each survey site.

Fish surveys

Species list

The fish species chosen for survey represent a range of species that are commercially important and those that play an important ecological role within the reef community as chosen by SNPA. This data can be used to assess the status of coral reef fish assemblages as well as giving an insight into coral reef dynamics and the state of local fisheries and the community responses to current fishing pressure compared with historical data spanning over a decade.

Fish are generally surveyed to the highest resolution possible with the majority, over 80, being surveyed to species level. Resolution is dependent on the commercial or ecological importance of each species. For example; the majority of parrotfish species fill the same ecological niche and are therefore surveyed to family level (Scaridae); whereas genera that encompass species of more than one feeding guild are generally identified to species level. For a full list of species surveyed and the taxonomic levels used please see Appendix B.

Stationary Point Counts (SPC)

Stationary point counts (SPC) are a commonly used underwater visual census technique for assessing reef fish populations (Kulbicki 1998; Engelhardt 2004) and have been employed, in different variations, by numerous studies internationally (Hill & Wilkinson 2004) as well as locally by several studies within Seychelles (Jennings et al. 1996; Spalding & Jarvis 2002; Graham et al. 2006; Engelhardt 2004). For coral reef assemblages point counts with a radius of 7 - 7.5 metres are thought to be the most appropriate for the size categories that reef fish typically fall into (Samoilys & Gribble 1997). The post bleaching surveys undertaken as part of the SEYMEMP project by Reefcare international utilised point counts with a radius of 7 m (Engelhardt 2001; Engelhardt 2004), when GVI took responsibility for the monitoring program in 2005 a similar point count methodology was adopted.

At each site eight SPCs were conducted, spread evenly between the deep and shallow zones (Figure 4). One SPC was conducted at the left and right sides of the site with two further point counts conducted at the centre of the site in both the deep and shallow areas. Surveys were always conducted by two divers, each responsible for counting a different selection of fish species thus reducing the number of species each person had to count in order to increase accuracy (Samoilys & Gribble 1997). A tape measure was used to delineate the 7 m radius of the SPC and also served as visual reference for the survey area. The tape was laid perpendicularly towards the shore and the depth of the centre of the point count was recorded as well as the start time of the survey period. Before starting the survey, divers waited for at least one minute at the centre of the point count for fish to resume normal behaviour after the disturbance of laying the tape. Each survey lasted a total of seven minutes with the two surveying divers hovering above the reef at the centre point whilst rotating slowly for the first six minutes so minimising behavioural disturbance. A brief search of the survey area was conducted for the final minute in order to give a more accurate count of cryptic species.

Belt Transects

Belt transects were used in conjunction with stationary point counts as they allow surveyors to cover a greater area for a similar level of effort (Colvocoresses & Acosta 2007). However behavioural avoidance of fish species towards divers has been frequently noted and may lead to lower densities of fish than those recorded from SPC’s; therefore, steps were incorporated into the methodology in order to minimise this (Samoilys & Gribble 1997; Hill & Wilkinson 2004).

At each site 4 transect belts were conducted running parallel to the shore, two in the deep zone and two in the shallow, completed in conjunction with the left and right SPC's (Figure 4).

On sites where it was not possible to follow a straight bearing, belts were set following a contour line parallel to the shore. Divers were instructed to count fish above the transect line if it did not touch the substrate. Transect belts were 50 m long and 5 m wide; a standard survey area used by a number of previous studies (Samoilys & Gribble 1997; Hill & Wilkinson 2004). Transects were conducted by a pair of divers with one diver leading and counting one group of fish, while the second diver laid the tape behind. The diver counts the commercially important species on this pass, which can include the more errant species that show a greater level of avoidance behaviours. This method of simultaneously surveying and laying the tape has been recommended by (Samoilys & Gribble 1997) as it avoids disturbing fish prior to the start of the survey. After the initial survey divers waited outside of the survey area for three minutes before the second diver returned down the belt counting the second group of fish (non-commercial) while the tape was reeled up behind them. Each diver completed their surveys in a time of between 8 and 12 minutes allowing a more accurate count of fish abundances as well as decreasing the impact of diver disturbance.

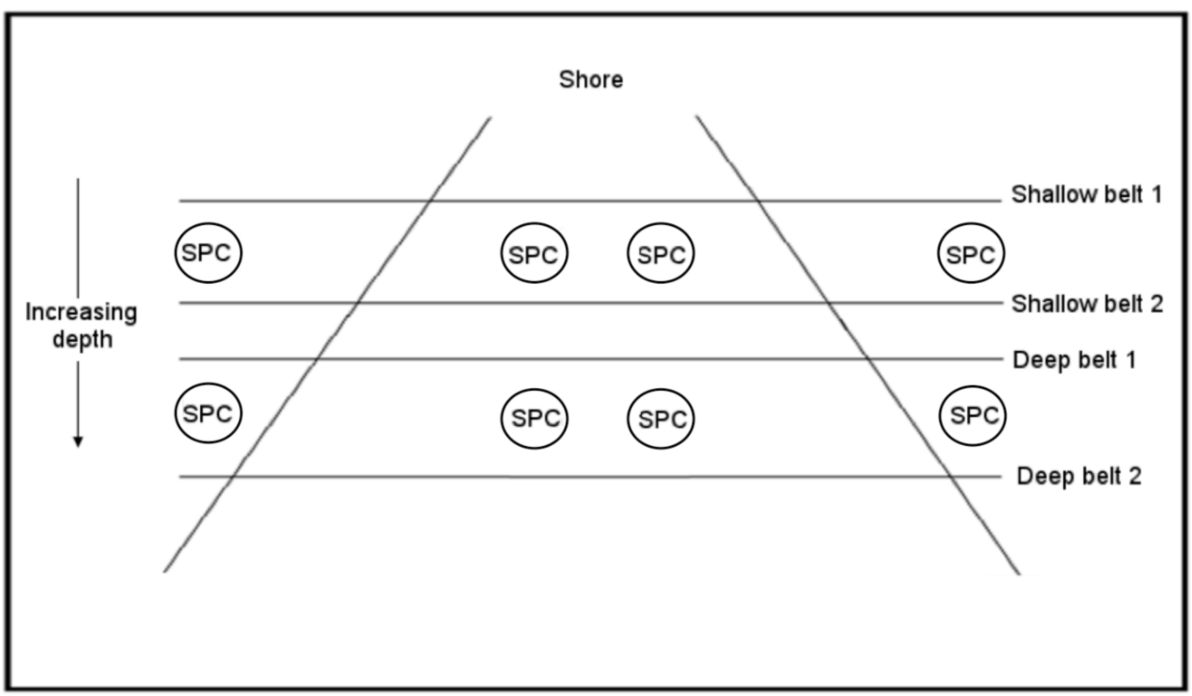

Figure 4: Layout of fish species point counts and 50 m fish visual census belt transects.

Commercial fish size estimation

As well as assessing the abundance, diversity and densities of commercial species from point counts and belt transects, size estimation was used as a surrogate for the biomass of commercial reef fish species and to assess community responses to fishing pressure (Jennings & Polunin 1997; Samoilys & Gribble 1997). Surveyed species that are considered commercially important include emperors (Lethrinidae), groupers (Serranidae), rabbitfish (Siganidae), snappers (Lutjanidae) and sweetlips (Haemuiidae). The diver surveying the commercially important fish recorded sizes in 10-centimetre bands during both the SPC and belt transect surveys. Observer bias was minimised by training volunteers on sizing during their species identification dives and ensuring that surveys matched that of the instructors.

Invertebrates surveys

Species surveyed

Invertebrate species, which influence and can indicate the health and conditions of coral reefs are surveyed along with commercially viable species which are under fishing pressure. The full list of surveyed invertebrate species is included in Appendix C.

Belt transects (10 m)

Invertebrate surveys are conducted in a buddy pair with the coral LIT diver, who lays out the 10 m transect tape (see 2.3.1 Coral Line Intercept Transects (LIT)). At each site, six 10 metre invertebrate surveys are carried out between January and June. After the coral LIT diver lays out the transect, the invertebrate surveyor begins, along the same transect belt, 5 minutes later to minimise underwater clashes. Each transect is placed parallel to the shore, with three transects placed within the shallow depth range (1.5 m – 5.0 m) and three transects placed within the deep depth range (5.1 m – 15.0 m). All survey depths are standardised to the respective chart datum at the time of the survey. Transects are haphazardly spread amongst the left, centre and right of the site with at least 15.0 m distance between them to avoid overlap. Using a systematic 'S' bend swimming pattern, targeted cryptic invertebrate species (see Appendix C) are recorded within 1 meter either side of the 10 m transect, covering a total 20 m^2 area.

Belt transects (50 m)

The 50 m belt transects aim to quantify the abundance of key macro-invertebrate groups in a given dive site. Two 50 m transect tapes are laid out at each site, from the shallow centre point towards the deep. Belt A runs 45° to the left and belt B 45° to the right at each site where possible; due to the topography of some sites, transects have to follow the reef instead of a 45° angle. All survey depths are standardised to the respective chart datum at the time of the survey. Each diver in the buddy pair surveys the target invertebrate taxa (see Appendix C) within 2.5 m on the left and right side of the transect, using the systematic 'S' shaped swimming pattern; surveying a 250 m^2 area.

The extent of hard coral predation is measured by the density of the gastropods in the genus *Drupella* and of two types of sea stars; the cushion stars (*Culcita* sp.) and the crown-of thorns sea stars (*Acanthaster planci*). Algal grazing pressure is measured through recording the density of sea urchins. Sea cucumbers and other species important to fisheries are also recorded.

Environmental parameters

During each survey dive, the boat captain records the following environmental parameters:

- Turbidity, as measured with a Secchi disk
- Cloud cover, as estimated in eights
- Wind speed, as evaluated via the Beaufort wind force scale
- Surface and bottom sea temperatures based on divers' personal dive computers.

2.4 Data analysis

Data collected was analysed with Microsoft Excel without calculation of statistical significance as it is outside the scope of this report. Simpson's Diversity Index ($D = 1 - [\sum (n / N)^2]$) was used to calculate diversity of different sites (for hard coral and fish diversity) taking into account evenness of taxa distribution.

3. Results

3.1 Coral

3.1.1 Live hard coral cover

Percentage hard coral cover was determined from line intercept transects completed across 20 survey sites between the survey period January – July 2018, equating to 120 LIT transects and 1200 m surveyed. Mean live hard coral cover was 17.93 (± 1.20) % across all sites; an increase of 12% compared to the previous survey period (2017: 15.96 ± 0.90 %). Mean hard coral cover has fallen under 2010 levels (34.66 ± 1.47 %), but it is still 50% higher than 2005 mean hard coral cover (2005: 11.95 ± 0.79 %). On carbonate reefs, mean hard coral cover was 16.53 ± 1.61 %, which represents a 6% increase compared to 2017 (15.51 (± 1.45) %). Granitic reefs showed a higher mean coral cover than carbonate reefs with 19.60 ± 1.74 %, an increase of 17% compared to 2017 (16.73 ± 1.12 %) (Figure 5).

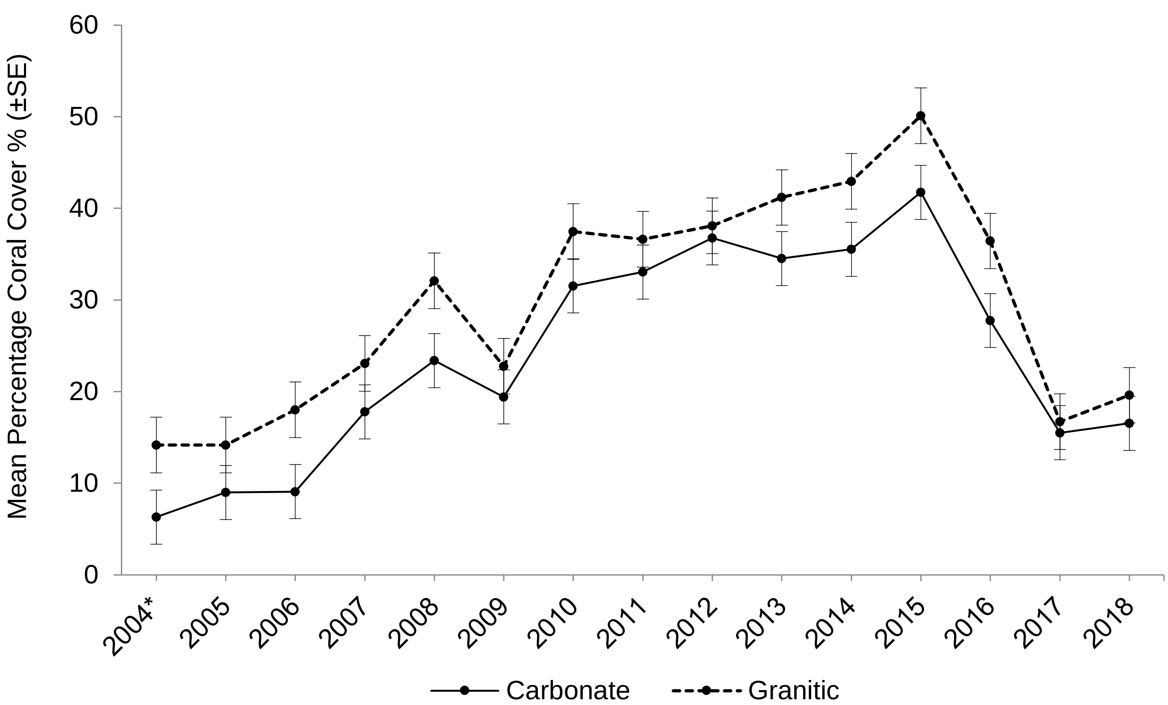

Figure 5. Mean percentage live hard coral cover at carbonate and granitic survey sites for each survey period from 2005 – 2018, including survey results (means only) of Engelhardt (2004*) prior to GVI's survey activities. Error bars indicate the standard error of the mean.

Lowest mean coral cover was found at the carbonate site Willies Bay Reef with 9.22 (±3.90) % and the granitic site Bay Ternay North West with 9.40 (±1.95) %. Highest mean coral cover was found at the granitic site Bay Ternay North East with 32.91 (±8.12) % and the granitic site Therese South with 26.78 (±4.94) % (Figure 6). The five reefs that lay within marine protected areas were found to have a combined mean coral cover of 20.78 (±2.68) % which is higher than the combined mean of sites outside the protected areas (mean coral cover: 16.98 (±1.32) %).

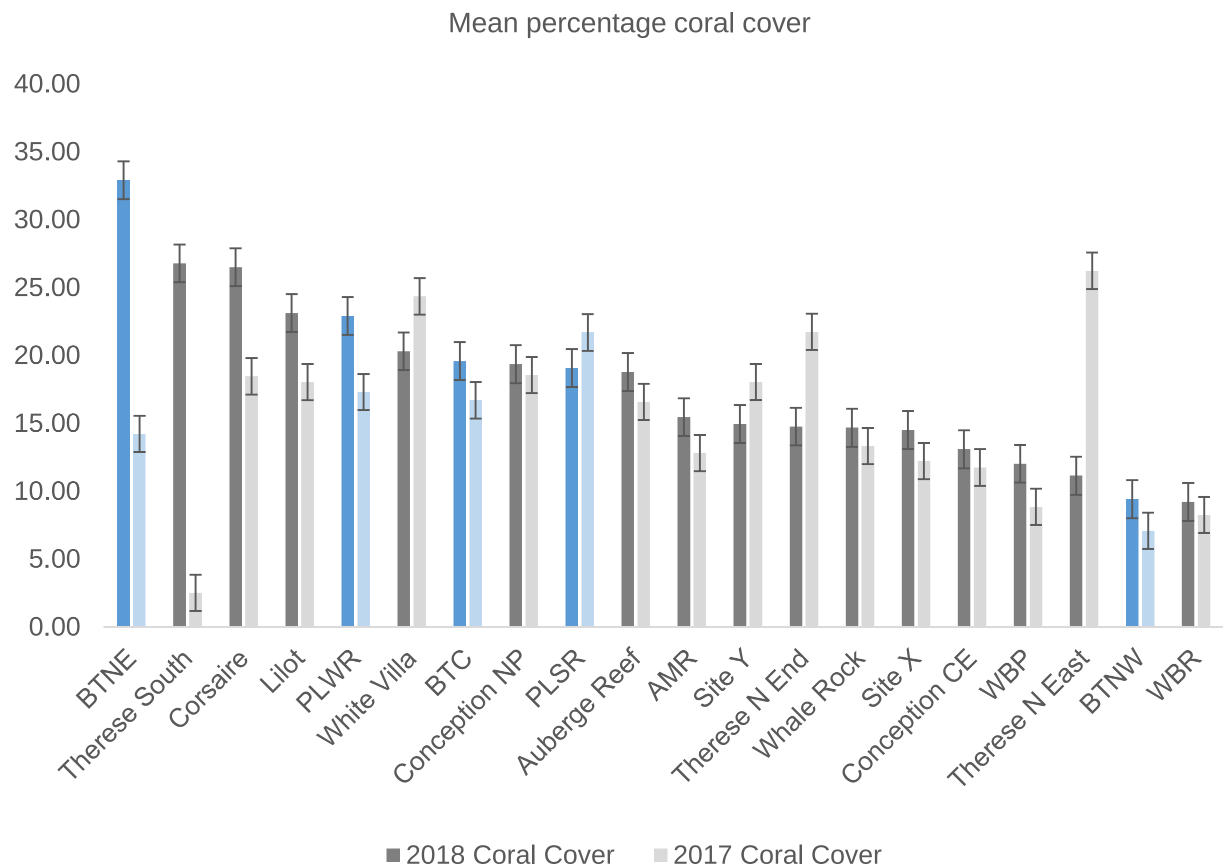

Figure 6. Mean percentage coral cover found at each site surveyed between January and July 2018. Sites are ranked highest to lowest (left to right) for coral cover in 2018. Error bars indicate the standard error of the mean. Blue bars indicate the location within marine protected areas.

3.1.2 Coral genera dominance

Mean cover of *Acropora* on granitic and carbonate reefs combined decreased from 0.85 (±0.26) in 2017 to 0.80 (±0.25) % in 2018 (Figure 7). With the modification of the survey methodology in 2009, identification of the most dominant corals became possible. In 2009, *Porites* were dominant on carbonate sites (~38% of mean live coral cover); *Acropora* and *Favites* dominated on granitic sites (~21% and 20% of mean live coral cover, respectively). In 2010, *Acropora* dominance was observed for the first time; with 2015 levels showing highest dominance with *Acropora* representing 52% of mean live hard coral cover. In 2018 *Acropora* corals decreased across all sites (comprising 4.46 % of mean live hard coral cover). *Porites* corals have also decreased in relative abundance across all sites, with 32.19 % of mean live hard coral cover compared with 2017 where they represented 48.17 % of mean live hard coral cover (Figure 7).

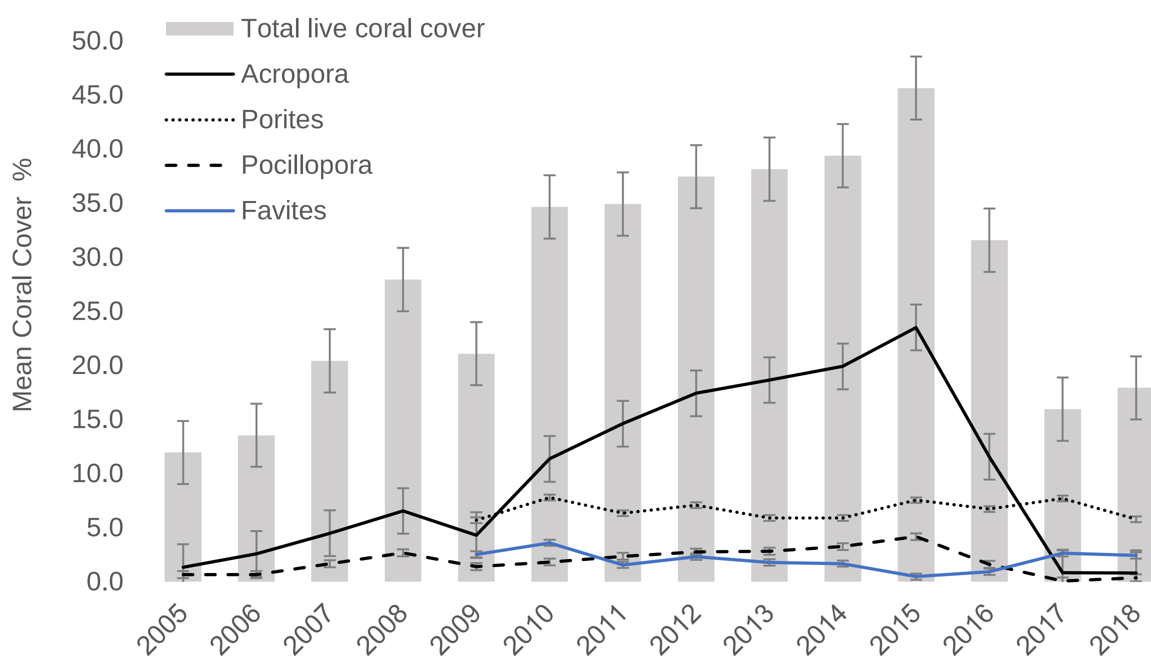

Figure 7: Percentage cover of the coral genera *Acropora, Pocillopora, Porites* and *Favites* against total life coral cover (%) of surveys conducted between 2005 and 2018. Error bars indicate the standard error of the mean. Note that pre-2009 only *Acropora* and *Pocillopora* where surveyed to genus level.

3.1.3 Coral growth forms as proxy for structural complexity

Since 2010 the branching growth form was the most dominant on carbonate reefs, a trend that can be observed throughout the years (Figure 8). In 2017, branching corals reached their lowest point with 0.33 % since the beginning of surveying in 2005. However, this year we can see an increase in their percentage to 3.88 %. Massive corals displayed a major decrease from 40.48% in 2017 to 9.32% in 2018. On granitic reefs, encrusting corals have been predominant since surveying began. In 2018, encrusting corals decrease to 67.48 % from 71.25% in 2017, still being the most dominant lifeform in comparison to 2.67 % of branching corals (Figure 9).

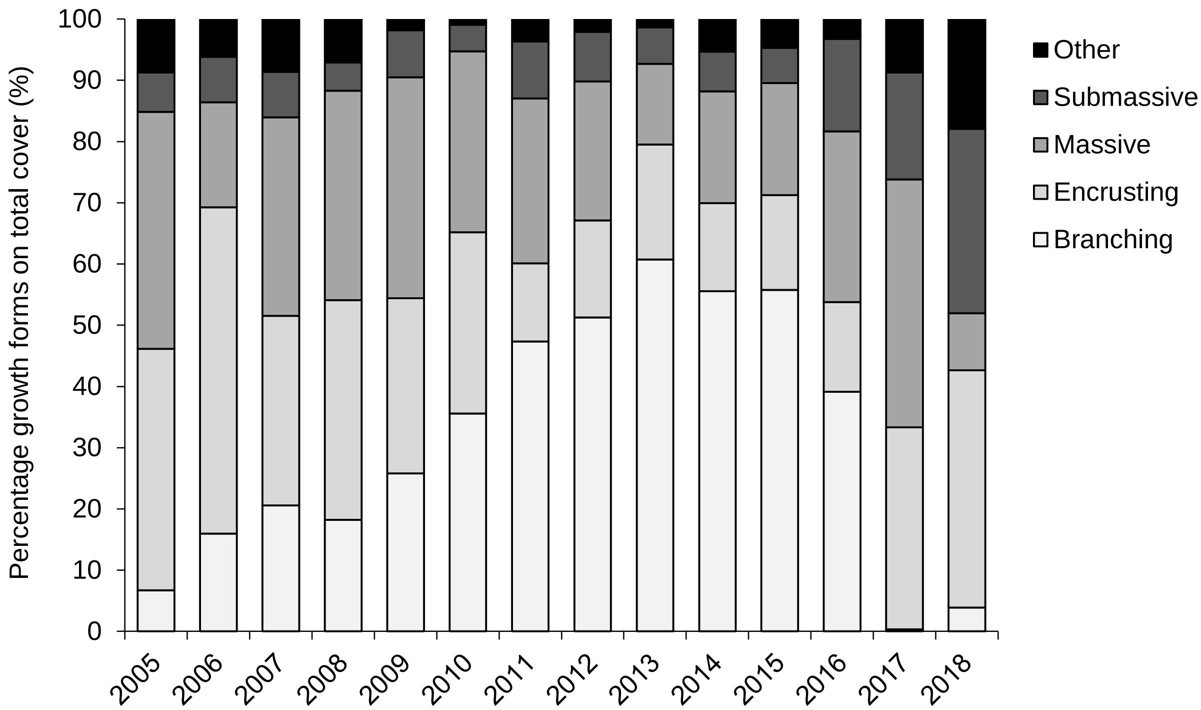

Figure 8. Percentage of coral growth forms on total coral cover on carbonate reefs between 2005 and 2018. The category “Other” includes digitate, foliose, mushroom and unrecorded/missing growth forms.

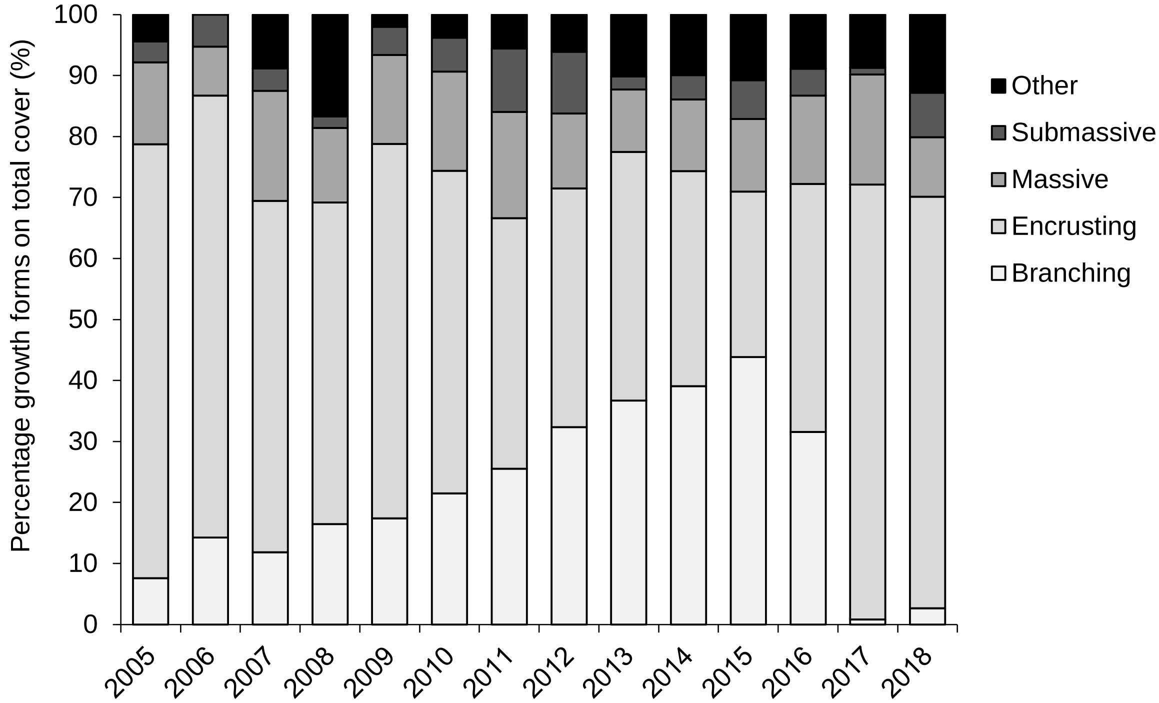

Figure 9. Percentage of coral growth forms on total coral cover on granitic reefs between 2005 and 2018. The category “Other” includes digitate, foliose, mushroom and unrecorded/missing growth forms.

3.1.4 Benthic assemblage

Turf algae represented the highest cover of non-scleractinian organisms in 2018 with a mean of 35.65 (±2.23) % across all sites surveyed with a noticeable decrease from 2017 (49.29 (±1.68) %). Macro algae cover stayed low throughout the years; the cover of coralline algae stayed relatively similar across the years with an increase to 5.01 (±0.79) % (2017: 3.43 (±0.39) %). Algae assemblage displayed a comparably big increase from 18.31 (±1.47) % in 2017 to 27.04 (±2.40) % this year (Figure 10).

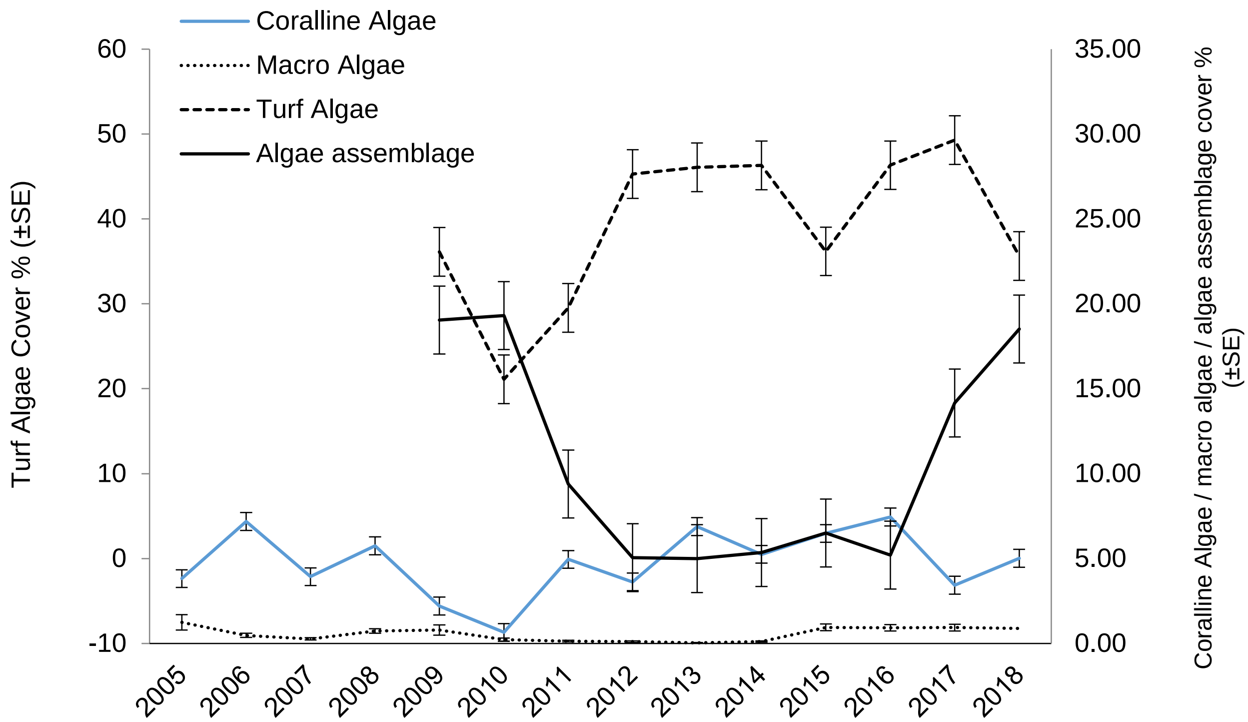

Figure 10. Percentage cover of different algae categories from 2005 – 2017. Error bars indicate the standard error of the mean. Note the categories 'Turf algae' and 'Algae assemblage' were added to the LIT categories in 2009.

Between carbonate and granitic sites, coralline algae cover is relatively similar (mean cover for carbonate sites: 5.37 ±1.29 %; mean cover for granitic sites: 5.20 ±1.07 %). Turf algae cover is slightly higher on carbonate sites for 2018 (mean cover for carbonate sites: 38.10 ±3.25 %; mean cover for granitic sites: 31.56 ±3.05 %).

The mean cover of other benthic organisms is shown in Figure 11. Across all sites in 2018, mean cover of corallimorphs and zoanthids was 3.19 (±0.44) %, whereas for soft corals and sponges a mean cover of 1.69 (±0.50) % and 1.82 (±0.32) % was recorded respectively. Zoanthids and corallimorphs are more prevalent than soft corals and sponges, a trend observed from 2005 to 2008 and last in 2014, in contrast to soft corals being dominant.

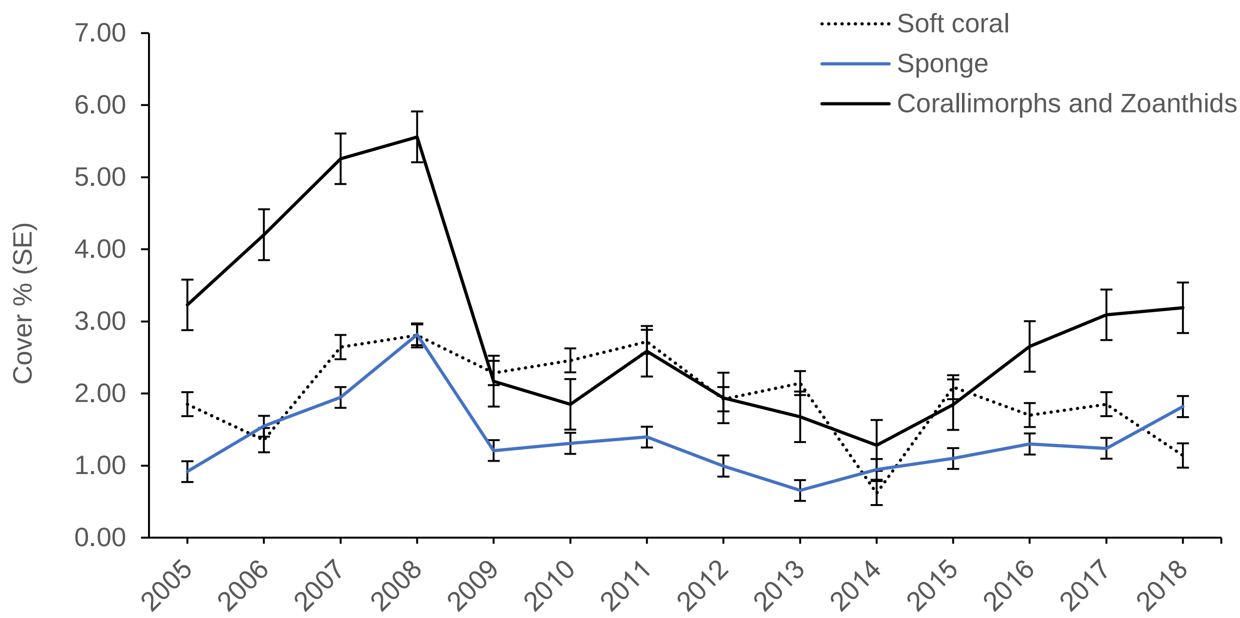

Figure 11. Mean percentage cover of other benthic organisms from 2005 to 2018. Error bars indicate the standard error of the mean.

3.1.5 Hard coral genera diversity

Mean hard coral genera richness seems to fluctuate between 24 and 33 genera since surveying began in 2005 (Figure 12). In 2018 a mean of 27.65 (±0.63) coral genera was recorded across 19 survey sites, which displayed a reduction from 2017: 28.05 (±0.74). The highest diversity of 33 coral genera was recorded from surveys conducted on the granitic site BTNE and Whale Rock, the lowest at the carbonate site Therese North East with 24 genera.

Rare genera are *Alveopora, Coeloseris, Diaseris, Pectinia, Seriatopora* and *Siderastrea*. *Seriatopora* and *Diaseris* were last recorded in 2005 and 2006, respectively. In the following years recordings of rare genera was reduced to a few recordings of especially *Alveopora,* with additional recordings of *Siderastrea* in 2007 and *Pectinia* in 2008. Recordings of rare genera were seldom, except for 2011 when *Coeloseris, Alveopora* and *Siderastrea* where recorded on different survey sites. In 2018, *Siderastrea, Coeloseris,* and *Pectinia* were not recorded and *Alveopora, Halomitra,* and *Merulina* were observed only once during surveys. However, from personal observations, *Oulophyllia* and *Symphillia* have been observed on several survey sites.

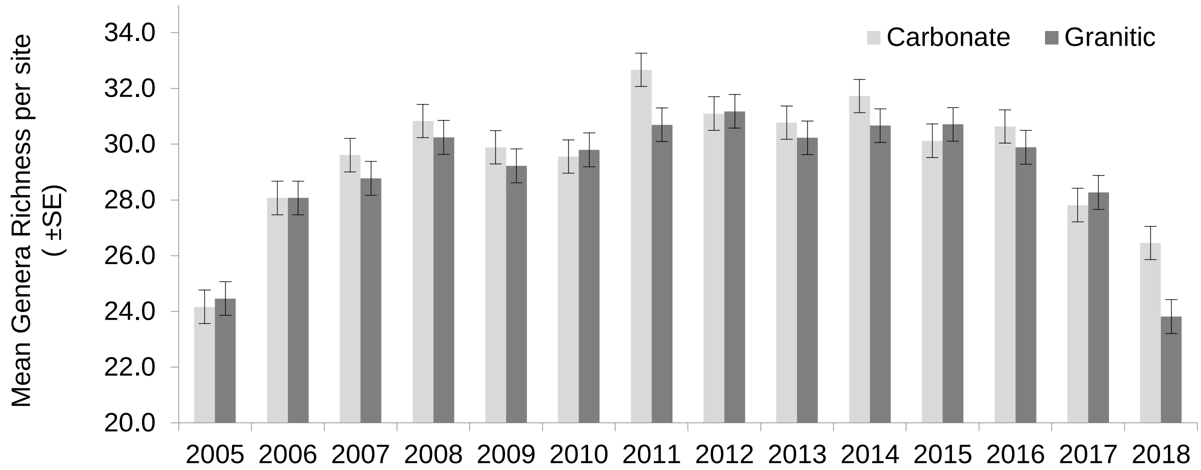

Figure 12. Mean number of coral genera found at all carbonate and granitic survey sites from 2005 to 2018. Error bars indicate the standard error of the mean.

The Simpson's Diversity Index considers species richness and abundance and is a standard ecological measure of biodiversity from 0 to 1, with higher values of D indicating higher diversity. The highest index is calculated for the granitic site Therese South (0.89), and lowest at the carbonate site Auberge Reef (0.38) (Figure 13).

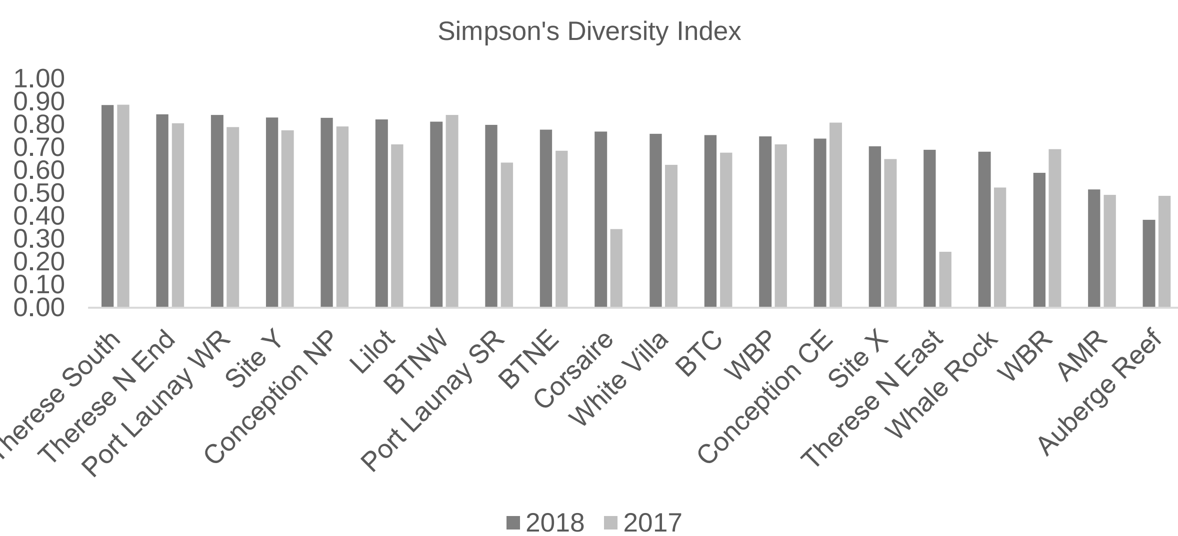

Figure 13. Simpson's Diversity Index at each site surveyed between January and July 2018 (Simpson's Diversity Index D = 1 – [$\sum (n / N)^2$]). Sites are ranked highest to lowest for 2017.

3.1.6 Coral Recruitment

The mean density of coral recruits per m^2 for all genera across survey sites decreased by 22% in 2018 to 7.81 (±0.13) recruits per m^2. This represents a 20% decrease in mean coral recruits per m^2 compared to 2005 (9.74 ±0.14 recruits per m^2) (Figure 14).

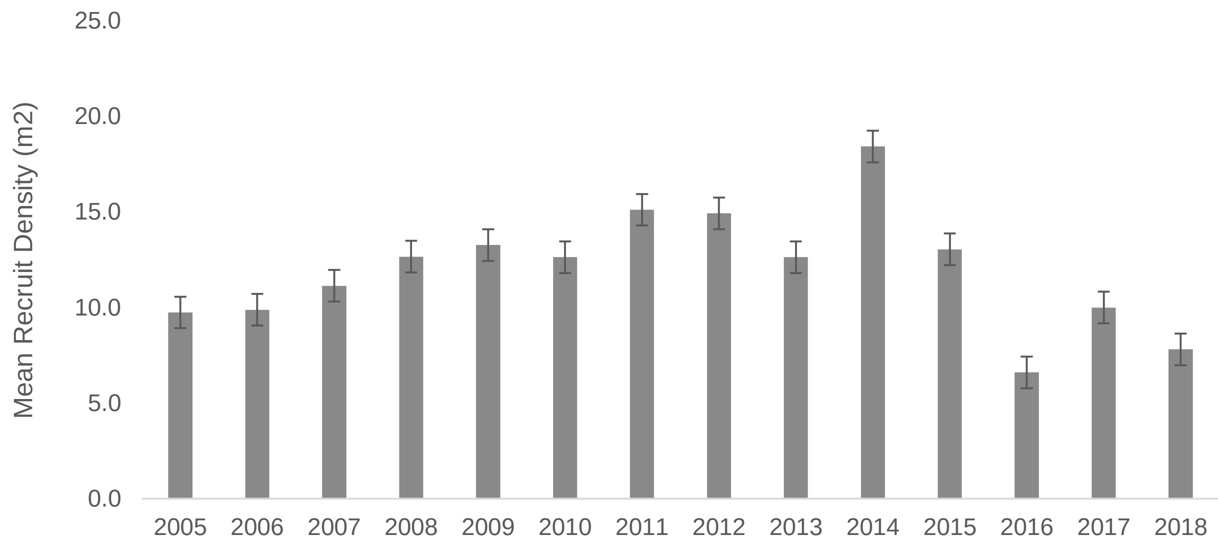

Figure 14. Mean coral recruit density per m^2 for all surveyed sites across all monitoring years 2005 – 2018. Error bars represent ± Standard Error (SE).

Mean coral recruits per m^2 in deep (5.1 – 16 m) and shallow (1 – 5.0 m) areas of surveyed sites both decreased from 2017 values (Figure 15). Recruits in deep areas decreased from 11.56 (±0.34) recruits per m^2 in 2017 to 8.27 (±0.25) in 2018. Similarly, those in shallow areas decreased from 8.44 (±0.20) in 2017 to 7.36 (±0.23) in 2018. Comparing these areas revealed a higher recruitment rate in deeper areas of the reef. The two different size classes of coral recruit recorded (0 - 2 and 2.1 - 5 cm) also decreased from 2017 values. Mean recruit density for the 0 - 2 cm class decreased from 4.34 (±0.10) in 2017 to 3.49 (±0.07) in 2018. Similarly mean recruit density for the 2.1 - 5 cm class decreased from 5.66 (±0.08) in 2017 to 4.32 (±0.06) in 2018.

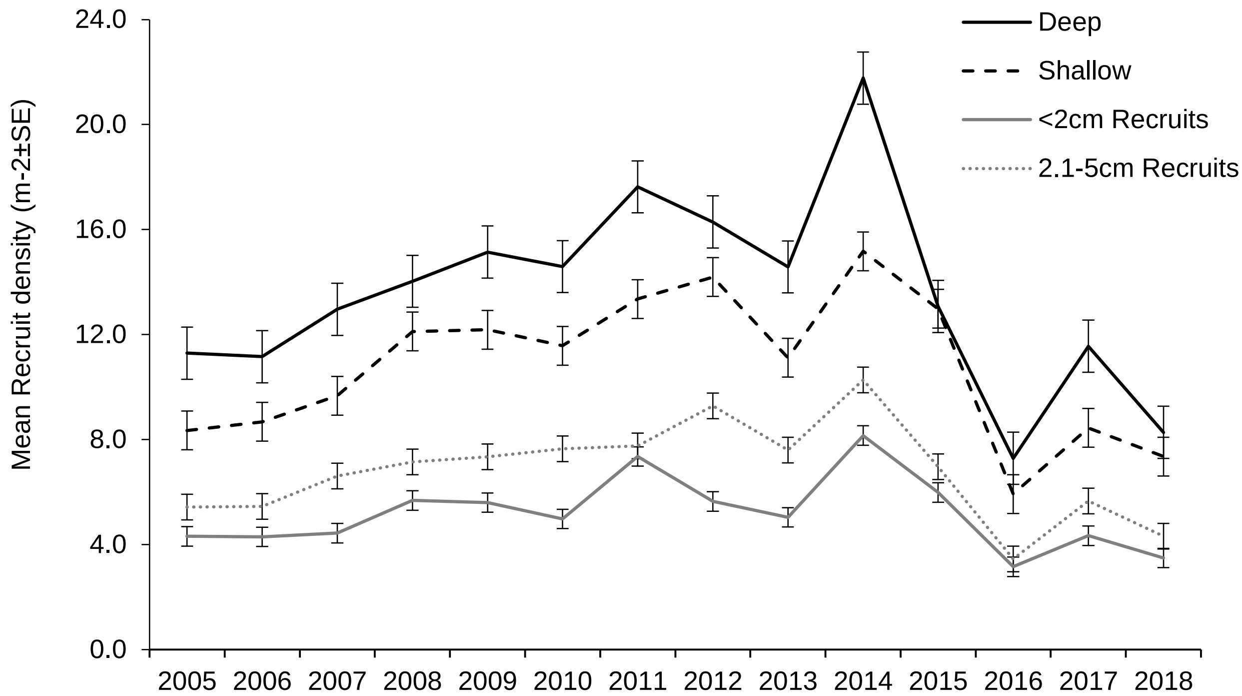

Figure 15. Mean coral recruit density per m^2 at shallow (1.5 – 5.0 m), deep (5.1 – 16 m) and for the two size classes of recruits (0 – 2 cm & 2.1 – 5 cm) for all surveyed sites across all monitoring years 2005 – 2018. Error bars represent (± SE).

Mean coral recruit density decreased at the majority of the sites surveyed in 2018 in comparison to 2017, with the exception of Baie Ternay Central, Therese North End, Willies Bay reef, Port Launay South Reef and Conception Central East Face (Figure 16). The highest recorded recruit density was found at Whale Rock with a mean of 7.54 (±2.25) coral recruits per m^2, the lowest mean density was found at Baie Ternay North West with 2.84 (±0.38) coral recruits per m^2. Granitic sites (n=8) had higher average of coral recruits per m^2 with 9.36 (±0.20) in comparison to carbonate sites (n=10) 6.58 (±0.13). On average unprotected sites (n = 13) had a higher density of coral recruits per m^2 with 8.43 (±0.16) in comparison to protected areas (n = 5) 6.21 (±0.13).

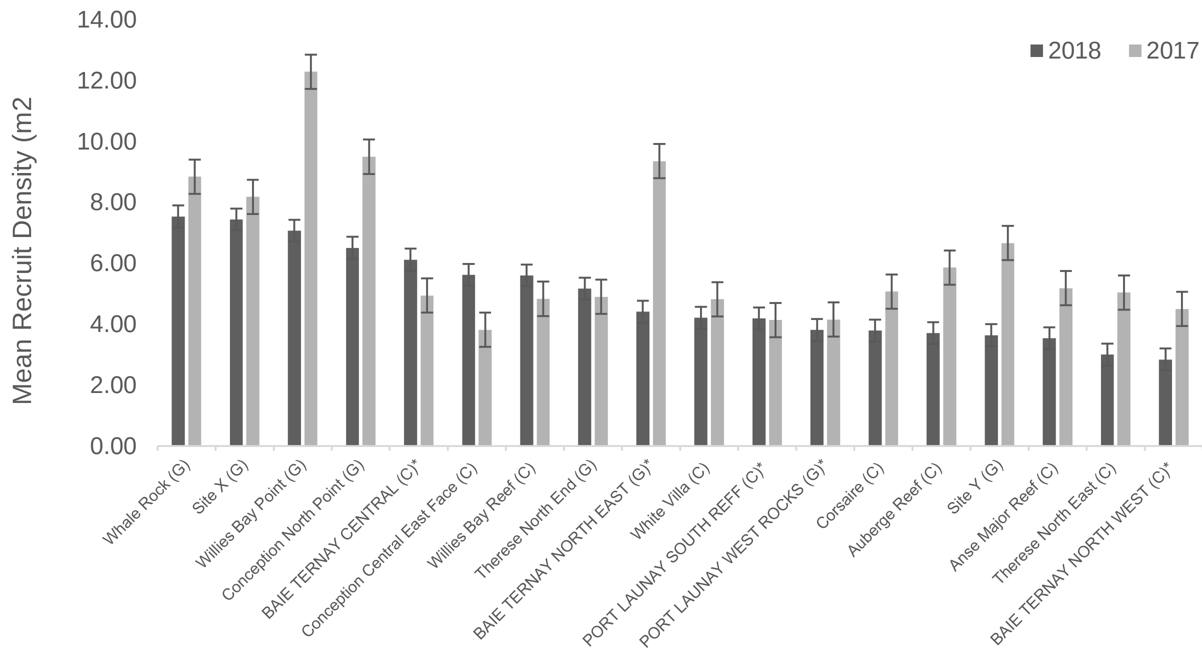

Figure 16. Mean coral recruit density per m^2 recorded at each surveyed site between July – December 2018. Site names in capitals and marked with an asterisk (*) indicate the site is located within a marine protected area. (G) indicates granitic reefs, (C) carbonate reefs. Error bars represent (± SE).

Across all sites surveyed for coral recruitment in 2018 a total of 39 coral genera were recorded from 13 different families, exactly as observed in 2017. This year the top four dominant coral recruit genera are *Porites, Favites, Pavona and Favia* accounting for an average 58% of the total composition of coral recruits. This is the first year that *Acropora* is not listed in the top 4 dominant genera. The genus *Porites* has the highest recruitment density with 1.78 (±0.06) coral recruits per m^2. It is also the dominant recruit on average (22.73% total recruits) decreasing from 2017: 3.12 (±0.08) coral recruits per m^2 (31.22% of total) (Figure 17). *Favites* decreased from 1.65 (±0.06) coral recruits per m^2 in 2017 to 1.30 (±0.04) in 2018, still being the second highest coral recruit genera (16.59% of total). *Pavona* displays the third highest recruiting rate increasing from 0.79 (±0.03) coral recruits per m^2 in 2017 to 0.89 (±0.03) in 2018 (11.35% of total). *Favia* displays the fourth highest coral recruitment, decreasing from 0.71 (±0.01) coral recruits per m^2 to 0.58 (±0.01); comprising 7.44% of total recruits. The previously dominant genera *Acropora* was recorded at its lowest value since records began with 0.41 coral recruits per m^2 in 2018 compared to 0.53 in 2005.

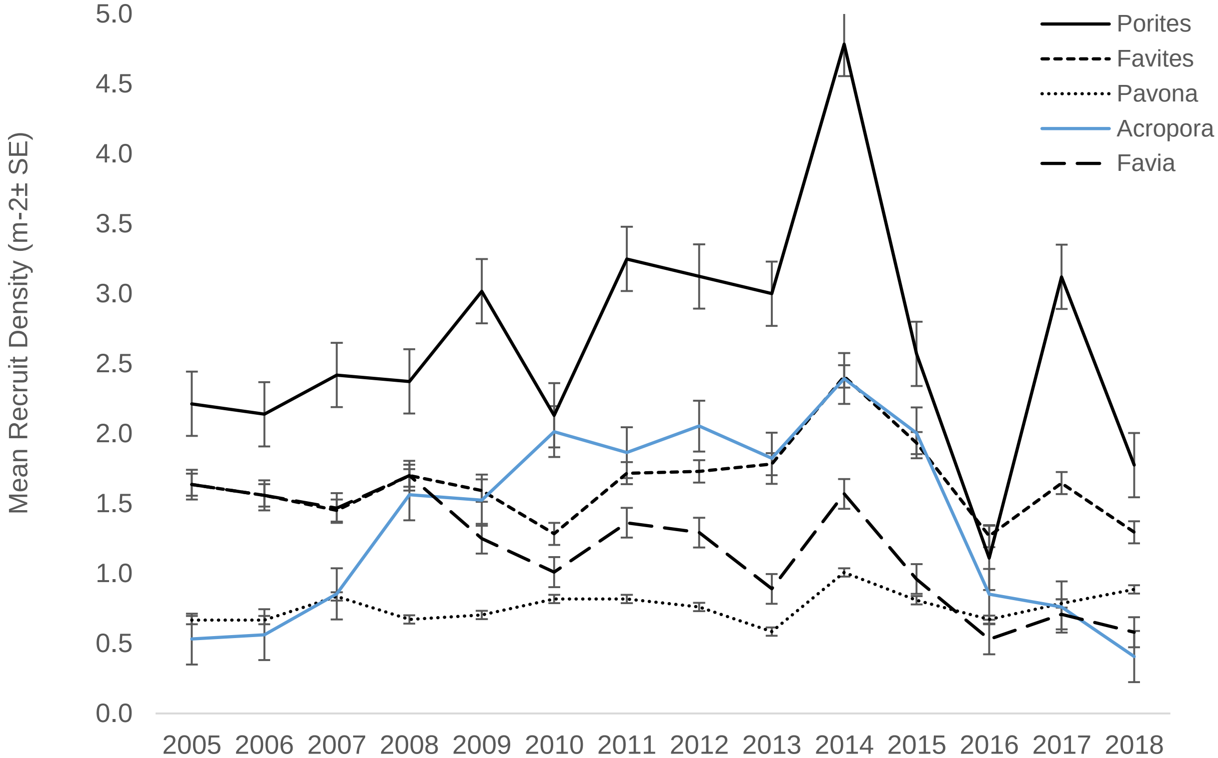

Figure 17. Mean coral recruit density per m^2 for the four most numerically abundant coral genera in 2018 from all surveyed sites from 2005 to 2018. Error bars represent (± SE).

The average diversity (Simpson's 1-λ') in 2018 was 0.888, with the highest diversity being recorded in 2010 at 0.911. Species richness in 2018 (39) was the same as 2017, but lower than the record high of 49 in 2006. Evenness calculated by *Pielou's* evenness (J') showed an increase from last year of 0.652 to 0.710 in 2018. The highest record was 0.717 observed in 2010.

3.2 Reef and Commercial Fish

3.2.1 Overall Densities

The mean fish density in 2018 for all survey sites was 0.395 (± 0.02) individuals per m^2 (Figure.18). Compared to 2017, density declined by 5.5% (2017: 0,417 ± 0.02). Since 2016, mean fish density has now declined by 12.7% compared to fish stocks prior to the bleaching event. While fish densities were found to show an increasing trend since the beginning of monitoring in 2005, with an all-time high in 2016, a continuous decline has been observed since the coral bleaching event in 2016 (Figure.18).

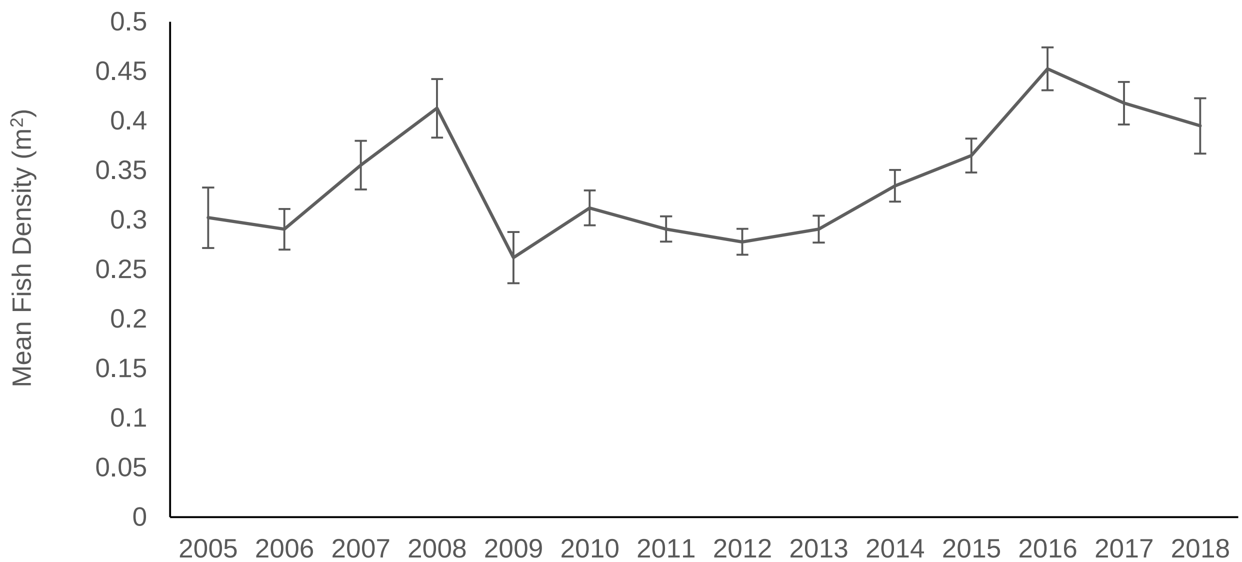

Figure 18. Mean total fish density (m^2) per year from 2005 to 2018. Error bars showing standard error.

3.2.2 Overall Fish Density/Phase

When assessing fish density by phase (January – June / July – December), a bi-annual trend can be seen for most survey years with lower fish densities in the second half of the year (Figure. 19). The fish densities for 2018 once again reflected this long-term pattern, after 2015 - 2017 did not display this trend. Densities were 13.25% (0.37 ± 0.0287) lower for the second phase of 2018 compared to the first phase (0.426 ± 0,291).

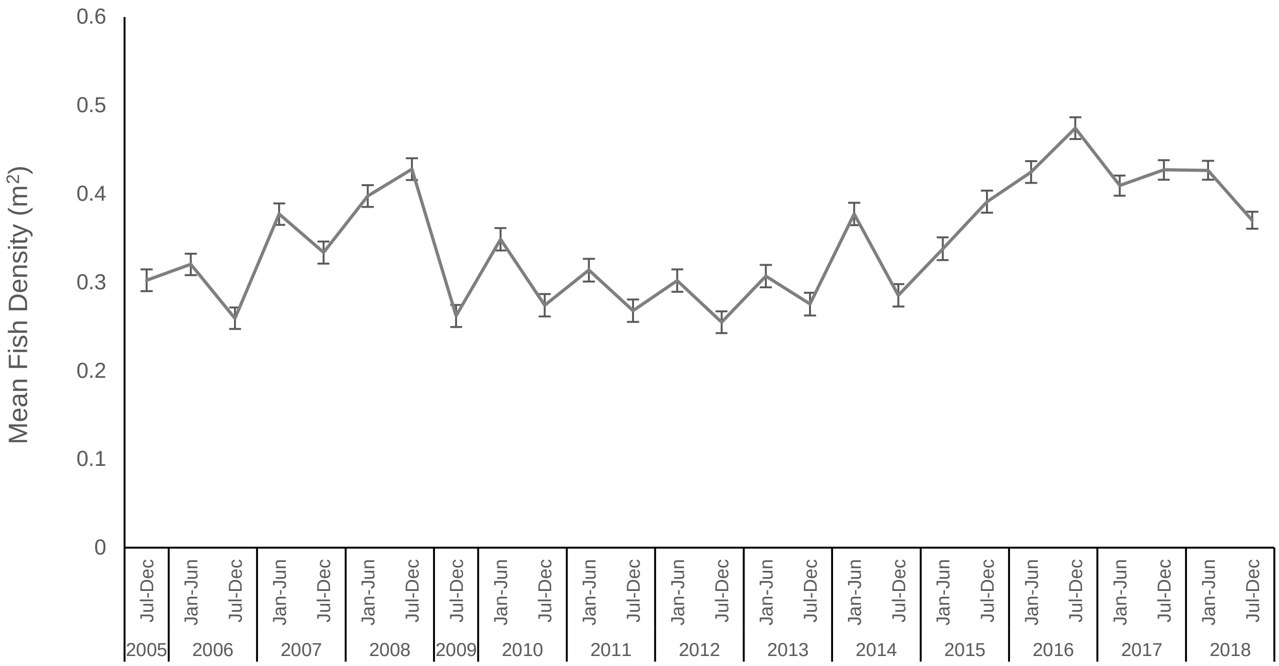

Figure 19. Mean total fish density per survey period for years 2005 to 2018. Error bars showing standard error.

3.2.3 Overall Fish Density / Site

Baie Ternay Centre once more supports the highest density of fish in 2018 (0.766 fish m^2), an increase of 23.9% compared to 2017 (0.618 fish m^2). All Marine Park sites show varying degrees of increase or decrease in fish densities for 2018 apart from BTNE, the only site where fish have decreased substantially by 25.02% over the last survey year (0.519 to 0.482). The other sites within the Baie Ternay Marine National Park have shown little variability in fish density since 2017; BTNW increase by 0.05% (0.518 to 0.519), Anse du Riz / Secret Beach decrease by 1.12% (0.423 to 0.418). For the other survey sites, a clear pattern can be observed with the most exposed / semi-remote sites located around the islands of Therese and Conception continuing to support higher fish densities than sites situated along the coast. Evidently not all semi remote sites showed an increase, with some decreasing by as much as 26% (Therese South). The biggest decrease was observed at Rays Point, with fish densities dropping by 41.5% compared to 2017 (Figure. 20).

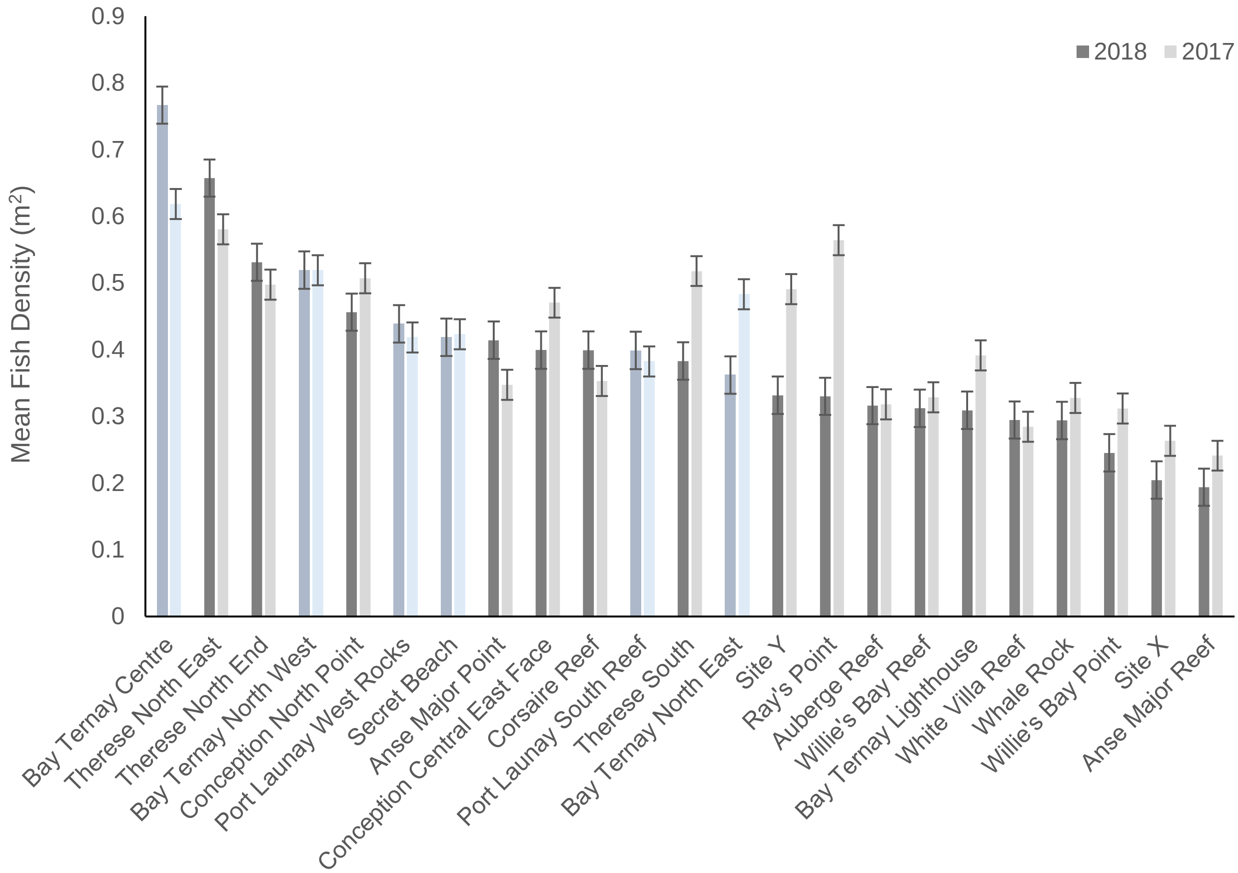

Figure 20. Mean fish densities displayed by survey site, sorted highest to lowest for 2018. Blue bars indicate sites located within a marine park. Error bars show the standard error of the mean.

3.2.4 Reef vs Commercial Fish

Both reef and commercial fish species follow a decreasing trend in 2018. Reef fish density decreased by 5.2% compared to the previous year (0.166 ± 0.01) and by a total of 15.3% since 2016 (0.186 ± 0.01). Commercial fish density decreased by 16,7% compared to 2017 (0.252 ± 0.01) and by a total of 10.9% since 2016 (0.265 ± 0.01). Commercial fish continue to show higher densities than reef fish, a trend that can be observed since 2015. A diverging pattern can be observed since the bleaching event in 2016 where commercial fish display higher densities than reef fish, after following a mostly congruent trend prior to 2016 (Fig. 21).

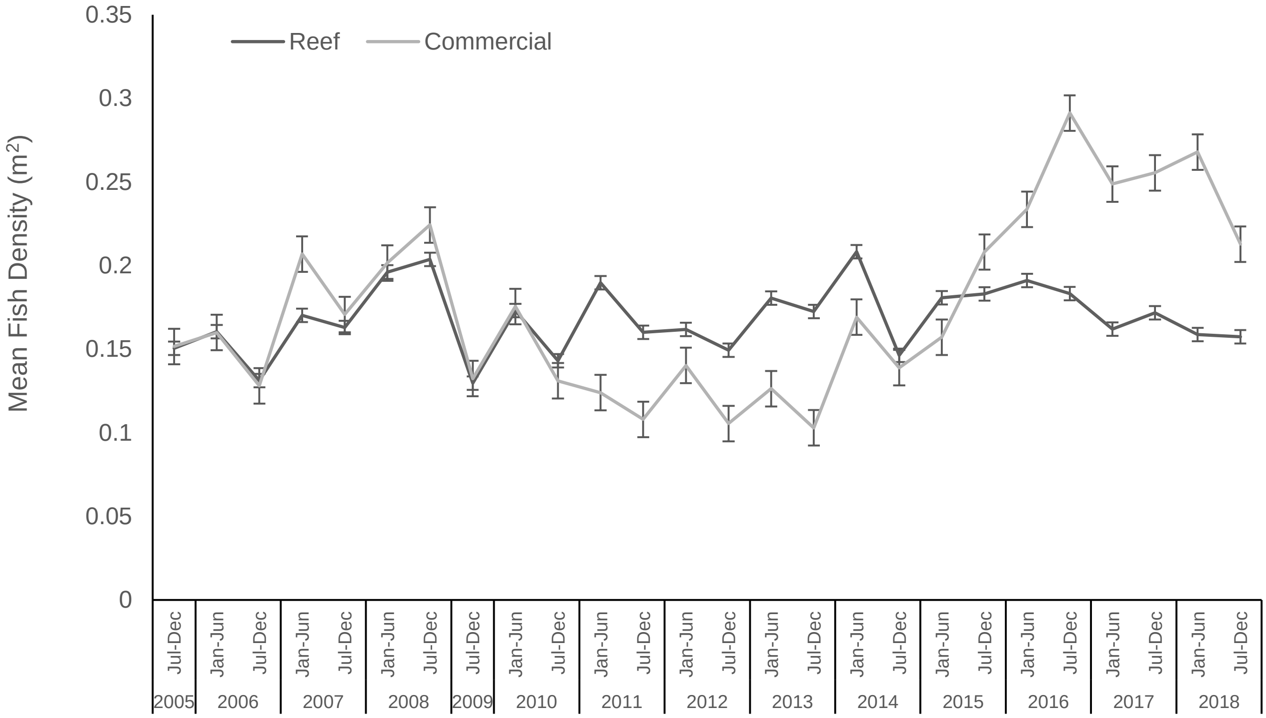

Figure 21. Mean reef and commercial fish density per survey period, for years 2005 to 2018. Error bars showing standard error.

3.2.5 Fish Densities and Substrate composition

Comparing the two prevalent forms of reef substrate around the Seychelles, carbonate and granitic, 2018 displays a major change since the beginning of monitoring. The first major divergence from an otherwise similar pattern occurred this year, with carbonate reefs breaking the downward trend since 2016. Fish densities on carbonate reefs increased by 4.1% in 2018 (0.427 ± 0.05) compared to 2017. Fish stocks on granitic reefs continued to decrease, now being 14.4% lower than 2017 (0.425 ± 0.03) and 18.4% lower than 2016 (0.445 ±0.02) (Figure. 22).

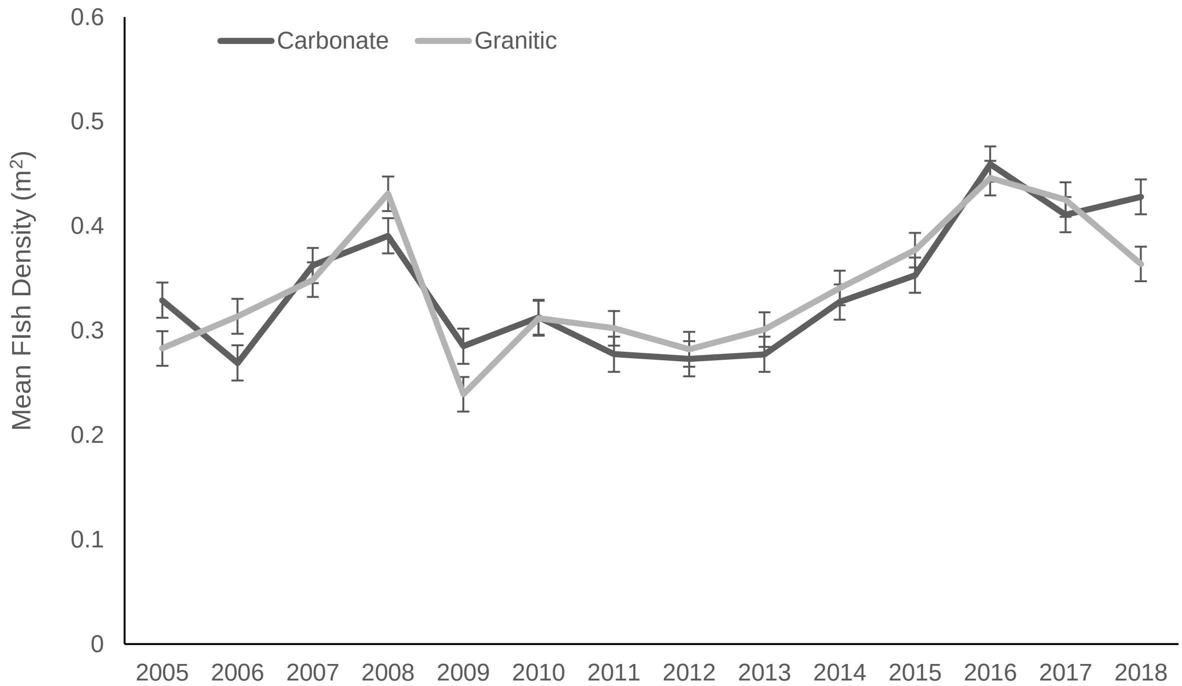

Figure 22. Mean total fish density (m^2) for granitic and carbonate sites from 2005 to 2017. Error bars showing standard error.

3.2.6 Management Strategies

3.2.6.1 Protected vs Unprotected

Protected sites continue to display a higher overall fish density than unprotected sites and show early signs of recovery with fish densities for 2018 increasing by 3.35% (0.489 ± 0.017) compared to 2017 (0.474 ± 0.032). Overall fish density still remains 9.83% lower than 2016 (0.543 ± 0.02). Unprotected sites continue to show a negative trend with fish densities for 2018 decreasing by 9.2% (0.36 ± 0.024) since 2017 (0.396 ± 0.021). Now a total of 14.2% lower than fish densities in 2016 (0.419 ± 0.013) (Figure. 23).

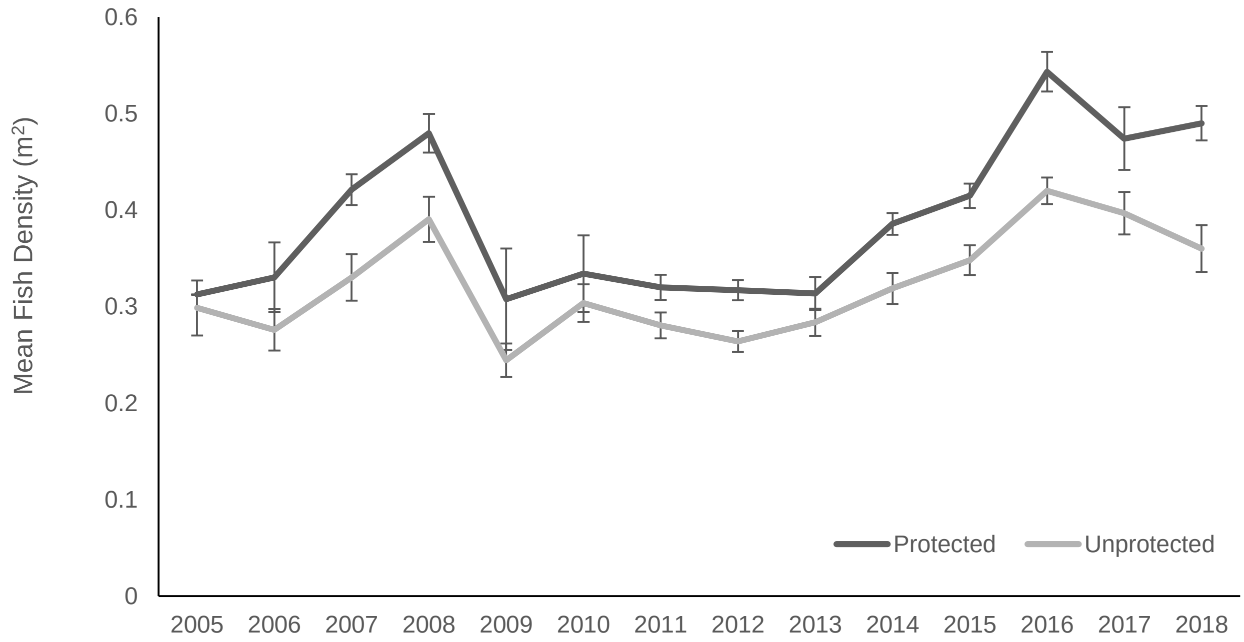

Figure 23. Mean total fish density (m^2) in protected and unprotected survey sites from 2005 to 2018. Error bars showing standard error.

3.2.6.2 Port Launay vs Baie Ternay

Comparing fish densities of the Port Launay (PL) and the Baie Ternay (BT) Marine Park to one another, a similar trend can be observed for fish densities in 2018. Both show a slight increase compared to 2017 with fish densities in Baie Ternay increasing by 3.86% (0.531 ± 0.089) and Port Launay by 4.59% (0.418 ± 0.019). The trend for both Marine Parks returns to the same congruent pattern that has been observed prior to 2016. The difference between the two areas stays at a similar level to previous years with both following a similar trend (Figure. 24).

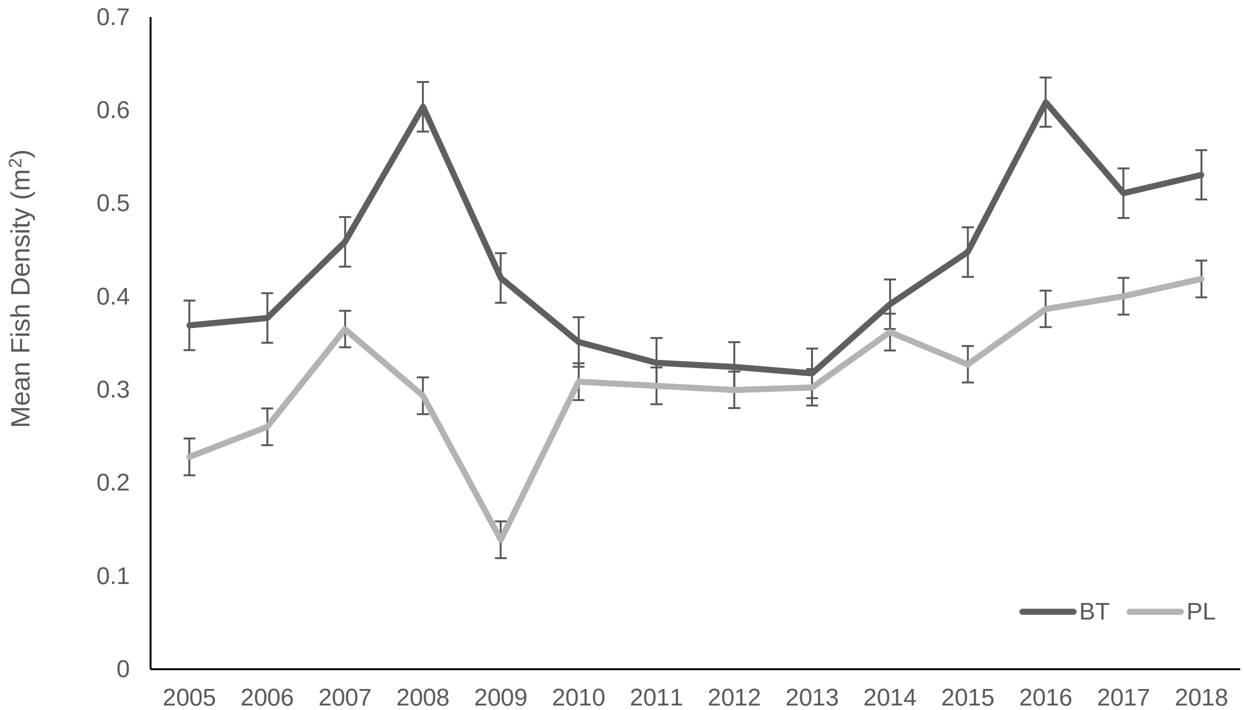

Figure 24. Mean reef fish density (m^2) for Port Launay (PL) and Baie Ternay (BT) protected areas. Error bars showing standard errors.

3.2.7 Comparation between Reef and Commercial fish in and outside the MPA

Commercial and reef fish density follow similar patterns for protected and unprotected areas respectively. Both show an increase inside the MPAs with reef fish densities increasing by 7.1% and commercial fish density by 1.3%. At unprotected sites, both target areas displayed a decrease in fish density with reef fish decreasing by 9.8% and commercial fish density by 9.2% (Figure. 25).

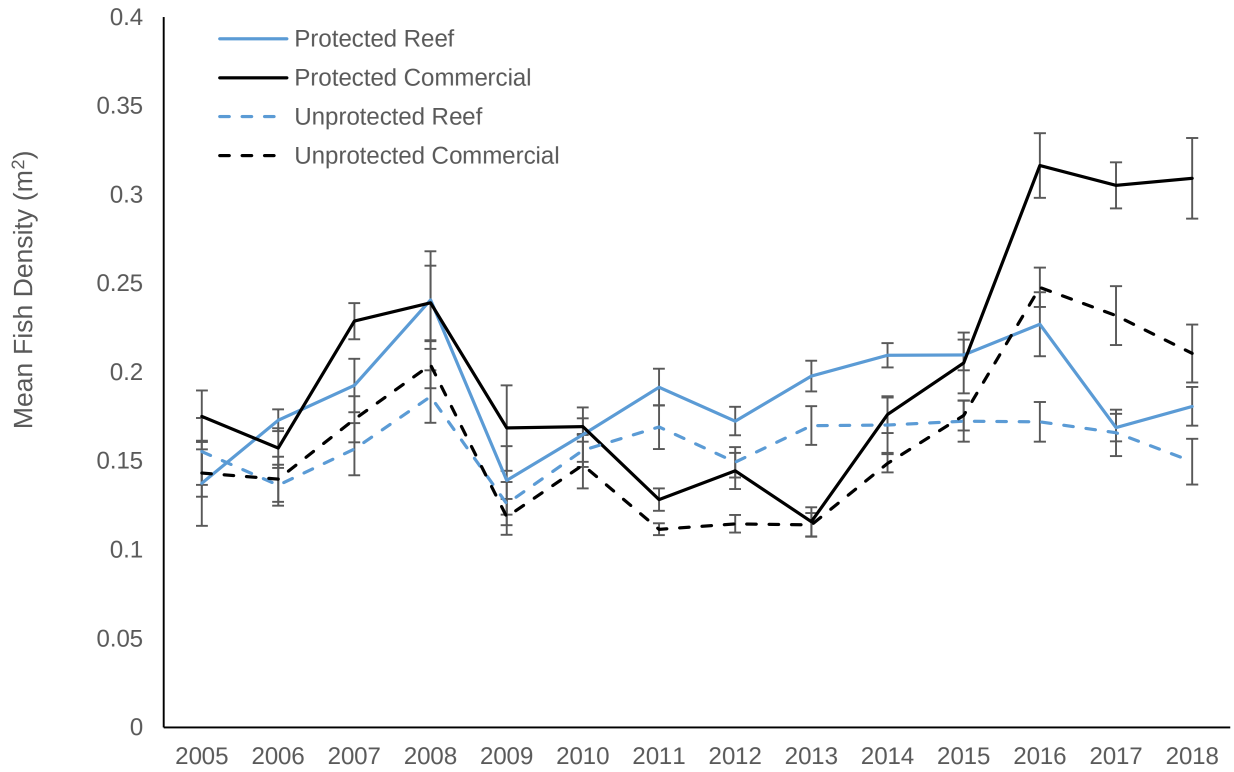

Figure 25. Mean reef and commercial fish density within and outside Marine Protected Areas. Blue color indicates Reef Fish. Dashed lines indicate unprotected areas. Error bars show standard error of the mean.

3.2.8 Density Changes

3.2.8.1 Commercial species density

Except for Scaridae, all commercial fish species show a decline in abundance over the past year after showing a slight increase for 2017 posterior to the bleaching event in 2016 (Figure. 26). For the Scaridae family this was the exact opposite, decreasing after 2016 and then increasing after 2017 by 3.29%. All other families show a significant decrease in density per m^2: Haemulidae has decreased by 35.57%, Lethrinidae by 30.3%, Siganidae by 28.84%, Serranidae by 13.13%, and Lutjanidae by 9.5%. Scaridae remain to be the most abundance family across all survey sites with densities up to 30x higher than other target species.

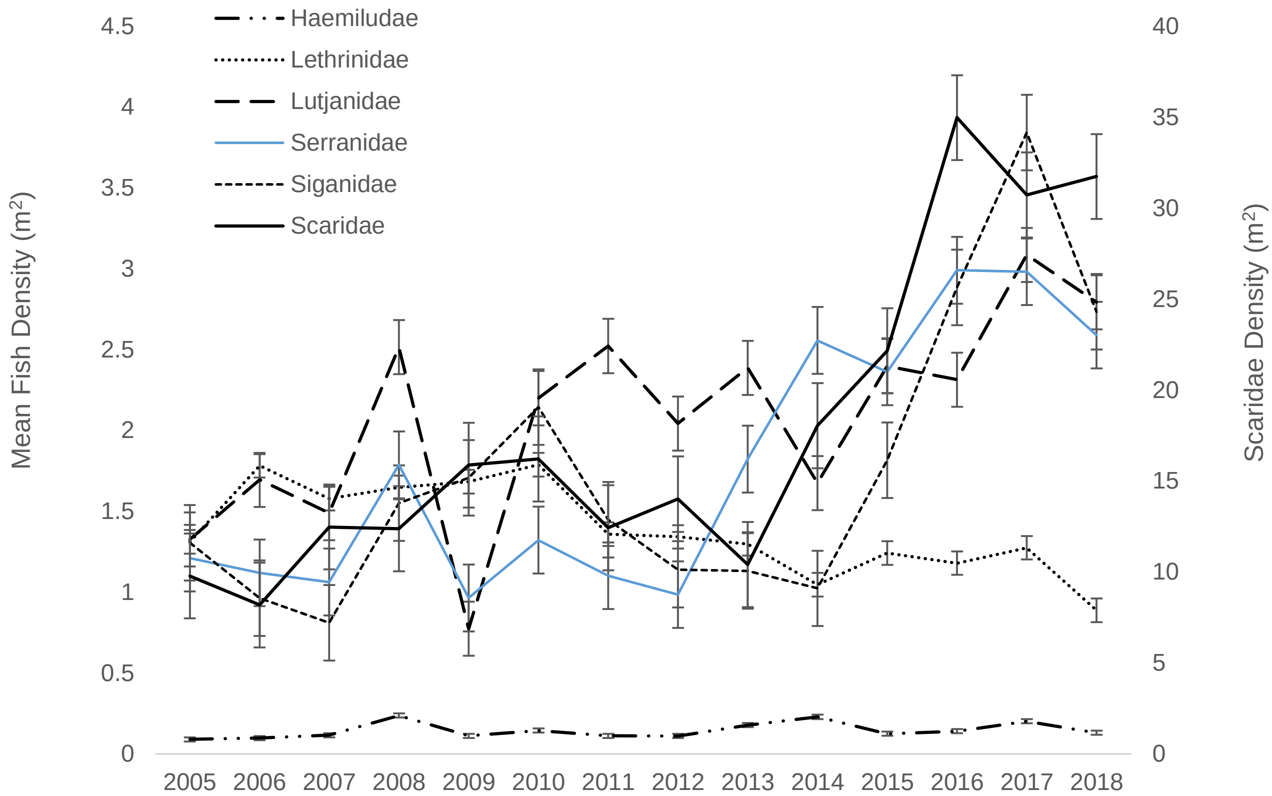

Figure 26. Mean density of commercial fish families from 2005 to 2018. Error bars show the standard error of the mean.

3.2.8.2 Reef fish density

Reef fish display varying changes in density throughout the past year after the bleaching event in 2016. The most noticeable and prominent change occurred in the Chaetodontidae family where a significant decline can be observed for each year. For 2018 their density reduced by 35.1%, now displaying a density 57.71% lower than 2016 levels (Figure. 27).

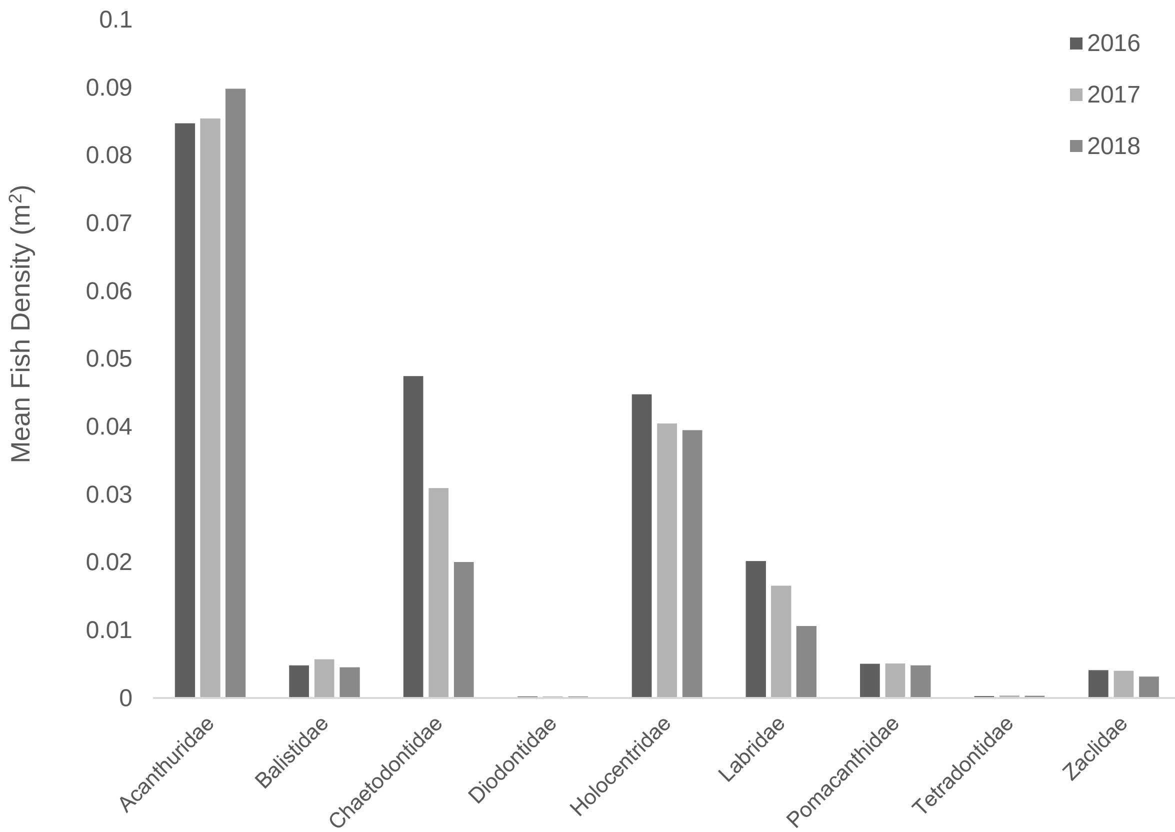

Figure 27. Mean total fish density (m^2) for each target species for 2016 to2018.

3.2.9 Commercial Fish Size Analysis

Overall fish abundance of smaller juvenile fish in-between 0-20 cm is higher at unprotected sites while adult fish sized between 21-50 cm display a higher abundance at protected sites. *Haemulidae* show higher densities at unprotected sites for fish between 0-20 cm, for fish bigger than 20 cm their densities are higher within protected areas. *Lethrinidae* have higher densities of smaller fish 0-10 cm in protected sites whereas a greater number of adult fish 21-40 cm at unprotected sites. *Siganidae* have higher densities of juvenile fish 0-10 cm at protected sites but then display higher densities of sub-adults 10-20 cm at unprotected sites; adult fish >20 cm show higher densities at protected sites. *Lutjanidae* display slightly higher densities of juvenile fish at unprotected sites and significantly higher densities of fish sized between 10-20 cm at unprotected sites. Adult fish of >30 cm display higher densities at protected sites. For Serranidae fish density is higher within the marine protected areas for all size classes bigger than 10 cm (Figure. 28).

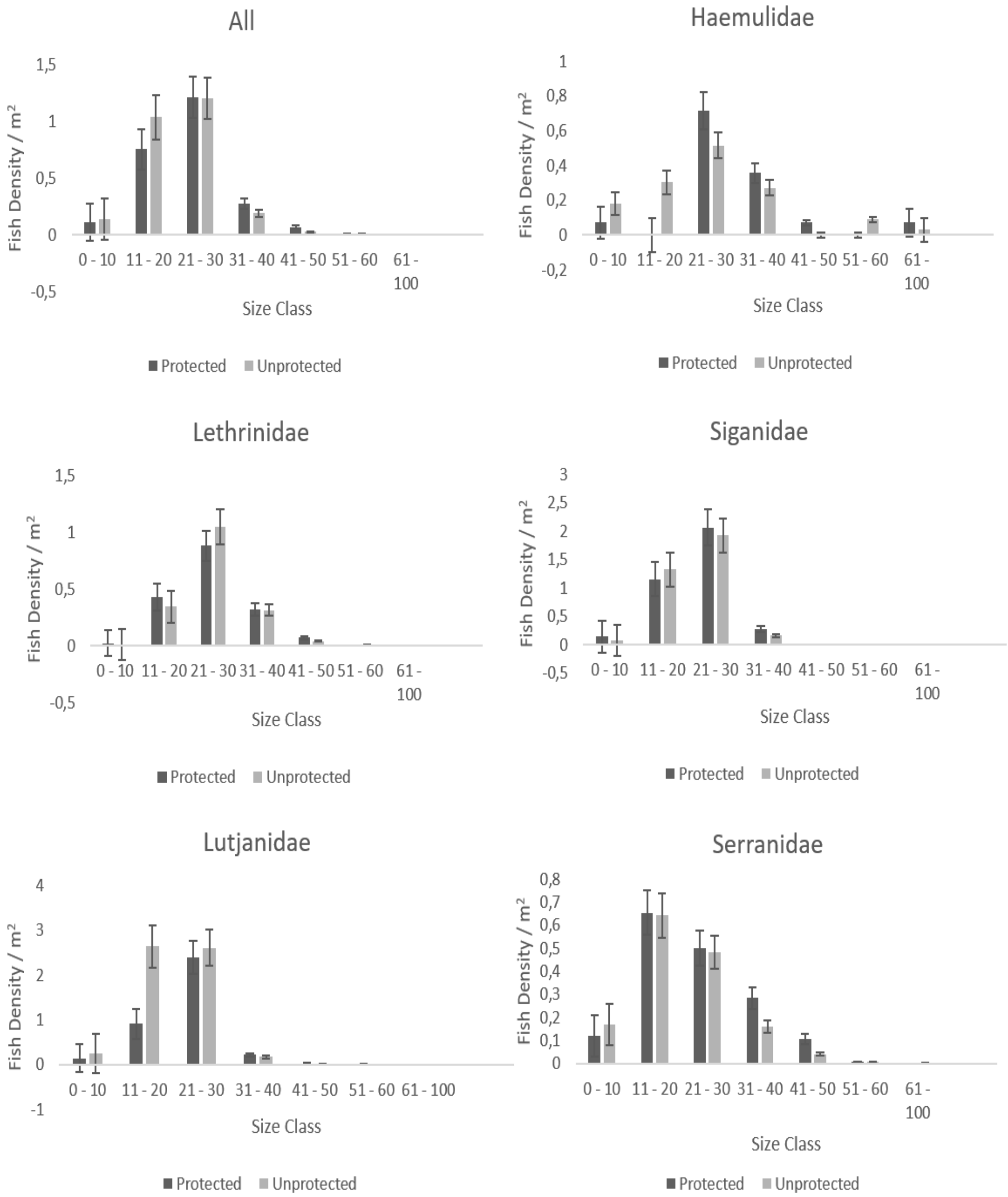

Figure 28. Mean density of commercial fish per size class for 2018. Size classes are displayed in cm. Comparison of marine protected areas and unprotected areas. Size class 100+ cm was disregarded as no species were recorded. Error bars show the standard error of the mean.

3.3 Invertebrates

3.3.1 10m Transects

The 2018 January – June surveys displayed a decrease in densities of Annelida, Arthropoda and Platyhelminthes. Particularly Platyhelminthes showed a sharp decline from 0.0010 (± 0.0006) in 2017 to 0. (Figure 29).

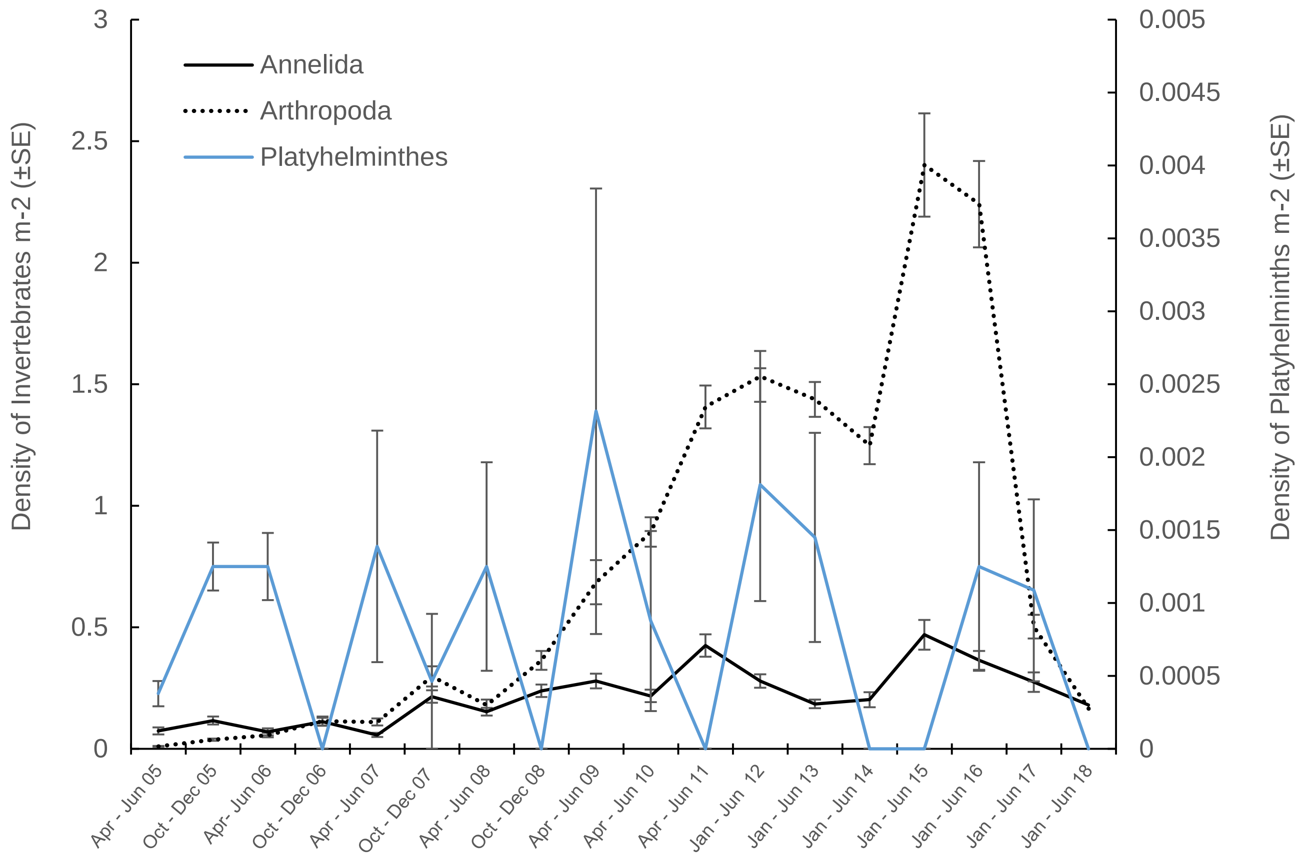

Figure 29. Mean density (individual per m^2) of Annelida, Arthropoda and Platyhelminthes for every survey period from 2005 to 2018 across all survey sites. Error bars show standard error of mean. Note Platyhelminthes are represented by the secondary y-axis.

Black spine urchins and the remaining Echinoderms (excl. black spine urchins) showed a decrease from 2017 to 2018, whereas Mollusca showed an increased in densities, from 0.67 (± 0.06) individuals per m^2 in 2017 to 0.91 (± 0.23) in 2018. All species displayed an increase from the 2005 densities, specifically Echinodermata density, which has increased from 0.76 individuals per m^2 (± 0.1) in 2005 to 1.27 (± 0.10) in 2018 (Figure 30).

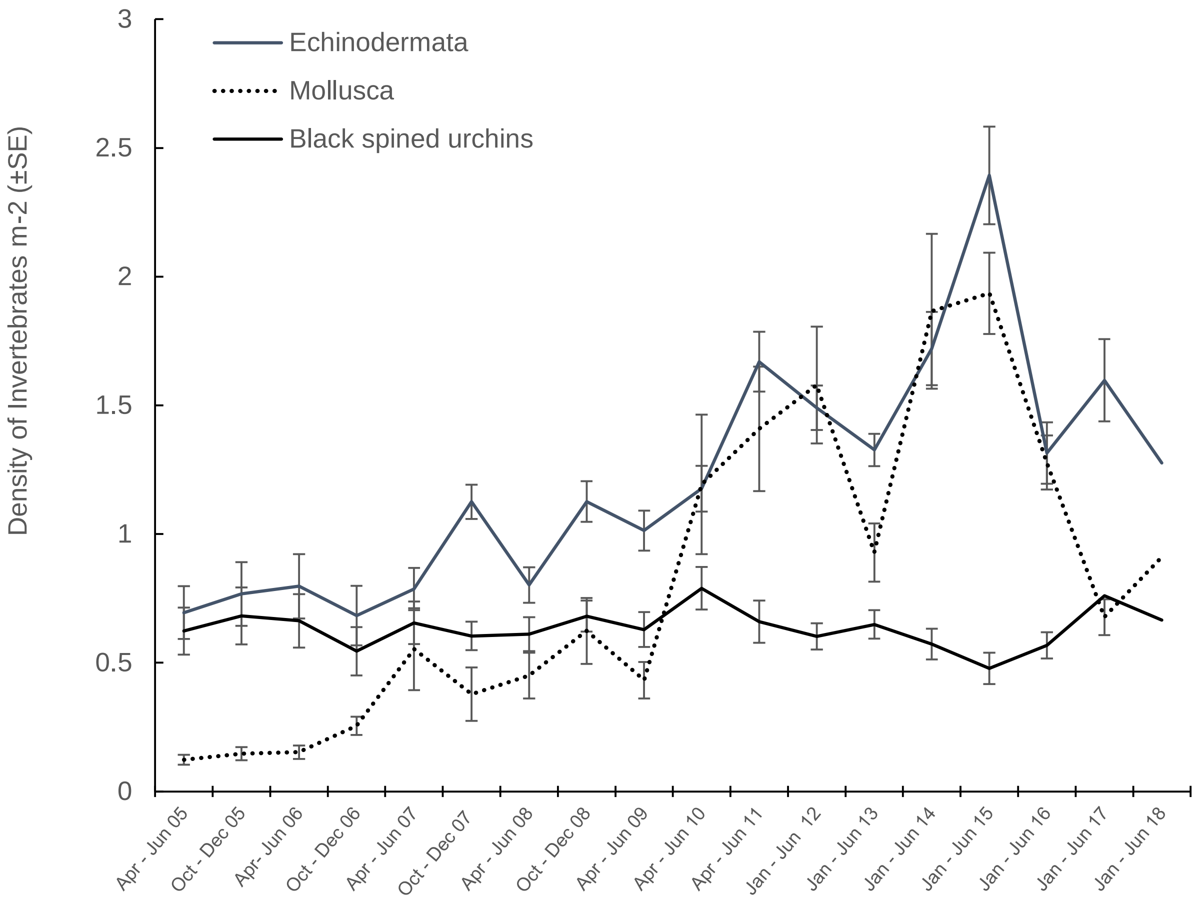

Figure 30. Mean density (individual per m^2) of Echinodermata, Mollusca and black spine sea urchins for every survey period from 2005 to 2018 across all survey sites. Note that black spine sea urchins are not included in the phylum Echinodermata on the graph due to their high abundance.

3.3.2 50m transects

In total 82 surveys were completed throughout 2018, comprising of 42 surveys during the January-June period and 40 during the July-December period, covering a total area of 20,500 m^2. *Echinothrix* sp. displayed a decrease in densities from 0.71 individuals per m^2 (± 0.09) in 2017 to 0.69 individuals per m^2 (± 0.12) in 2018 whereas *Diadema* sp. showed a big increase from 0.14 individuals per m^2 (± 0.05) in 2017 to 0.33 individuals per m^2 (± 0.09) in 2018 reaching one of the highest densities recorded since monitoring started in 2009 (Figure 31).

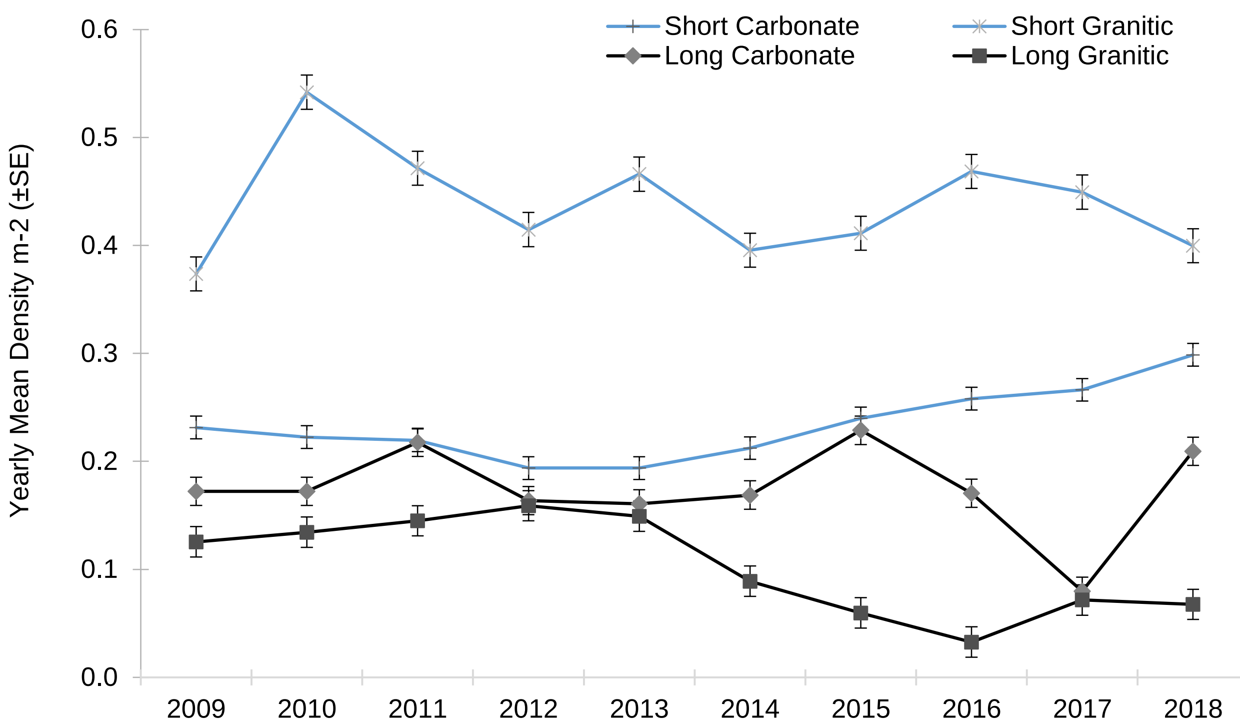

Figure 31. Mean density of individuals per m^2 of short (*Diadema* sp.) and long spine (*Echinothrix* sp.) urchins across all sites surveyed from 2009 to 2018. Error bars show the standard error of the mean.

Mathaes Urchins have always displayed a fluctuation in densities. This year they decreased from 0.015 (± 0.005) in 2017 to 0.003 (±0.0010) in 2018. Pencil Urchins showed a decreased in densities from 0.020 (± 0.006) in 2017 to 0.007 (± 0.0023) in 2018 (Figure. 32).

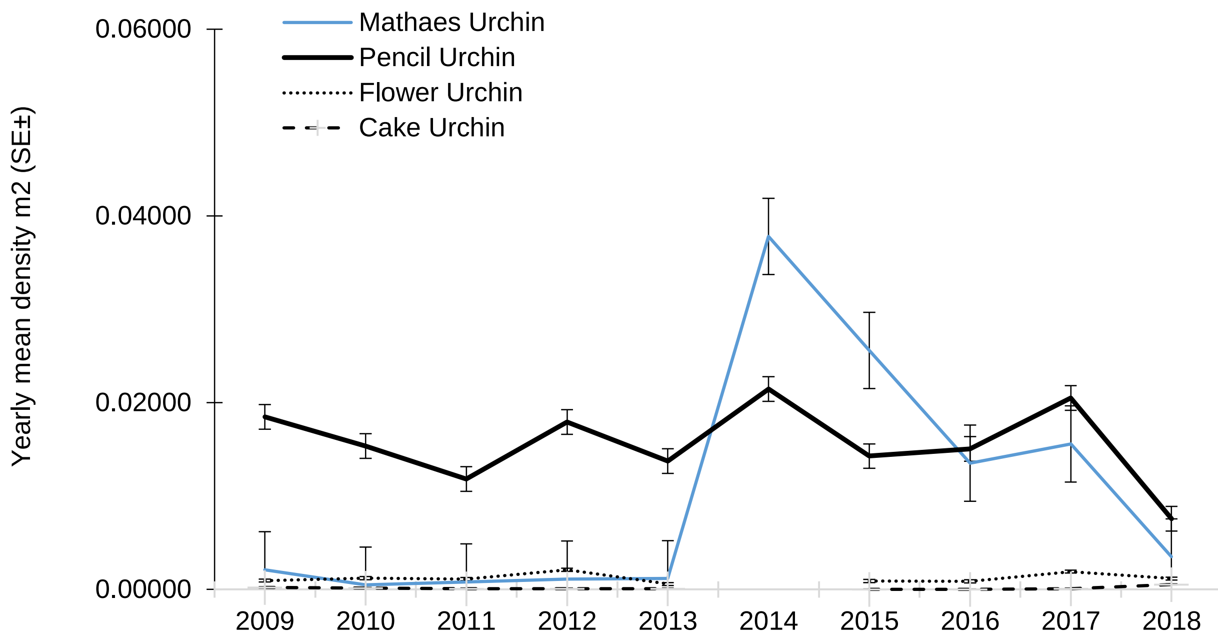

Figure 32. Mean density of individuals per m^2 of Mathaes urchins, pencil urchins, cake urchins and flower urchins across all sites from 2009 to 2018. Error bars show the standard error of the mean. Note 2015 only contains January-June 2015 dataset, as the July-December 2015 dataset had only two surveys and skewed results.

Drupella sp. showed a decline from 2017 to 2018, maintaining levels greater than that of 2009; 0.006 individuals per m^2 (± 0.0045) compared to 0.013 (± 0.0025) in 2018. Cushion and Other sea stars increased from 0.0076 (± 0.0013) in 2017 to 0.0081 (± 0.0013) in 2018 (Figure. 33).

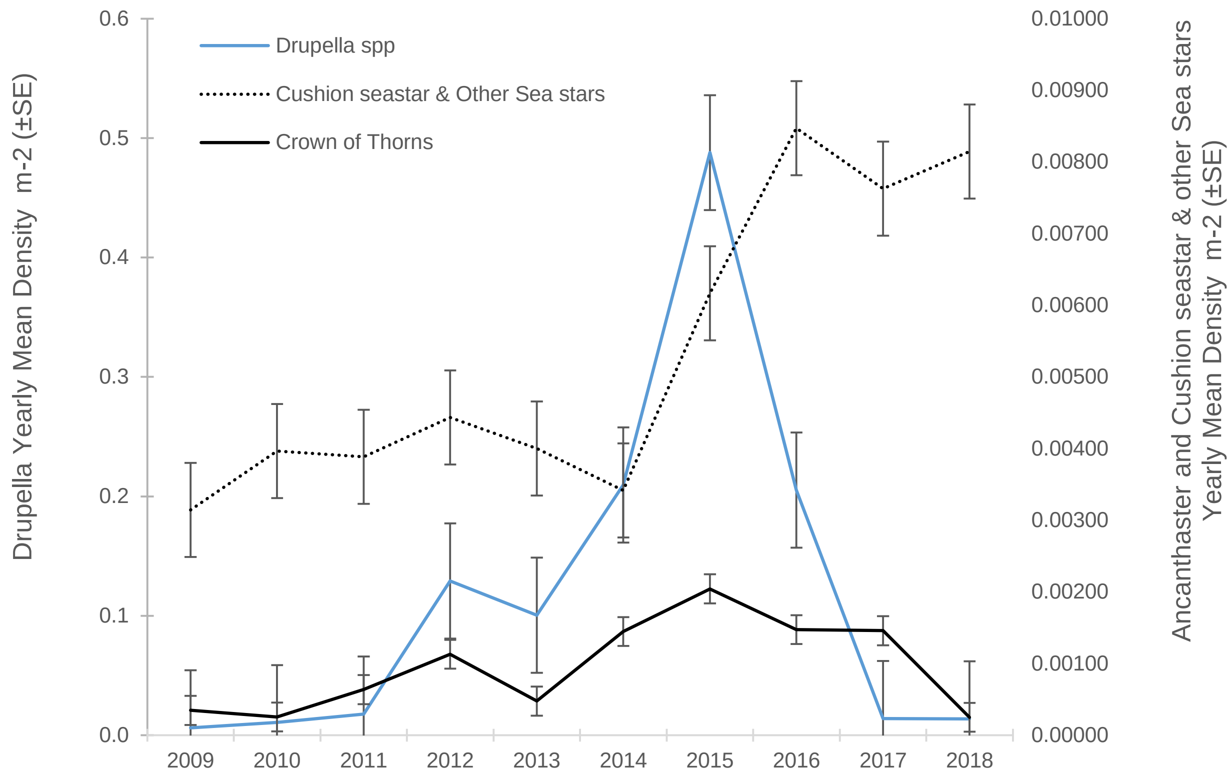

Figure 33. Mean density of individuals per m^2 (±SE) of corallivorous invertebrates surveyed; cushion sea star (*Culcita* sp.) combined with other sea stars, crown-of-thorns (*Acanthaster planci*) and *Drupella* spp from all survey periods across all sites. The densities of cushion sea star (*Culcita* sp.) combined with other sea stars and crown-of-thorns (*Acanthaster planci)* is indicated on the secondary y-axis. Error bars show the standard error of the mean. Note: 2015 data includes reduced surveys.

The mean number of sea cucumbers per site surveyed increased between 2017 and 2018, from 23.31 (± 2.4) to 29.36 (±0.36) respectively (Figure 34). 2018 densities indicated that *Pearsonothurian graeffei* was the most abundant species at 0.02 (± 0.003) individuals per m^2. *Holothuria fuscopunctata, Holothuria* sp. (Pentard) and *Thelenota anax* were not observed in any transects throughout the year (Figure 35). All specific cucumber species except *Bohadaschia* sp. and *Actinopyga mauritiana* displayed an overall increase from 2017 to 2018.

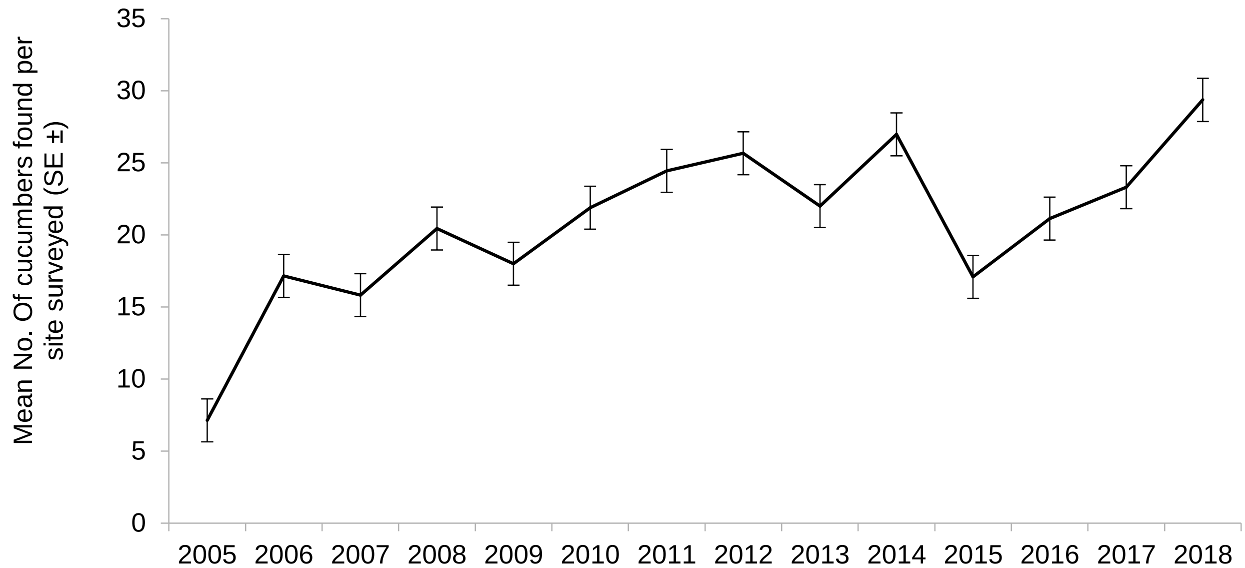

Figure 34. Mean number of sea cucumbers recorded across all surveyed site per year from 2005 until 2018. Error bars show the standard error of the mean. Note, in July-December 2015 only two carbonitic sites were surveyed.

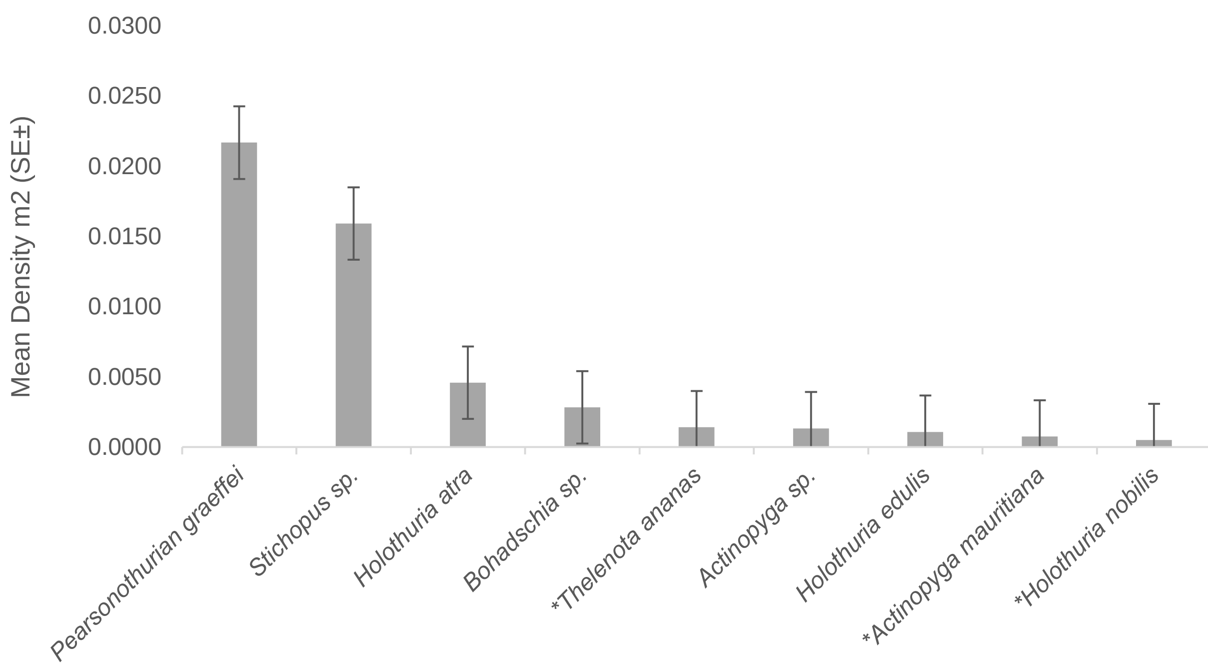

Figure 35. Mean density of sea cucumber taxa across all surveyed sites for 2018 survey period. Error bars show the standard error of the mean. *Holothuria fuscopunctata, Holothuria fuscogilva, Holothuria* sp. (Pentard) and *Thelenota anax were* all excluded from the graph due to no data or minimal data.

4. Discussion

With the completion of the 2018 surveys GVI's data set now represents 14 years of coral reef monitoring activities. Data sets of this size and detail are vital in furthering our understanding of coral reef population dynamics, as well as providing an insight into the mechanisms and trajectory of coral reef recovery following large-scale stochastic events.

4.1 Live Scleractinian Coral cover

The coral bleaching event that affected the Republic of Seychelles in 2016 caused significant hard coral mortality, with a 65% decrease in hard coral cover between 2015 and 2017. This year live hard coral cover increased 12% from 2017.

Continuing the same trend as previous years, the mean percentage coral cover was found to be slightly higher on granitic reefs than carbonate reefs. The highest mean coral cover was found on the granitic site Bay Ternay North East, followed by the granitic site Therese South. Granitic sites offer greater substrate stability than carbonate reefs. The higher abundance of juveniles recorded last year suggests effective survivorship of coral recruits within this habitat (Harris et al. 2014).

Corals from the genus *Acropora* presented a decrease in percentage cover. These fast growing branching corals are highly susceptible to bleaching, a pattern consistent over a wide geographic range (Loya et al. 2001; Baird & Marshall 2002; McClanahan et al. 2004). Despite *Acropora* numbers decreasing this year, branching coral growth forms displayed a big increase in percentage cover. This can be attributed to the increase of the branching form of *Pocillopora*. In the case of massive lifeforms, they displayed a dramatic decrease in overall cover percentage. We are uncertain as to why this decrease happened, as massive colonies are normally found to be more resistant following a disturbance event due to their morphological advances such as tissue thickness and shape-dependent energy transfer efficiency (Loya et al. 2001; Baird & Marshall 2002). We hypothesise that the observed decrease in numbers can be attributed to several factors such as the random selection of the study area which does not necessarily show a clear representation of the amount of massive corals found in the reef. It will be interesting to see if in the next report we can observe a pattern that can give us an idea on the behaviour of these colonies.

4.2 Scleractinian Coral Diversity

Coral genera richness belt transects are useful to cover a wide area of the survey sites and increase the chances of finding rare and hidden coral genera. However, several coral genera have not been recorded at all (*Siderastrea, Coeloseris, Alveopora*, *Symphillia* and *Pectinia*) or were recorded only once (*Halomitra, Oulophyllia,* and *Merulina)*. Although rare, several of these coral genera were present at sites (personal observations) during non-survey dives. Komyakova et al. 2013 showed the correlation of coral species richness and fish species richness and abundance. They suggested using coral species richness as a good indicator of a healthy and diverse reef ecosystem, equally important as coral cover, therefore emphasizing the significance of coral diversity belts as a monitoring method. This year the granitic site Therese South displayed the highest diversity index and the carbonate site Auberge Reef the lowest. There were no records of “unusual” corals for 2018.

4.3 Non-Scleractinian benthic composition

The benthic community composition analysis displayed that the fast-growing and short-lived turf algae (Birkeland 1977) massively decreased in numbers yet still remains the dominant non-scleractinian organism. Macroalgal cover is still comparably very low, which is a potential indication of sufficient herbivorous pressure preventing algae succession (Pratchett et al. 2011). Coralline algae showed an increase in mean overall cover. A possible explanation of this effect can be linked to the decrease in urchin populations thus causing an increase in coralline algae growth.(O’Leary & Mcclanahan 2010).

The increase in algae assemblage suggests that different species of algae are colonising niches now available following the bleaching disturbance, such as bare rock, dead coral and rubble. Growth of algae on these patchy habitats is primarily due to coral cover loss. It is predicted as competition increases and environmental conditions begin to stabilize, diversity will decrease as the original algal pioneer species are displaced (Huston 1994).

Sponges, soft corals, corallimorphs and zoanthids are present in low numbers. Corallimorphs and zoanthids, are found in higher proportions than soft corals and sponges, a trend also observed in 2015 and 2008. As explained in previous reports (G.V.I 2017), these organisms are secondary colonizers that have a competitive advantage over corals, therefore increases in these organisms may impact negatively on reef health competing with coral species for space.

4.4 Scleractinian Coral Recruitment

Scleractinian coral recruitment in 2018 has decreased 22% from 2017. Considering that the increase in percentage of live hard coral cover occurs largely as a result of the growth of recently settled hard coral recruits rather than that due to ongoing growth of remnant corals (Engelhardt 2004), a decrease in recruit densities indicates a negative impact for the recovery of the reef. At present a clear pattern of increased coral recruitment, as identified following previous disturbance events, is not evident. It will be interesting to observe coral recruitment patterns in the upcoming years, to provide a better understanding of how they adapt to the ever-changing conditions.

Porites has been the dominant Scleractinian coral recruit genera in terms of overall abundance since monitoring began, with the exception of *Favites* in 2016. Both genera are known for their temperature stress tolerance and resilience to disturbance events (Baird & Marshall 2002). As such, *Porites* and *Favia* appear to be less affected by bleaching than other surveyed genera with different life histories, for example fast-growing genera in the families of *Acroporidae* and *Pocilloporidae* (Loya et al. 2001; McClanahan et al. 2004). This is a phenomenon reported in other studies in the Seychelles and Western Indian Ocean Region after the previous 1998 bleaching event (Hagan & Spencer 2008; Tamelander 2002; McClanahan et al. 2004).

The important and once dominant genera *Acropora* was heavily impacted by the 2016 bleaching event and recruitment this year is showing a continued decreasing trend. This fast-growing genera is a key structural component of healthy and diverse coral reef ecosystems in the Indo-Pacific (Engelhardt 2004). With its high morphological diversity and structural complexity known to promote ecological relationships with other reef organisms, including many invertebrates and fish species (Engelhardt 2002). *Acropora* recruitment and survival is important for the recovery of these reef communities. In 2018 the previously dominant genera *Acropora* was recorded at its lowest value since records began in 2005.

Coral recruit diversity and genera evenness increased slightly from last year’s records. These results still indicate a downward trend of coral recruit diversity at surveyed sites throughout the years, however statistical analysis is required to correctly quantify this observation.

Mean coral recruitment increased at the majority of the 18 surveyed sites in 2018. Granitic sites had higher coral recruitment than carbonate. Specifically, the granitic site Whale Rock and carbonate site Baie Ternay North West had the highest and lowest mean coral recruitment respectively. Considerable differences have been recorded in the rate of recovery posterior to the 1998 bleaching event between carbonate and granitic reefs (Payet et al. 2005). Granitic reefs are thought to have greater resilience due to their substrate stability in comparison to carbonate reefs. Such sites are more suitable for coral larvae to settle, and subsequently have lower post recruitment mortality due to less moving rubble from wave action, less sedimentation, and higher levels of invertebrates, such as *Diadema* spp. and *Echinotrix* spp., grazing on algae competing with new recruits for space and light. Finally, sites located outside the Baie Ternay and Port Launay Marine National Parks showed higher average coral recruitment than sites within protected areas. This is likely due to less sites surveyed within the localised marine parks (n=5), in comparison to the sites surveyed outside these areas which are spread over a larger geographical range, likely to skew any conclusive result.

Depth specific density results seem to conform to previous norm with deep surveys showing higher densities compared to shallow. Variation in coral recruitment rates at this spatial scale may be due to changes in light intensity with both depth and orientation, differences in algal biomass, sediment and grazing intensity and wave action (Babcock & Mundy 1996). Recruit size class analysis showed that both 0 - 2 cm and 2.1 – 5 cm categories have decreased in overall mean density of coral recruits in 2018. These observations could indicate a decrease in spawning adult colonies and/or increased mortality for newly settled larvae (Engelhardt 2002).

In conclusion, the inner islands of the Seychelles are mostly reliant on self-recruitment. This coupled with the fact many coral species have small and disconnected brood stocks, especially fast-growing branching species, coral reef recovery is slowed down (Graham et al. 2006). Monitoring reef recovery by documenting new recruitment densities, survival of viable adult colonies alongside other important ecosystem health indicators (Bellwood et al. 2004), will prove crucial. These measures are critical in advising effective management of marine ecosystems, thus enabling informed responses to future disturbances, such as crown-of-thorns starfish outbreaks, storms and repetitive bleaching events (Babcock et al. 2003).

4.5 Overall Fish density trends

Assessing the changes in fish stocks after a severe disturbance event is crucial to deepen our understanding about coral reef systems and their associated species (Yusuf and Ali, 2004; Graham, Nash and Kool, 2011). While analyzing population fluctuations of reef fish (i.e. grazing species and herbivory species) can give an insight about the state of the reef itself, it is also important to consider possible changes to commercially important predatory fish species that rely on the reefs as habitat and nursery grounds (Sandin *et al.*, 2008; Hempson *et al.*, 2018). of the reefs and their associated organisms, but also their recovery.

Fish density has been found to correlate with live coral cover (Bouchon-Navaro and Bouchon, 1989; Samoilys *et al.*, 2018), therefore a major decrease in live coral cover following the bleaching event of 2016 should be reflected by an overall decrease of fish density, as observed between 2008 and 2009 (Figure.18). As the initial decline in fish density was smaller than expected for 2017, it is hard to hypothesize how fish populations will change and develop in 2018. Fish populations are known to show a lag effect posterior to changes to coral cover (Motta, 1989; Graham *et al.*, 2007; Mcclanahan, 2009), with densities often declining long after the reef starts recovering and returning to pre-bleaching coral state. Our findings reflect this, as densities are still following a negative trend even though overall coral cover began to increase in 2018 (Figure. 5). The lag-effect is something that was proposed by Graham et al. in 2007 and has been recorded amongst multiple reefs with fish populations either slowly reflecting changes to the reef with an "offset reaction" or taking longer to recover after a severe disturbance event. Coral bleaching affects the reefs in a multivariate way, taking away habitat and protection for many fish and destroying the direct food source for others (Graham, Nash and Kool, 2011).

Fish densities were previously found to follow a general pattern of higher numbers in the first half of the year and slightly lower densities for the latter half of the year for most of our survey periods (Figure. 19). A return to this pattern can be observed for 2018, which hints at a return to normal processes and at first signs of recovery after the bleaching event.

4.6 Comparing commercial and reef fish densities

Different fish species take varying amounts of time to reflect changes if the ecosystem around them is being impacted negatively (McClanahan *et al.*, 1999; Hempson *et al.*, 2018). It was found that commercial fish density was affected more severely than reef fish density (Fig. 21). For commercial fish species, not only is their habitat destroyed by a bleaching event, but in many cases their food source is indirectly reduced when the smaller fish upon which they prey decline due to a lack of habitat (Westera, Lavery and Hyndes, 2003).

Reef fish species, which are either corallivorous or grazers, are directly affected in various ways and often reflect changes swiftly, as their habitat and food source is reduced immediately after a bleaching event (Gregson *et al.*, 2008; Hempson *et al.*, 2018). Once the density of reef fish declines over time, it can be expected that commercial fish density will subsequently decline, due to the ability to cope longer with suboptimal conditions without showing extensive changes in population structure (Spalding, M.; Jarvis, 2002). This is reflected in our findings, as the decline in reef fish density appears to start leveling out for 2018 (Figure. 21). Commercial fish density showed an incline for the first half of the year but then displays a sharp decline from July onwards.

As it is difficult to determine fine scale patterns on such a broad scale, we assessed fish density by including various factors such as substrate, protection status of the survey sites and trends between family, to see if we can narrow down the processes affecting the fish populations along Mahé's northwest coast.

4.7 Reef and substrate type

GVI surveys 11 carbonate and 13 granitic reefs along Mahé's northwest coast. Since the beginning of monitoring in 2005 both reef types followed a similar pattern in terms of fish densities (Figure. 22). After 2017, a divergent pattern can be observed with fish densities on carbonate reefs showing early signs of recovery, while granitic sites appear to be declining at an even steeper rate.

This is in contrary to the findings of our coral cover surveys, where live coral cover on granitic reefs has increased by almost three times as much as carbonate reefs, indicating that these changes in fish density may not be directly linked to the type of reef substrate. Changes in coral cover were found to directly affect fish abundance and density (Feary, McCormick and Jones, 2009), however our results do not confirm this as coral cover and fish show differing trends.

It can be argued, that of the 6 protected sites surveyed, 4 sites are carbonate reefs, which may influence the observed trend due to higher fish densities and earlier recovery found at protected sites. However, many offshore sites with naturally high fish densities (i.e. Therese Island) consist of granitic sites once again differing from our observations and not giving us a clear answer as to why we observe these patterns.

As only 6 of GVI's 23 survey sites are protected, the other sites are affected by a plethora of different disturbances such as fishing, direct anthropogenic disturbance by boats, snorkelers and divers and potential terrestrial runoff and pollution, all of which may directly or indirectly influence the fish densities at each site but are outside the scope of our survey efforts (Francis, Nilsson and Waruinge, 2002; Gilmour *et al.*, 2013; Samoilys *et al.*, 2018). It is important to mention that both Marine Protected Areas see a lot of boat traffic and snorkelers and scuba divers however here we believe that the lack of fishing pressure positively affects fish stocks within the area.

4.8 Influence of Management Strategies and semi remoteness as a surrogate

One of the biggest factors affecting changes to fish densities after a severe disturbance events is the protection status of the reef, with protected reefs often showing earlier signs of recovery than unprotected sites (Westera, Lavery and Hyndes, 2003; Wielgus *et al.*, 2007; Jörgensen, Martin and Burt, 2015).

An overall comparison of mean fish densities within and outside protected areas showed that protected sites start displaying early signs of recovery with increasing densities (Figure. 23), whereas unprotected sites are showing a further decline. A site by site comparison of all surveyed sites for 2018 showed that almost all sites located within MPA's had increasing fish densities compared to 2017 (Figure 20).

The survey sites located inside the two marine parks showed an incline for both reef and commercial fish density, indicating that protection measures in fact facilitate an earlier recovery of the reef and its associated species from severe pulse disturbance events, like other MPAs around the world (Ledlie *et al.*, 2007; Graham, Nash and Kool, 2011). This trend can be observed not only in both marine parks, Baie Ternay and Port Launay (Figure. 24), but also for both reef and commercial fish species whilst both their densities are still declining outside of the protected areas (Figure. 25). The fish densities inside the Port Launay marine park show a particularly interesting pattern, as no decline in fish density is observed after the bleaching event, retaining healthy fish populations throughout. This is a great example of how both marine protected areas not only facilitate an earlier onset of recovery after a coral bleaching event, but can also protect the fish stocks to an extent that they remain unchanged throughout. In a time of severe weather anomalies and when a plethora of anthropogenic impacts are affecting coral reefs, effective management strategies are crucial for the protection of reef ecosystems and of healthy and sustainable fish stocks.

Fish densities at unprotected survey sites continue to decline in 2018, showing no signs of recovery. It is hypothesized that this trend is the result of a lag effect, where fish densities show a prolonged decline following a severe disturbance event, taking more time to reflect changes and adapt to the new habitat than corals and other marine organisms.

One other point to consider here is the semi-remoteness of the islands off Mahé's northwest coast. Therese and Conception Island are not designated Marine Protected Areas, however with their offshore locations, which are often subjected to strong currents and significant wave action, they are less accessible to fisherman at certain times of the year. It can therefore be hypothesized that their distance from the shore may be a surrogate for protective measures, retaining healthy fish populations independently. The fish densities around the islands remain some of the highest surveyed, with the sites around Therese only topped by Baie Ternay Centre in mean overall density. In many cases extreme remoteness has been found to act as a buffer in times of disturbance. Extremely remote reefs such as Wake Atoll or Diego Garcia were surveyed to see how quick these would recover after bleaching or hurricanes in the absence of any anthropogenic impacts. Coral communities around these islands displayed extraordinary recovery potential with the reef communities returning to pre-disturbance levels only a few years after the event (Riegl and Piller, 2003; Williams *et al.*, 2013). Healthy fish communities, the lack of fishing and the lack of direct pollution, all of which is granted for sites situated inside marine protected areas, were found to play a major role facilitating the recovery of the reef (Williams *et al.*, 2013). Therese and Conception with their retained high fish densities may help the reef return to pre-bleaching coral cover levels faster than sites with more anthropogenic disturbance. Retaining healthy fish populations is crucial for coral reefs that have been affected by large-scale structural changes. Many reefs have fallen into a macroalgal dominated or sponge and zoanthid dominated state, due to overfishing and a lack of grazing species to keep algal levels in check and to keep coral from being outcompeted.

4.9 Families density trends

Commercial fish species show very complex changes especially for 2018 and 2017. Whilst many species showed an initial incline in 2017 after the bleaching event (Siganidae, Lutjanidae, Haemulidae, Lethrinidae), others initially declined (Scaridae, Serranidae) (Figure. 26).

These patterns were reversed for 2018, hinting at a species-specific lag effect reflecting the changes to the coral reef later than the coral and invertebrate organisms. Scaridae still remain the most abundant family surveyed with extremely high densities across all survey sites. Their densities are remaining about 30x higher than other commercial fish species. As grazers, this change in densities was expected after the bleaching event when algal cover on the reef was peaking due to the freed substrate.

With all commercially important fish species but Scaridae displaying a decline in population density, fishing pressure needs to be closely monitored around north west Mahé. For groupers and snappers, some of the key species of the local artisanal fishing sector (Clifton *et al.*, 2012), numbers need to be monitored closely, as a reduction in their population could directly affect reef health, productivity and the ecosystem structure, with the two species acting as the top predators on many reefs in the region (Rogers and Beets, 2001). The initial incline of commercial fish species, especially predatory ones, can be attributed to the reduction of the overall coral cover after 2016, making Chaetodontidae, Holocentridae and Labridae more vulnerable to predation as they are known to use the high topographical complexity of the coral for protection (J., S. and P., 2008; Pisapia, Cole and Pratchett, 2012).

Reef fish patterns for 2018 have developed as expected with almost all species decreasing in density apart from the Acanthuridae, the primary grazers on the reef. The Chaetodontidae family showed a profound drop of more than 35% compared to the previous survey year with densities dropping to one third of pre-bleaching levels compared to 2016. As many species of this family are corallivorous, the pattern that we observed over the last three years was foreseeable. With the reef now showing early signs of recovery and coral cover increasing for 2018, it will be interesting to see how this affects the corallivorous Chaetodontidae species in 2019 and whether the lag effect continues or fish densities start increasing.

4.10 Size differences of protected and unprotected areas

Size estimation is crucial when assessing maturity of fish and healthy fish stocks. Our surveys enable us to analyze and compare the abundance of different size classes for our commercially important fish families. When plotting the sizes of fish within protected areas compared to non-protected areas, it was found that protected sites harbor higher densities of all adult commercial fish (Figure.28). The reduced fishing pressure positively affects size of fish species living within the protected areas. Mature individuals have a chance to reproduce and therefore fish populations can be retained with healthy predatory prey levels, often directly affecting community structure and aiding overall reef health.

We are disregarding the overspill effect, as no clear trends can be observed comparing sites closer to and further away from protected areas. Semi remoteness on the other hand is an interesting area to investigate. It gives us a chance to see how reefs located further away from anthropogenic impacts are coping with a severe disturbance event and directly compare this to protected and unprotected but less remote sites.

The trends in fish densities that have been observed for 2018 are overall positive. Whilst the hypothesized lag effect has occurred, sites situated within protected areas have shown early signs of recovery, underlining how crucial protection is to retain healthy fish stocks. Both marine protected areas display positive trends for fish densities for 2018 and the management strategies put in place by the Seychelles National Parks Authority appear to be facilitating a healthy coral reef environment. When assessing the marine life in protected areas, many are often deemed a failure or appear to not work as well as hoped (Francis, Nilsson and Waruinge, 2002; Indab and Suarez-Aspilla, 2004; Halpern *et al.*, 2006). With positive initial trends for coral recovery and certain fish densities inside the surveyed MPAs, both marine national parks appear to be aiding coral reef recovery following the severe disturbance event.

4.11 Invertebrates

Invertebrates have been studied as biological indicators within terrestrial and aquatic ecosystems extensively, including coral reef habitats (Uthicke 2001). Their significance lies in their interactions with the reef habitat, and species density may reflect changes in reef composition and structure. Densities of surveyed invertebrates from the 10 m belt transects in 2018 show an overall decrease in all surveyed invertebrates from 2017 except for the group Mollusca. The decline in the density of Arthropoda from 2017 to 2018 can still be attributed to a loss of habitat, a lack of structural complexity as a result of hard coral cover loss from the bleaching event (Graham et al. 2007). The decrease in all other invertebrate taxa can also be attributed to the overall low coral cover. A decline in coral cover naturally leads to a reduction of certain invertebrates, which have a high affinity with coral and therefore suffer from a loss of food, productivity and rugosity (Heck & Wetstone 1977).

The survey list for invertebrates on the 50 m belts focuses on commercially important invertebrates and key species, which indicate ecosystem change. Short spine urchins and long spine urchins were the most commonly observed invertebrate taxa, excluding the sea cucumbers. Black spine sea urchins are keystone herbivores in coral reef systems and control benthic algae populations. Short spine *Echinothrix* spp. densities have decreased slightly from 2017 levels, but population density appears stable. Long spine urchins, *Diadema* spp., increased significantly from 2017 densities, which can be attributed to the increasing algal growth observed (Dudgeon et al. 2010). *Drupella* spp., an obligate corallivore, displayed a slight decline from the 2017 records. This indicates that live coral cover, specially the branching lifeforms, are still at a low density level (Dudgeon et al. 2010). Other sea stars displayed increased densities. These taxa include any species other than those specifically surveyed, which have various feeding styles, behaviors and responses to bleaching events. This makes it difficult to determine why such an increase has occurred and hinders predictions on future population trajectories. Mathaes urchins and pencil urchins had a dramatic decrease in densities for 2018, the drivers behind this trend remain unclear and need to be addressed further.

Sea cucumbers displayed a strong recovery from the dramatic decline of 2015 (this decline could be a result of the limited surveys conducted in 2015 and is not representative of other years). The overall abundance of sea cucumbers, mainly *Pearsonothurian graeffei* and *Stichopus* sp. reached its highest point since the start of monitoring in 2005, correlating with the decline of hard coral cover and consequent loss of habitat complexity. Abundance and species richness of invertebrates have been found to be affected significantly more by substrate complexity rather than live coral cover (Nelson et al. 2016).

5. Additional Ecosystem Monitoring

5.1 Crown-of-thorns sea stars (COTs)

Outbreaks of the coral predator, the crown-of-thorns sea star (*Acanthaster planci*) were first reported in 1996 and were active until 1998, when the reefs suffered from the bleaching-induced coral mortality (Engelhardt 2004). Normal density levels are less than one individual per hectare (Pratchett 2007) and in these numbers *A. planci* can benefit coral diversity by feeding on the faster growing corals such as *Acropora* and *Pocillopora*; its preferred prey genera (Pratchett 2007) and early colonisers of degraded reefs that out-compete the slower growing corals (Veron 2000). In high numbers however, the level of competition for food drives the sea star to eat any species of corals which can severely degrade reefs and reduce coral cover to as little as 1% (CRC Reef Research Centre 2003). The causes of outbreaks are still not completely understood; it may be connected to overfishing of *A. planci* predators, such as the giant triton shell, which is popular with shell collectors, or to natural fluctuations (CRC Reef Research Centre 2003). The most widely accepted theory is that increased nutrient levels in the ocean could be an influential factor, originating from agricultural, domestic or industrial sources. *A. planci* are surveyed as part of the invertebrate abundance and diversity belts and incidental sightings are also documented after every dive.

Incidental COT sightings

In 2018 we had 134 COT sightings across all our diving sites being Baie Ternay Centre and Baie Ternay North East the locations with more sightings with 18 individuals each. (Table 2). Further analysis cannot be performed as the data is qualitative and not standardized (data is dependent on variables such as; individual divers, site, dive purpose).

Crown of Thorns	**134**
Anse Major Point	5
Anse Major Reef	3
Auberge Reef	3
Baie Ternay Lighthouse	6
Baie Ternay Reef Centre	18
Baie Ternay Reef North East	18
Baie Ternay Reef North West	9
Conception	2
Conception Central East Face	3
Conception North Point	8
Port Launay South Reef	5
Port Launay West Rocks	16
Site X	5
Site Y	11
Therese North End	1
Therese North Point	1
Whale Rock	4
White Villa Reef	1
Willie's Bay Point	12
Willie's Bay Reef	3
Grand Total	**134**

Table 2. Total number of Crown-of-Thorns sightings recorded for each site in 2018.

5.2 Sightings of other fauna

Since 2006, various fauna sighted during every dive undertaken by GVI has been recorded. This data is purely qualitative and should be taken cautiously as recordings are not systematic, nor standardized, thus vary according to many factors e.g. visibility, observer, site, dive purpose. 2018 data is presented for sharks, rays and turtles sightings divided into two six months' periods.

	Jan - Jun	Jul - Dic
Ray	**254**	**149**
Devil Ray	5	7
Feathertail Ray	12	4
Manta Ray	3	-
Marble ray	48	30
Spotted Eagle Ray	173	105
Thornback Ray	13	3
Shark	**197**	**79**
Black Tip reef Shark	3	2
Grey Reef Shark	-	1
Guitar Shark	4	1
Whale shark	2	1
White Tip Shark	188	74
Turtle	**148**	**74**
Green Turtle	36	20
Hawksbill Turtle	112	54
Grand Total	**599**	**302**

Table 3. Number of sharks, rays and turtles recorded for each dive logged in 2018. Note that number of dives has been omitted, as data is only exploratory, and number of dives has been fairly constant over the year.

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Appendix A

A.1. Coral genera (LIT, coral diversity belts and recruitment quadrat surveys)

Family	Genus	Family	Genus
Acroporidae	*Acropora* *Astreopora* *Montipora*	Fungidae	*Cycloseris* *Diaseris* *Fungia* *Herpolitha* *Podabacia* *Polyphyllia* *Halomitra*
Agariciidae	*Coeloseris* *Gardineroseris* *Leptoseris* *Pachyseris* *Pavona*	Merulinidae	*Hydnophora* *Merulina*
Astrocoeniidae	*Stylocoeniella*	Mussidae	*Acanthastrea* *Blastomussa* *Lobophyllia* *Symphyllia*
Dendrophyllidae	*Turbinaria*	Oculinidae	*Galaxea*
Euphyllidae	*Physogyra*	Pectiniidae	*Echinophyllia* *Mycedium* *Pectinia*
Faviidae	*Cyphastrea* *Diploastrea* *Echinopora* *Favia* *Favites* *Goniastrea* *Leptastrea* *Leptoria* *Montastrea* *Oulophyllia* *Platygyra* *Plesiastrea*	Pocilloporidae	*Pocillopora* *Seriatopora* *Stylophora*
		Poritidae	*Alveopora* *Goniopora* *Porites*
		Siderastreidae	*Psammocora* *Pseudosiderastrea* *Siderastrea* *Cocinarea*

Note: The genus Montastrea (Faviidae) has included 3 species since 2009 (start of recording coral genera). One of those species belongs to the genus Favia (Faviidae), but due to strong similarities has been included in the genus Montastrea to simplify the identification process. The genus Blastomussa (Mussidae) also includes one species from the genus Acanthastrea (Mussidae). Considering this has been systematic since 2009, identification procedure has not been changed so as not to skew the data.

A.2. Benthic organisms and substrate types (LIT surveys)

Algae	Macro algae, turf algae, *Halimeda* sp., coralline algae, algae assemblage
Other lifeforms recorded	*Heliopora* sp., *Millepora* sp., *Tubipora* sp., soft coral, sponge, corallimorphs, zoanthids
Organisms recorded as 'other'	Bryozoans, ascidians, bivalves, anemones, black corals, gorgonians, sea Stars, holothurians
Substrate	Sand, rubble, rock, dead coral, silt, water

Appendix B

B.1. Fish species list

Family	Scientific name	Common name	Feeding guild	Relevance*
Butterflyfish (Chaetodontidae)	*Chaetodon vagabundus*	Vagabond	C/I	Coral recovery
	Chaetodon auriga	Threadfin	C/I	Coral recovery
	Chaetodon trifascialis	Chevroned	C	Coral recovery
	Chaetodon melannotus	Black-backed	C/I	Coral recovery
	Chaetodon madagaskariensis **	Seychelles	C/I	Coral recovery
	Chaetodon triangulum	Triangular	C	Coral recovery
	Chaetodon trifasciatus	Indian redfin	C	Coral recovery
	Chaetodon interruptus	Indian Ocean teardrop	C/I	Coral recovery
	Chaetodon bennetti	Bennett's	C	Coral recovery
	Chaetodon lunula	Raccoon	C/I	Coral recovery
	Chaetodon kleinii	Klein's	C/I	Coral recovery
	Chaetodon citrinellus	Speckled	C/I	Coral recovery
	Chaetodon guttatisimus	Spotted	C/I	Coral recovery
	Chaetodon lineolatus	Lined	C/I	Coral recovery
	Chaetodon falcula	Saddleback	C/I	Coral recovery
	Chaetodon meyersi	Meyer's	C	Coral recovery
	Chaetodon xanthocephalus	Yellow-headed	C/I	Coral recovery
	Chaetodon zanzibariensis	Zanzibar	C	Coral recovery
	Forcipiger sp.	Longnose	C/I	Coral recovery
Angelfish (Pomacanthidae)	*Apolemichthys trimaculatus*	Three-spot	V	Visual appeal
	Pomacanthus imperator	Emperor	V	Visual appeal
	Pomacanthus semicirculatus	Semicircle	V	Visual appeal
	Pygoplites diacanthus	Regal	V	Visual appeal
Surgeonfish (Acanthuridae)	*Acanthurus* sp.	Surgeonfish	H	Algae vs. coral
	Ctenochaetus sp.	Bristletooths	H	Algae vs. coral
	Naso sp.	Unicornfish	Pl	Algae vs. coral
Moorish Idol (Zanclidae)	*Zanclus cornutus*	Moorish idol	V	Visual appeal
Rabbitfish (Siganidae)	*Siganus puelloides*	Blackeye	H	Algae vs. coral
	Siganus corallinus	Coral	H	Algae vs. coral
	Siganus stellatus	Honeycomb	H	Algae vs. coral

	Siganus argenteus	Forktail	H	Algae vs. coral
	Siganus sutor	African whitespotted	H	Algae vs. coral
Snappers (Lutjanidae)	*Lutjanus gibbus*	Paddletail	Pi	Fishing pressure
	Lutjanus sebae	Red emperor	Pi	Fishing pressure
	Lutjanus fulviflamma	Longspot	Pi	Fishing pressure
	Lutjanus kasmira	Blue-lined	Pi	Fishing pressure
	Lutjanus bengalensis	Bengal	Pi	Fishing pressure
	Lutjanus monostigma	Onespot	Pi	Fishing pressure
	Lutjanus vitta	Brownstripe	Pi	Fishing pressure
	Lutjanus fulvus	Flametail	Pi	Fishing pressure
	Lutjanus argentimaculatus	Mangrove jack	Pi	Fishing pressure
	Lutjanus bohar	Red	Pi	Fishing pressure
	Lutjanus russelli	Russell's	Pi	Fishing pressure
	Macolor niger	Black	Pi	Fishing pressure
	Aprion virescens	Green jobfish	Pi	Fishing pressure
Triggerfish (Balistidae)	*Balistoides viridescens*	Titan	I	Sea urchins & COTs
	Sufflamen chrysopterus	Flagtail	I	Sea urchins & COTs
	Balistida	Other triggerfish	I	Sea urchins & COTs
Emperors (Lethrinidae)	*Monotaxis* sp.	Bigeye bream	I	Sea urchins & COTs
	Gymnocranius grandoculis	Blue-lined large-eye bream	I	Sea urchins & COTs
	Lethrinus olivaceous	Longnosed	I	Sea urchins & COTs
	Lethrinus nebulosus	Blue-scaled	I	Sea urchins & COTs
	Lethrinus rubrioperculatus	Redear	I	Sea urchins & COTs
	Lethrinus xanthochilus	Yellowlip	I	Sea urchins & COTs
	Lethrinus harak	Thumbprint	I	Sea urchins & COTs
	Lethrinus lentjan	Pinkear	I	Sea urchins & COTs
	Lethrinus obsoletus	Orange-striped	I	Sea urchins & COTs
	Lethrinus erythracanthus	Yellowfin	I	Sea urchins & COTs
	Lethrinus mahsena	Mahsena	I	Sea urchins & COTs
	Lethrinus variegatus	Variegated	I	Sea urchins & COTs
Groupers (Serranidae)	*Anyperodon leucogrammicus*	Slender	Pi	Fishing pressure
	Cephalopholi sargus	Peacock	Pi	Fishing pressure
	Cephalopholis urodeta	Flagtail	Pi	Fishing pressure
	Cephalopholis miniata	Coral hind	Pi	Fishing pressure
	Cephalopholis sonnerati	Tomato	Pi	Fishing pressure

	Epinephelus merra	Honeycomb	Pi	Fishing pressure
	Epinephelus spilotoceps	Foursaddle	Pi	Fishing pressure
	Epinephelus polyphekadion	Camouflage	Pi	Fishing pressure
	Epinephelus caeruleopunctatus	Whitespotted	Pi	Fishing pressure
	Epinephelus fuscoguttatus	Brown-marbled	Pi	Fishing pressure
	Epinephelus tukula	Potato	Pi	Fishing pressure
	Epinephelus fasciatus	Blacktip	Pi	Fishing pressure
	Aethaloperca rogaa	Redmouth	Pi	Fishing pressure
	Variola louti	Yellow-edged lyretail	Pi	Fishing pressure
	Plectropomus laevis	Saddleback	Pi	Fishing pressure
	Plectropomus punctatus	African coral cod	Pi	Fishing pressure
Sweetlips (Haemulidae)	*Plectorhinchus orientalis*	Oriental	I	Sea urchins & COTs
	Plectorhinchus picus	Spotted	I	Sea urchins & COTs
	Plectorhinchus gibbosus	Gibbus	I	Sea urchins & COTs
Parrotfish (Scaridae)	*Bolbometopon muricatum*	Bumphead parrotfish	C/H	Coral damage
	Scaridae	Other parrotfish	H	Algae vs. coral
Wrasse (Labridae)	*Cheilinus trilobatus*	Tripletail	I	Sea urchins & COTs
	Cheilinus fasciatus	Redbreasted	I	Sea urchins & COTs
	Oxycheilinus digrammus	Cheeklined splendour	I	Sea urchins & COTs
	Cheilinus undulatus	Humphead	I	Sea urchins & COTs
Puffers (Tetradontidae)	Tetraodontidae	Puffers	I	Sea urchins & COTs
Porcupinefish (Diodontidae)	Diodontidae	Porcupinefish	I	Sea urchins & COTs
Soldierfish & squirrelfish (Holocentridae)	Holocentridae	Soldierfish	Pl	Upwelling areas
		Squirrelfish	Pl	Upwelling areas

* based on Engelhardt (2004)

** *Chaetodon madagaskariensis* (Seychelles) was formerly *Chaetodon mertensii* (Merten’s)

B.2. Fish species lists divided into commercial and reef fish

Commercial fish families*	**Reef fish families and species***
Siganidae (rabbitfish)	Chaetodontidae (butterflyfish)
Lutjanidae (snappers)	Pomacanthidae (angelfish)
Lethrinidae (emperors)	Acanthuridae (surgeonfish)
Serranidae (groupers)	Balistidae (triggerfish)
Haemulidae (sweetlips)	Labridae (wrasse)
Scaridae (parrotfish)	Tetradontidae (pufferfish)
	Diodontidae (porcupinefish)
	Holocentridae (soldierfish & squirrelfish)
	Zanclus cornutus (moorish idol)
	Bulbometopon muricatum (bumphead parrotfish)

*according to GVI Seychelles (Mahé) methodology

B.3. Fish feeding guilds as referred to in B.1.

Code	Feeding guild	Description (adapted from Obura and Grimsditch, 2009)	Key species
Pl	Planktivorous	Resident on reef surfaces, but feed in the water column. Their abundance is related to quality of reef habitat for refuge, and water column conditions.	Soldierfish, squirrelfish, unicornfish
Pi	Piscivorous	High level predators. Exert top-down control on lower trophic levels. Important fisheries species but very vulnerable to overfishing thus good indicators of the fishing pressure on a reef.	Groupers, snappers
C	Corallivorous	Relative abundance is an indicator of coral community health	Butterflyfish (chevroned, triangular, Bennett's, Indian redfin, Meyer's, longnose)
V	Varied diet	Feed on coral competitors such as soft corals and sponges. Relative abundances may be an indicator of abundance of these prey items and of a phase shift.	Angelfish, moorish idol
I	Invertivorous*	Second-level predators with highly mixed diets including small fish, invertebrates and dead animals. Important fisheries species thus abundances are a good indicator of fishing pressure.	Sweetlips, emperors, pufferfish, porcupinefish, wrasse (tripletail, redbreasted, cheeklined splendour, humphead), triggerfish (titan, flagtail, other triggerfish)
H	Herbivorous	Exert the primary control on coral-algal dynamics. May indicate phase shift from coral to algal dominance in response to mass coral mortality or pressures such as eutrophication.	Parrotfish, surgeonfish, bristletooth, rabbitfish
C/H	Corallivorous/Herbivorous	Relative abundance is a secondary indicator of coral community health	Bumphead parrotfish
C/I	Corallivorous/Invertivorous	Relative abundance can be a secondary indicator of coral community health	Butterflyfish (vagabond, threadfin, blackbacked, Merten's, Indian Ocean teardrop, racoon, Klein's, speckled, spotted, lined, saddleback, yellow headed, Zanzibar)

Appendix C

C.1. List of invertebrates surveyed on 50 m belts

Mollusca (Gastropoda)	*Drupella* spp	*Drupella* spp
Mollusca (Bivalvia)	Tridacnidae	Giant clam
Sea stars (Asteroidea)	*Culcita* spp.	Cushion sea star
	Acanthaster planci	Crown-of-thorns sea star
		Other sea stars
Sea urchins (Echinoidea)	*Diadema spp.*	Long-spine urchin
	Echinometra spp.	Mathae's urchin
	Echinothrix spp.	Short-spine urchin
		Pencil urchin
	Toxopneustes pileolus	Flower urchin
		Cake urchin
Sea Cucumbers (Holothuroidea)	*Holothuria artra*	Lollyfish
	Holothuria fuscopunctata	Elephant trunk
	Holothuria fuscogilva	White teatfish
	Holothuria nobilis	Black teatfish
	Holothuria sp.(undescribed)	Pentard
	Bohadschia spp.	*Bohadschia* spp.
	Actinopyga spp.	*Actinopyga* spp.
	Actinopyga mauritiana	Yellow surfish
	Stichopus spp.	Stichopus
	Thelenota ananas	Prickly redfish
	Pearsonothurian graeffei	Flowerfish
	Thelenota anax	Royal
	Holothuria edulis	Edible sea cucumber
(Cephalopoda)	*Octopus* spp.	Common reef octopus
Lobsters (Palinura)	*Panulirus* spp.	Spiny lobster
	Parribacus spp./*Scyllarides* spp.	Slipper lobster

C.2. List of invertebrates surveyed on 10 m belts

Annelida (Polychaeta)	**Sabellidae**	Feather duster worms
	Serpulidae	Christmas tree worms
	Terebellidae	Spaghetti worms
(Platyhelminthes)	**Polycladida**	Flatworms
Arthropoda (Crustacea)	**Caridea**	Shrimps
	Stomatopoda	Mantis shrimps
	-	Crabs
Mollusca (Gastropoda)	**Muricidae**	Murex
	Drupella sp.	*Drupella* spp.
	Strombidae	Conch
	Cypraeidae	Cowrie
	Ranellidae	Triton
	Conidae	Cone
	Trochidae	Top
	Cassidae	Helmet
	-	Other shells
	Nudibranchia	Nudibranchs
Mollusca (Bivalvia)	**Ostreidae**	Oysters
	Tridacnidae	Giant clam
Mollusca (Cephalopoda)	**Sepiidae**	Cuttlefish
	Loliginidae	Squid
Sea Stars (Asteroidea)	***Culcita* sp.**	Cushion sea star
	Acanthaster planci	Crown-of-thorns sea star
		Other sea stars
Brittle stars (Asterozoa)	**Ophiuroidea**	Brittle stars
Feather stars (Crinozoa)	**Crinoidea**	Feather stars
Sea urchins (Echinoidea)	***Diadema* sp.**	Long-spine urchin
	***Echinometra* sp.**	Mathae’s urchin
	***Echinothrix* sp.**	Short-spine urchin
		Pencil urchin
	***Toxopneustes* sp.**	Flower urchin
	***Tripneustes* sp.**	Cake urchin Other urchins