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Trends and other temporal changes recorded in coral reef and seagrass areas during ten years of the Anguilla Marine Monitoring Programme

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ABSTRACT

Concerns relating to continued habitat degradation in the coastal areas around Anguilla, combined with the recognition that ecological data is essential before effective management decisions can be made, led to the initiation of the Anguilla Marine Monitoring Programme in 2007. Now in its tenth year, surveys are undertaken at fifteen sites around the island that collect a suite of data on fish, plants, algae, corals, and other invertebrates, and represents the first time series dataset for the island. This report examines five primary variables identified as crucial when establishing habitat health in both coral reef and seagrass areas: coral cover; algae cover; fish species diversity; overall fish abundance; and size of commercially and ecologically important fish families. A small number of other variables are also examined as related to habitat health. Over the study period, an 11% drop in coral cover was recorded, with a mean value across all sites combined falling from 5.6% to 5.1%, and algae cover increased by 9.5%, from 14.2% to 15.0%. The most common coral species was *Porites astreoides*, although it has decreased in frequency by 45% since this work began. The massive reef building species, *Acropora palmata* today only accounts for 2% of coral colonies recorded. Fish species number decreased across all sites by 21% with a mean count of 48 at each site at the beginning of the survey, dropping to 38 by 2016. Fish abundance also decreased with a reduction of 27% from a mean of 621 individuals counted per survey to 451. Similarly, commercially and ecologically important fish family sizes decreased by 24% over the study period, with a mean of 10.6 cm lowering to 8.1 cm. This trend in fish size could be seen amongst all families analysed, with Acanthuridae, Scaridae and Serranidae decreasing by 31% (9.5 cm to 6.5 cm), 9% (11.5 cm to 10.4 cm), and 54% (17.5 cm to 8.1 cm) respectively. In terms of seagrass sites, overall plant/algae cover increased by 15% from 66.4% to 78.2%, although this was seen to be due primarily to increases in the invasive seagrass *Halophila stipulacea* and weedy algal species *Dasycladus vermicularis.* Percentage cover of the native seagrass *Thalassia testudinum* declined by 21% from 56.2% to 44.6%. These results illustrate that Anguilla's coastal coral reef habitat are in an increasingly poor state of health, having been in continual decline since the 1970's. Seagrass beds in Anguilla are also in decline, although currently still in a relatively healthy state. As many of the stressors influencing habitat health are regional in nature, management measures are unlikely to be able to reverse these negative trends, although local mitigation measures may help to slow them. It is recommended that existing marine parks be used to preserve relic populations, with restrictions applied to extractive and/or damaging practices. It is also recommended to introduce minimum landing sizes for key reef fish species, regulation of all septic tanks in coastal areas, introduction/enforcement of development setbacks, and minimization of any sources of terrestrial runoff. If marine habitat health continues to decline, beach erosion will increase, coastal fisheries will collapse, and almost all livelihoods around the island will be negatively impacted.

Introduction

Coral reefs throughout the Caribbean have been in decline since the 1970's, with an average 80% reduction in coral cover on reefs reported over the three decades prior to 2000 (Jackson, 2001; Gardner *et al*, 2003). The loss of the massive reef building corals is of particular concern, as the geologic record shows they have thrived for over 1.5 million years in the Caribbean, creating the breakwaters and fish habitats that we see the remnants of today (Kuffner & Toth, 2016). The demise of these crucial species of coral, followed by their gradual bioerosion, is now influencing erosional processes and having a negative impact on reef fish densities along with other profound ecological changes (Paddack *et al*., 2009). Although other theories have been proposed as to the primary cause of this initial coral cover decline, for example the *Diadema antillarum* mass mortality event of the early 1980's (Jackson, 2001), it is now widely considered that the onset was due to disease outbreak (Aronson & Precht, 2001), with other stressors adding subsequent pressure. Indeed, although many now attribute the continued coral cover loss to temperature induced bleaching and ocean acidification (Anthony *et al*., 2011) little convincing evidence exists that this was the initial cause, or even a significant contributor to the current status of reefs in the Caribbean (Gardner *et al*., 2003), outside a small number of 'bleaching events'. Instead, it seems that local factors originating both naturally (e.g. storms) and anthropogenically (e.g. overfishing) have been interacting with regional stressors (e.g. eutrophication) to cause this degradation at Caribbean-wide scales (Wynne, 2016a).

This appears to be the case in Anguilla where white band disease began causing significant *Acropora sp.* mortality in the 1970s (Bythell & Buchan, 1996), ultimately resulting in its almost entire loss. The source and insurgence of this (and other) coral disease is still under discussion, although it has been suggested that it originated from African dust, as the onset coincided with large increases in transatlantic dust transport. The soil fungus *Aspergillus sydowii*, the cause of an ongoing Caribbean-wide sea fan disease, has also been cultured from samples (Shinn *et al*., 2000). Another theory is that pathogens were introduced into the region with the opening of the Panama Canal, or via ship's ballast (Bythell, pers. comm.). It is also believed that at this time background nutrient levels began increasing in the Caribbean, via the Amazon and Orinoco river plumes, enriched due to deforestation and increasing intensive farming in the Amazon Basin (Wynne, 2016a). These nutrients fed other organisms that were damaging to corals, such as disease pathogens (Bruno *et al*., 2003), and help promote macroalgae proliferation that can inhibit coral regrowth (Webster *et al*., 2015). During subsequent hurricanes the remaining reef-building corals failed to recover from storm damage - events which previously would have broken up and dispersed corals allowing reef areas to ultimately expand (Ball, 1967; Bonem, 1988). With Anguilla's coastlines as well as fishing and tourism industries almost entirely undeveloped during this early stage of degradation, it is unlikely any locally-induced stressors were involved, as has been suggested in other parts of the Caribbean (Gardner *el al*., 2003). It is however probable that more developed areas, through the input of local sources of organic materials (e.g. sewage, agricultural runoff), would have added to the overall regional build-up of nutrients and so contributed to the situation. This would explain the similarity of the situation currently seen in Anguilla with other parts of the Caribbean, that of: low hard coral cover, especially the massive reef builders (e.g. *Acropora sp*. & *Montastraea sp.*); high macroalgae cover; increasingly turbid water; and increased incidences of cyanobacteria coral infections.

As with a number of other places in the Eastern Caribbean, one problem with drawing these temporal conclusions in Anguilla is the lack of historic ecological information. It is not possible to discuss the changes that happened here during the 1970's and 1980's without quantifiable data. Similar situations around the region have left others to wonder if indeed there ever were long-lasting expansive stands of *Acropora palmata* or how much macroalgae is actually natural for the region (Bruno *et al*., 2014). This phenomenon, known as shifting baselines, is a problem many managers face, especially when the only information available is anecdotal local reports of years gone by. With the demise of the main reef builders almost complete by the early 1990's it has been suggested that coral cover decline has plateaued and levels now remain relatively constant (Jackson, 2012), albeit populated by hardier, more weedy species that will not result in the formation of formidable coastal defences (Green *et al*., 2008). In order to clarify statements such as this, to establish local vs regional variation, and to build a current baseline that is quantifiable and thus permanent, it is essential to collect ecological information, preferably regularly and over the long-term.

The first ecological data on record collected in Anguilla was a study by the Bellairs Institute (Oxenford & Hunte, 1990) undertaken at the time to assess potential marine park sites while attempting to set up a long-term monitoring scheme. Unfortunately this long-term monitoring scheme was discontinued beyond the initial surveys, and despite Reef Check attempting to establish permanent monitoring sites in the early 2000's, none were maintained. It was for this reason that in 2007, combined with continued concerns over habitat health, that the Department of Fisheries and Marine Resources (DFMR) established the Anguilla Marine Monitoring Programme (AMMP). Since then annual surveys have been conducted, with long-term monitoring now in its tenth year. The initial year(s) of monitoring served to establish a methodology, fully train staff, and establish this all important baseline – with reports produced to document this (Wynne, 2007; Wynne 2008a; Wynne 2008b). By 2009 fifteen monitoring locations had been established around the island (see Figure 1), with ten on coral reefs areas and five on seagrass beds. Some of those data collected in 2009 were used for the first temporal comparisons against those collected by Oxenford & Hunte (1990). This comparison (found in Wynne, 2010) concluded that between the two study periods coral cover had declined by up to 99% in some areas (e.g. Forest Bay). Most sites suffered a loss between 50 and 90%, bringing mean coral cover in the waters around Anguilla down to below 10%.

A few years later in 2012, a further temporal analysis was conducted, and the first using solely AMMP data (Gumbs, 2012). This report concluded that few trends were currently identifiable aside from the reduction of seagrass at one site (Crocus Bay) and changes in coral cover at two sites (increase at Limestone Bay and decrease at Sandy Island). It is probable that this was due to the short data span (five years). With monitoring now in its tenth year it is hoped that this work will now be able to reveal temporal trends taking place in Anguillian waters. In order to focus this search, this current paper will concentrate on the five key reef indicator variables identified by Wynne (2016a) as crucial when establishing a 'relative health index' (RHI) value for sites. Newly obtained historic underwater photographic evidence will also be examined, and assessed against recent ground-truth work to look into the demise of the massive reef building coral species, disease occurrence and overall water clarity. Inferences will also be made in relation to other key ecological species, especially potential population recovery of *Diadema antillarum* and occurrences of the invasive *Pterois volitans* (lionfish) across study sites. From these results, 'business as usual' future scenarios will be presented along with management recommendations that may serve to, at least in part, mitigate against any negative trends identified.

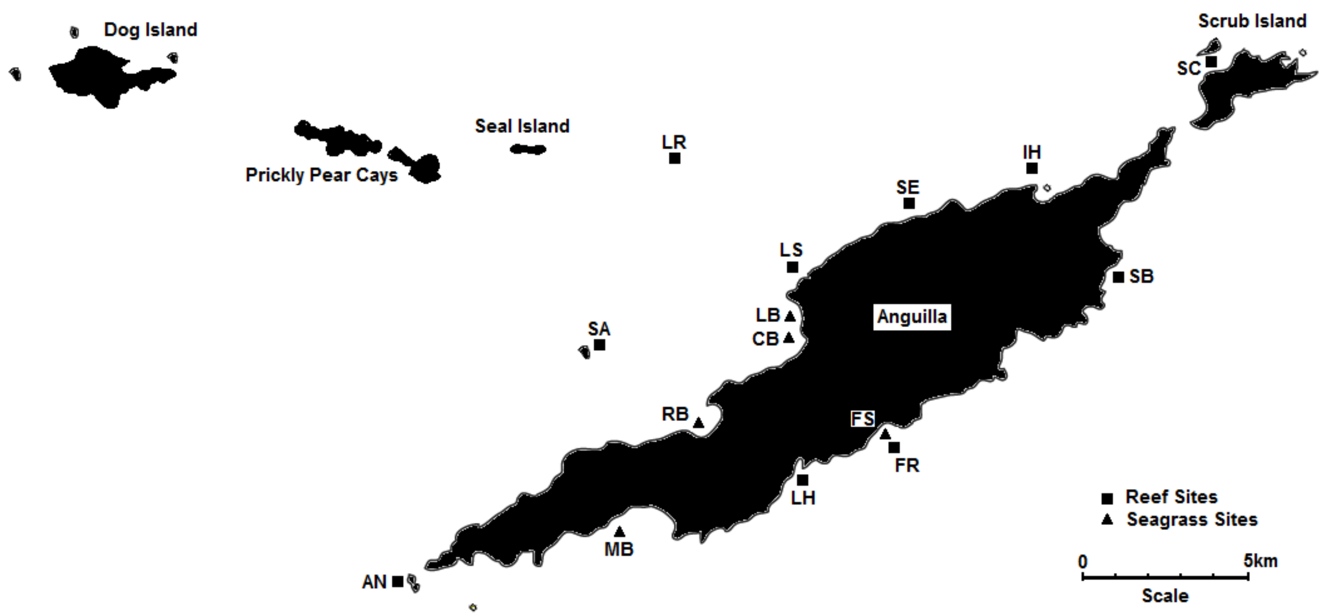

Figure 1 – Map showing locations of both coral reef and seagrass monitoring sites. Reef site codes (square): AN Anguillita; SA Sandy Island; LR Long Reef; LS Limestone Bay; SE Shoal Bay East; IH Island Harbour; SC Scrub Island; FB Forest Bay; LH Little Habour; and SB Sile Bay. Seagrass site codes (triangle): MW Merrywing Bay; RB Road Bay; LB Little Bay; FS Forest Bay; CB Crocus Bay.

Methods

The methodology used for AMMP closely follows that laid out by the Atlantic and Gulf Rapid Reef Assessment (AGGRA – see Kramer *et al*., 2005) with some slight modifications taken from English *et al*., 1997) to expand upon variables collected. A detailed description of the AMMP protocol can be found in Wynne (2007). The protocol was designed to be undertaken by 2 to 4 divers using SCUBA equipment.

Fifteen permanent monitoring sites, each marked with a fixed subsurface marker, were established between 2007-2009 - ten on variable reef areas and five on seagrass beds. Monitoring begins once the ground sea (winter swell) season has ended (usually before the end of May) and aims to be completed before the beginning of August to avoid potential bleaching and other biases that may exist as sea surface temperature increase and hurricane events become more frequent. Surveys are conducted between 7am-11am to avoid peak daylight biases.

First, two divers enter the water and lay 2x50 m transect tapes in a cross, one parallel to the coastline/reef slope and one perpendicular to it, with the PMS subsurface marker at its center. Once laid, the divers allow the tape to settle and fish to reacclimatize before beginning their surveys. Each diver swims along the transect at a slow, steady pace (5 m/min recommended), counting and estimating the size of fish species within 2.5 m either side of the transect tape. Thus a total area of 250 m^2 is surveyed per transect and, with two replicates of each conducted, this gives a total survey area of 1000 m^2. Fish counted during this survey are limited to those of commercial or ecological importance, and size estimations place each counted into 5 cm size classes. Species are split by family equally between divers and primarily focus on Acanthuridae (Surgeonfish), Haemulidae (Grunts), Lutjanidae (Snapper), Serranidae (Grouper), Holocentridae (Squirrelfish), Ballistidae (Triggerfish), Mullidae (Goatfish), Pomacanthidae (Angelfish), Chaetodontidae (Butterflyfish), Carangidae (Jacks) and Scaridae (Parrotfish).

Once completed, the two divers begin habitat surveys with 50x50 cm quadrats placed every 5 m along the transect tapes (excluding the central 25 m mark where the central marker is located), giving a total of ten quadrats per transect and an overall survey area of five square meters. Within each quadrat the physical parameters measured are: rugosity, using a fine link chain (Risk, 1972); depth, using a dive computer; and relief, which is a measure of the vertical height difference in cm between the lowest and highest sides of the quadrat when laid on the substrate. Underlying substrate percentages are recorded (sand, rubble or hard rock/coral), along with a full benthic biota assessment that records percentage cover of: turf/sediment covered rock; coralline algae; calcareous algae; fleshy algae; other plants/algae; cyanobacteria; fire coral; hard coral; soft coral; sponge; and other invertebrates. Where possible everything is identified to species or a minimum of genus, and hard coral colonies are counted and tallied by species recorded. The mean height of each soft coral is also measured within the quadrat.

Meanwhile, the remaining two divers enter the water and begin their surveys once the transect tapes have been laid. The first conducts two 30 min surveys using the roving diver technique (RDT), which is conducted in a circular shape with approximately a 25 m radius around the central marker. Swimming at a constant speed of 5 m/min, one circuit around the study site can be completed in the allocated time (Circumference 155 m, distance covered by surveyor 150 m). All fish seen within 2.5 m of the diver are counted and identified to species, giving total fish abundance and diversity values for the site, although care has to be taken to not count the same fish twice. This gives a total survey area of 750 m^2.

During this time, the remaining diver conducts line intercept surveys along each of the transect tapes. Starting at the zero mark, the diver follows along the tape and records accurately (nearest cm) changes in substrate (sand, rubble, hard). When a live coral colony is reached its maximum length, width and height are recorded, along with details on colony health (percentage alive, recent dead or long dead) and tissue health (percentage live tissue healthy vs that showing signs of disease or other sickness). This process continues along both the 50 m transect tapes. On a separate survey sheet, the diver also records any macro-invertebrates seen 1 m either side of the tape, with special care to look under ledges and overhangs. Of particular interest are crustaceans and echinoderms due to their potential commercial or ecological importance.

All divers are also required to record temperature (if dive computer available), visibility (using transect tape and reference point/secchi disc), weather, sea conditions, and time/date of surveys, date. Divers are also encouraged to bring along cameras and photograph anything of interest, especially the central marker and surroundings.

Seagrass surveys are conducted in a similar manner to that described above, although during the fish belt transects all species are recorded by one diver (due to lower abundance and diversity in these areas) and as such no RDT is necessary. During the quadrat surveys certain variables are not collected (e.g. rugosity) due to habitat differences, and other variables are added (sea grass blade length, and number of blades per plant). Line intercept surveys are also not conducted.

Results

Results presented in this section will focus on the five key indicator variables used to establish the relative health index (RHI) for sites as used in Anguilla previously, as detailed in Wynne (2016a). These indicators are: coral cover; macroalgae cover; fish species number; fish abundance; and fish size. Where appropriate, results expand upon these variables where more detail is warranted or where relevant temporal trends have been identified. Results from seagrass sites are also included, along with observations on coral diseases, water clarity, *D. antillarum*, and *P. volitans*.

Coral Cover

Massive reef building coral species belonging to the genus *Acropora* are today uncommon in Anguilla, accounting for less than 2% of colonies recorded over the study period. The genus *Orbicella*[1] also historically formed extensive reefs around Anguilla but now accounts for approximately 9% of colonies. *Porites porites,* which formed extensive large colonies, now accounts for over 11% of those encountered, and the third most common coral species in Anguilla. However, the majority of these are

[1] Formally classified as *Montastraea*

small fledgling colonies, aside from a small number of large examples still visible at the Sandy Island study site and close to the Long Reef study site. Brain corals from the genus *Diploria*, also with the potential to be a reef building species, accounted for approximately 5% of colonies recorded, although their historical abundance is not clear. A histogram illustrating total coral colony counts during the study period is presented in Figure 2.

As Figure 2 illustrates, the dominant coral species in Anguilla over the last decade are hardier, weedier species unable to build the reef structures we see the remnants of today, with *Porites astreoides* being the most common. Although this species is considered unsusceptible to most coral syndromes (Green *et al*., 2008), it has become increasingly common to observe diseased colonies over recent years. This is illustrated as an overall decline in *Porites astreoides* occurrence between 2008-2016, as presented in Figure 3.

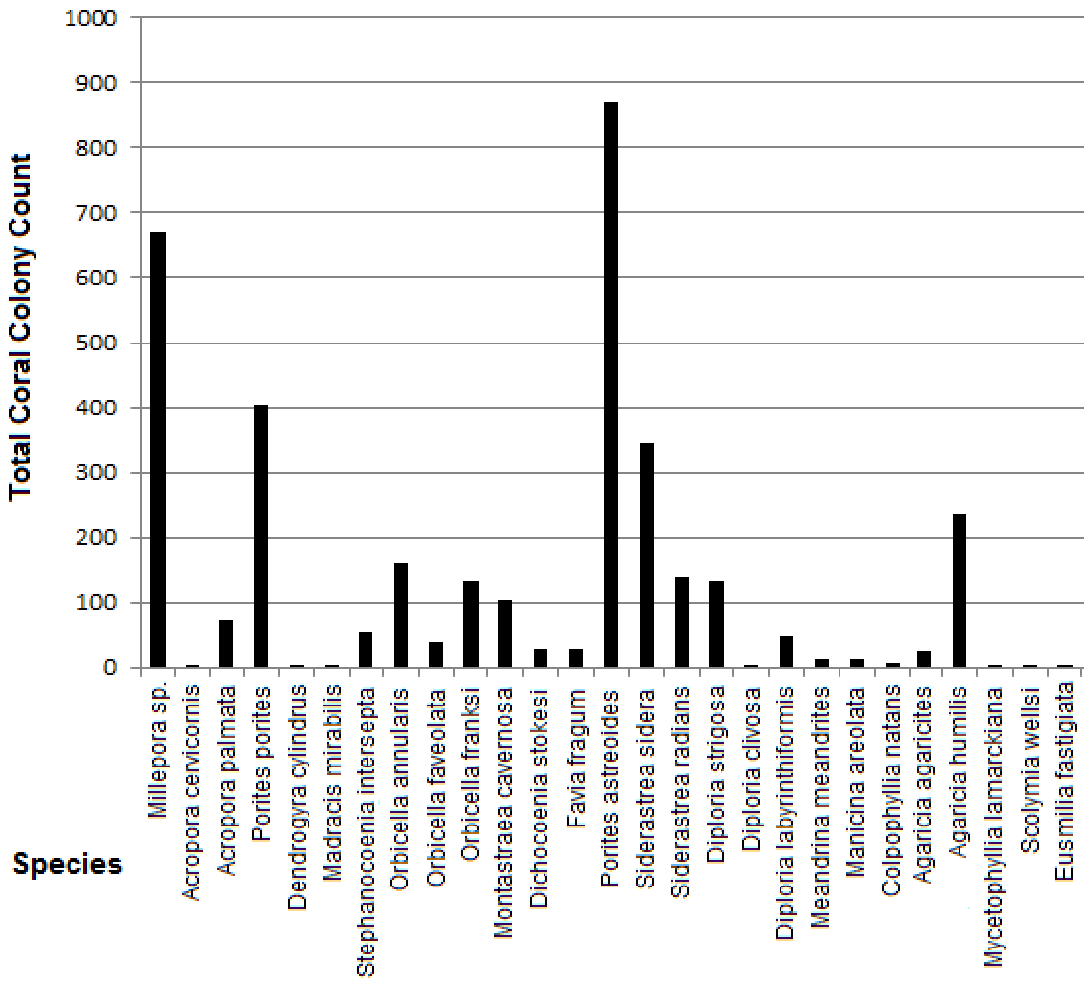

Figure 2 – Histogram of coral species occurrence during the 2008-2016 study period, as collected during both habitat quadrat and line intercept methodologies. Results clearly illustrate the most common species found are *Porites astreoides* and *Millepora sp*. Massive reef building species, for example *Acorpora sp*. and *Orbicella sp*. are far less common than the historical record suggest.

Similar declines were seen in the counts for virtually all other species including both the massive reef builder generas (*Orbicella*, *Porites* and *Diploria*) and the hardier, weedier generas (*Millepora* and *Siderastrea*), although there was much variability between sites and most temporal trends were relatively weak. Positive trends were recorded for *Acropora palmata* and *Agaricia sp.* although again much variability existed between sites. In terms of *Acropora palmata*, although a positive trend is encouraging, colony numbers are still so dramatically low that overall percentage cover has not changed during the study period. When looking at total hard coral percentage cover, these weak negative trends still remain discernible, illustrating continued small colony size and an ongoing paucity of the massive reef builders (see Figure 4). The trend presented in this figure shows an 11% loss of coral cover over the study period, with a reduction from 5.6% to 5.1%, which is dramatic when considering the low starting percentage cover.

Further to the trend illustrated in Figure 4, the mean data presented in Table 1 help to understand this relationship, and provides more detail as to what is specifically taking place in the coastal areas around Anguilla. For example, it can be seen that both Anguillita and Sandy Island have suffered an almost continual decline in overall coral cover, with Long Reef, Limestone Bay and Shoal Bay East fluctuating across the years but with a small positive tendency. The sites that began the study with low percentage covers (Island Harbour, Scrub Island, Forest Bay, Little Harbour and Sile Bay) have all remained with a continued low level of cover.

Algae Cover

Over the study period, in a similar way to coral cover, percentage cover of macroalgae has varied greatly between sites. However, although remaining relatively stable overall, a positive trend representing an increase in cover was identified (Figure 5). A site by site annual breakdown of mean levels illustrating this variability can be found in Table 2.

Fish Species Diversity, Abundance and Size

Graphs representing core fish data across the survey years are presented in Figures 6 (species diversity), Figure 7 (species abundance) and Figure 8 (size class). Species diversity is the mean sum total of all species recorded at each study site per year using the RDT methodology, with abundance calculated using the same methodology and being the mean sum total of all fish counted at each study site. Fish sizes are taken from belt transect surveys and include only species of commercial or ecological importance. Trends were investigated for other variables from this survey type (relative biomass and number per family for example), but size gave the overall highest R^2 values. Figures 9-11 go on to break this size class information down into three key families of special importance: Acanthuridae (Figure 9); Scaridae (Figure 10); and Serranidae (Figure 11). All figures illustrate negative trends of varying strengths, although interestingly that seen in Figure 10 (Scaridae) is the weakest, possibly due to a paucity of larger size classes when monitoring began in 2008.

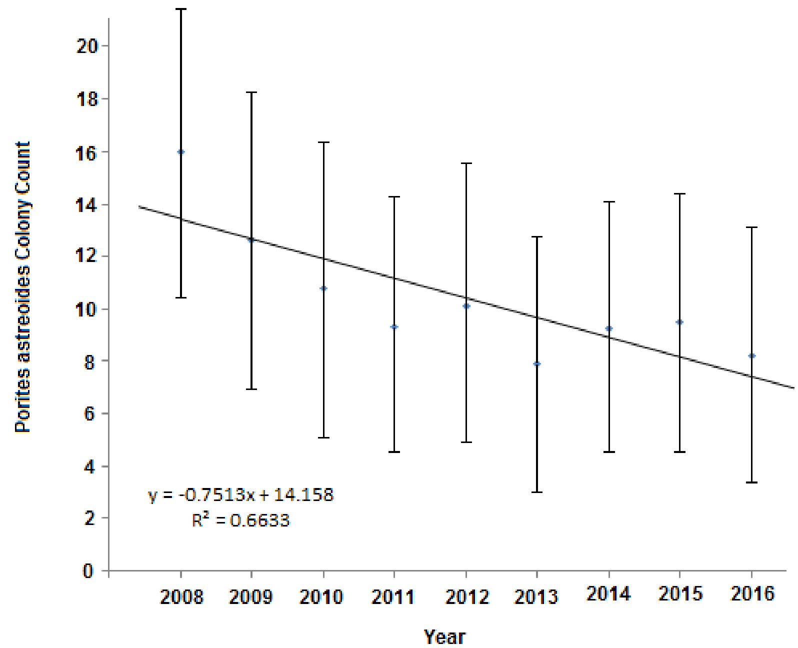

Figure 3 – Graph illustrating mean count of *Porites astreoides* colonies across all study sites during both habitat quadrat and line intercept methodologies, showing a 45% drop in frequency from 13.4 colonies per survey site to 7.4. Regression equation is presented where x = year series number (2008 = 1, 2009 = 2, etc.).

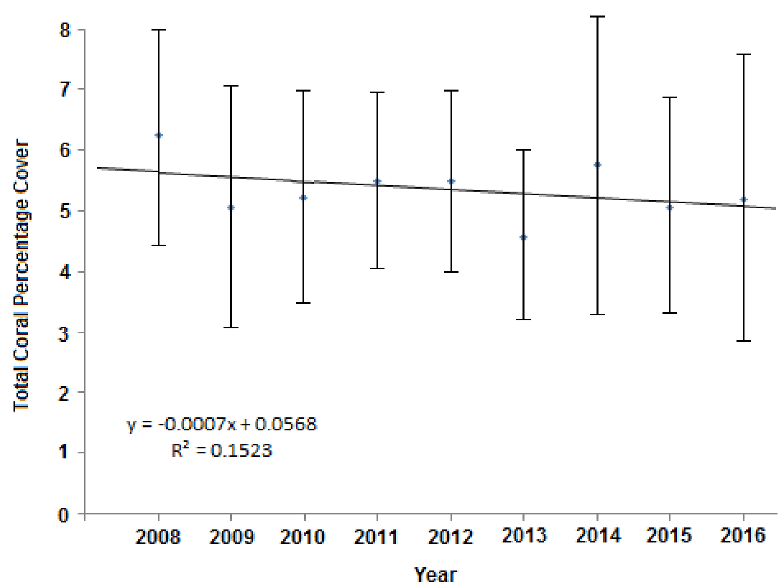

Figure 4 – Mean percentage coral cover for all sites combined across the study period. The trend illustrated shows a 0.5% reduction in coral cover from 5.6% to 5.1%, which equates to an overall decline of 11%. Regression equation is presented where x = year series number (2008 = 1, 2009 = 2, etc.).

Table 1 – Mean coral covers for each study site and year, with results combined from both habitat quadrat and line intercept methodologies. Site codes are: AN Anguillita; SA Sandy Island; LR Long Reef; LS Limestone Bay; SE Shoal Bay East; IH Island Harbour; SC Scrub Island; FR Forest Bay; LH Little Harbour; and SB Sile Bay. N/S Not Surveyed.

YEAR	AN	SA	LR	LS	SE	IH	SC	FR	LH	SB
2008	4.5%	12.5%	4.6%	5.1%	6.7%	1.2%	1.4%	1.8%	n/s	n/s
2009	3.2%	11.1%	9.4%	5.5%	5.6%	0.3%	1.5%	1.5%	2.3%	0.6%
2010	3.2%	9.4%	6.9%	6.9%	10.2%	2.1%	2.0%	0.2%	1.6%	1.0%
2011	3.7%	8.9%	4.2%	7.6%	9.3%	1.6%	2.9%	0.2%	2.1%	4.0%
2012	2.4%	6.8%	n/s	8.0%	9.0%	3.0%	2.4%	0.6%	4.3%	0.0%
2013	2.9%	7.5%	n/s	5.3%	6.0%	1.3%	3.6%	0.0%	1.8%	n/s
2014	3.2%	8.0%	n/s	9.0%	12.6%	2.0%	2.9%	1.4%	2.0%	0.6%
2015	3.1%	8.9%	7.4%	8.7%	7.9%	0.7%	1.1%	1.4%	3.0%	0.0%
2016	1.5%	6.4%	8.8%	11.5%	7.3%	1.6%	1.4%	0.7%	1.9%	1.4%
Means	3.1%	8.8%	6.9%	7.5%	8.3%	1.5%	2.1%	0.8%	2.3%	1.1%

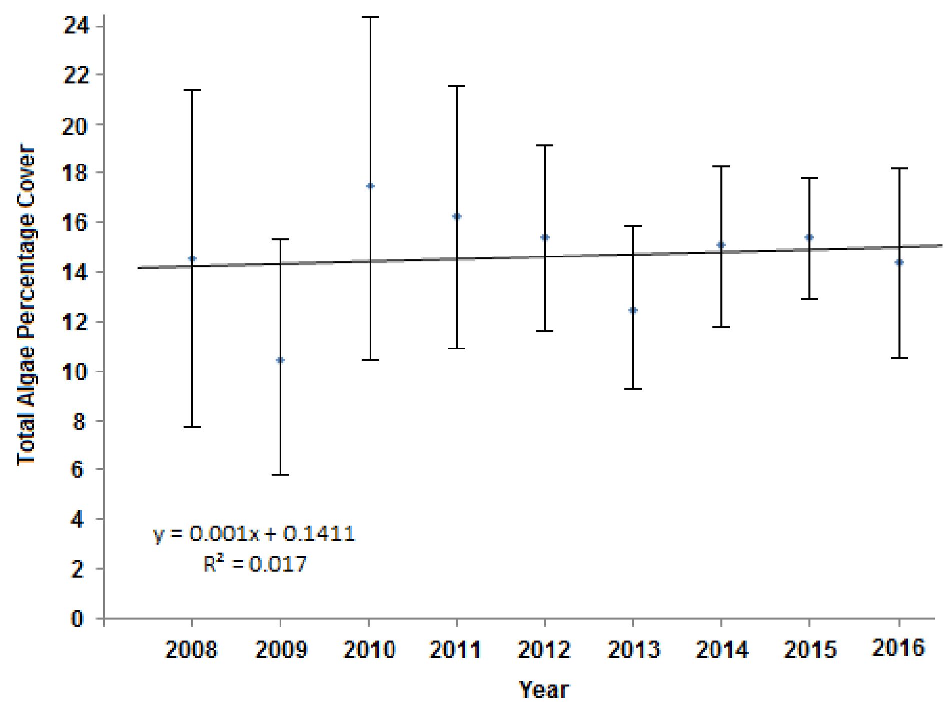

Figure 5 – Mean percentage macroalgae cover for all sites combined across the study period. Although relatively stable (2008 and 2016 values are almost identical), an overall increasing trend was identified with an increase of 9.5% from 14.2% to 15.0%. Regression equation is presented where x = year series number (2008 = 1, 2009 = 2, etc.).

Table 2 – Mean macroalgae covers for each study site and year, with results taken from habitat quadrat surveys. Site codes are: AN Anguillita; SA Sandy Island; LR Long Reef; LS Limestone Bay; SE Shoal Bay East; IH Island Harbour; SC Scrub Island; FR Forest Bay; LH Little Habour; and SB Sile Bay. N/S Not Surveyed.

YEAR	AN	SA	LR	LS	SE	IH	SC	FR	LH	SB
2008	0.9%	7.7%	0.4%	35.7%	9.6%	32.3%	21.7%	8.2%	n/s	n/s
2009	0.3%	1.7%	0.4%	17.3%	27.2%	20.7%	6.8%	7.8%	9.3%	13.1%
2010	5.4%	5.7%	3.6%	10.4%	20.5%	42.6%	19.7%	38.5%	8.5%	20.2%
2011	6.2%	13.3%	4.3%	22.7%	23.6%	26.5%	14.0%	30.3%	0.0%	21.9%
2012	5.9%	12.9%	n/s	21.3%	14.6%	28.6%	14.7%	13.2%	4.9%	22.8%
2013	5.4%	9.0%	n/s	9.2%	14.3%	24.0%	13.1%	17.0%	7.5%	n/s
2014	4.5%	10.6%	n/s	17.5%	24.4%	18.9%	11.7%	16.4%	12.7%	19.5%
2015	11.1%	10.0%	12.4%	22.3%	14.0%	14.9%	21.5%	21.2%	9.6%	17.6%
2016	8.0%	11.0%	5.3%	16.9%	17.9%	27.3%	16.9%	17.2%	6.1%	18.1%
Means	5.3%	9.1%	4.4%	19.2%	18.4%	26.2%	15.5%	18.8%	7.3%	19.0%

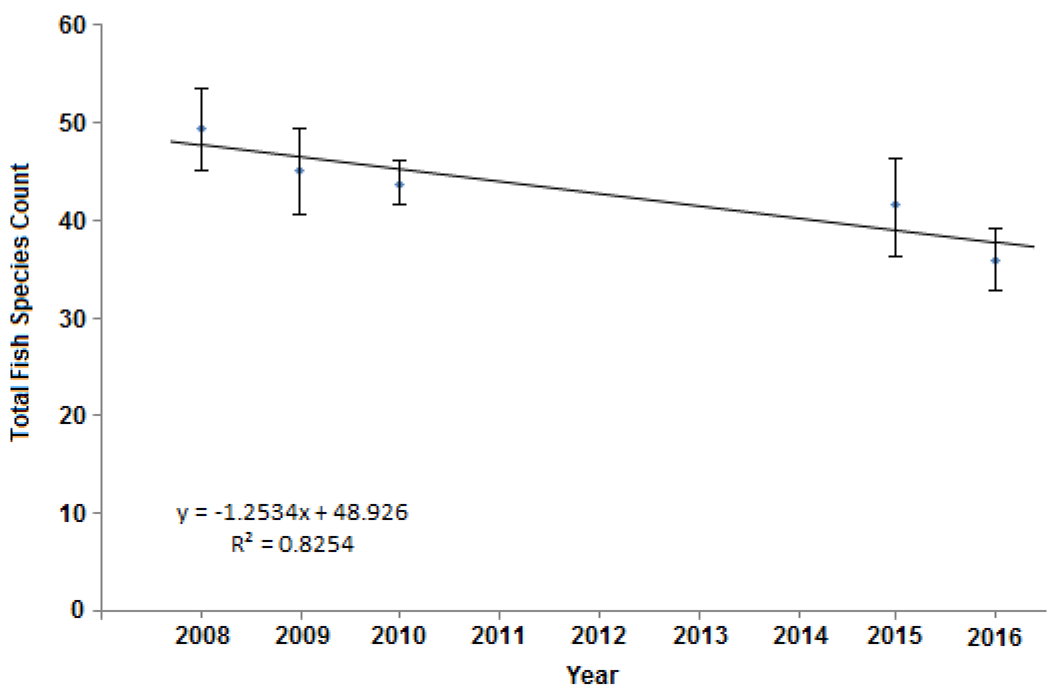

Figure 6 – Mean total fish species count for all sites combined across the study period using RDT methodology, showing a 21% drop in species number from 48 per site to 38. Years between 2011 and 2014 were removed from the analysis due to concerns over RDT data quality. Regression equation is presented where x = year series number (2008 = 1, 2009 = 2, etc.).

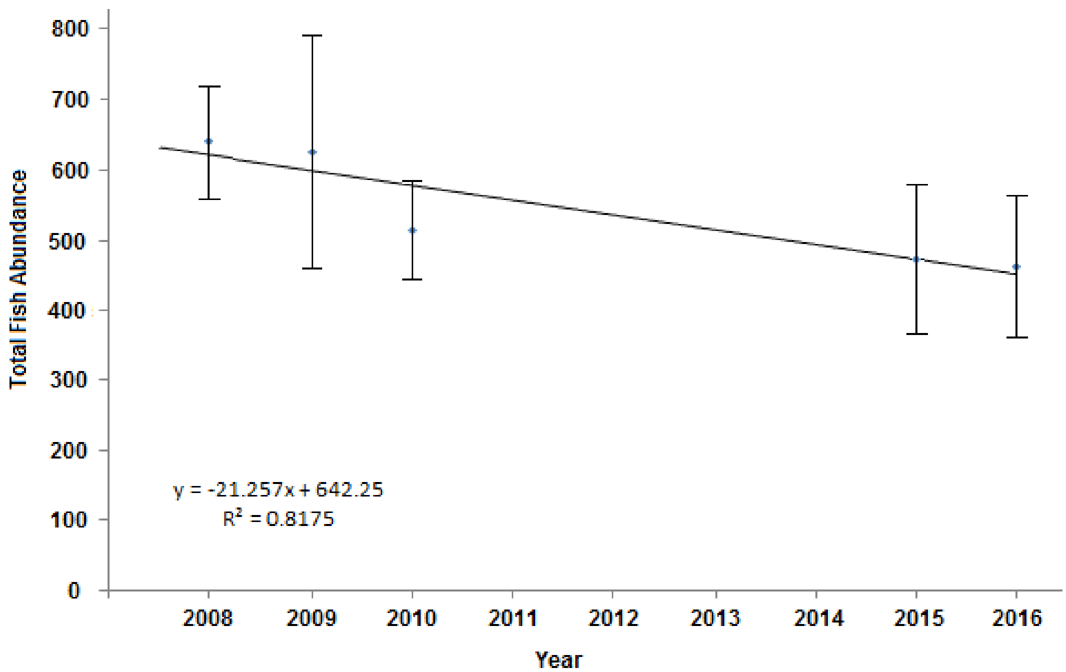

Figure 7 – Mean total fish abundance for all sites combined across the study period using RDT methodology, showing a 27% drop in overall abundance from 621 per survey to 451. Years between 2011 and 2014 were removed from the analysis due to concerns over RDT data quality. Regression equation is presented where x = year series number (2008 = 1, 2009 = 2, etc.).

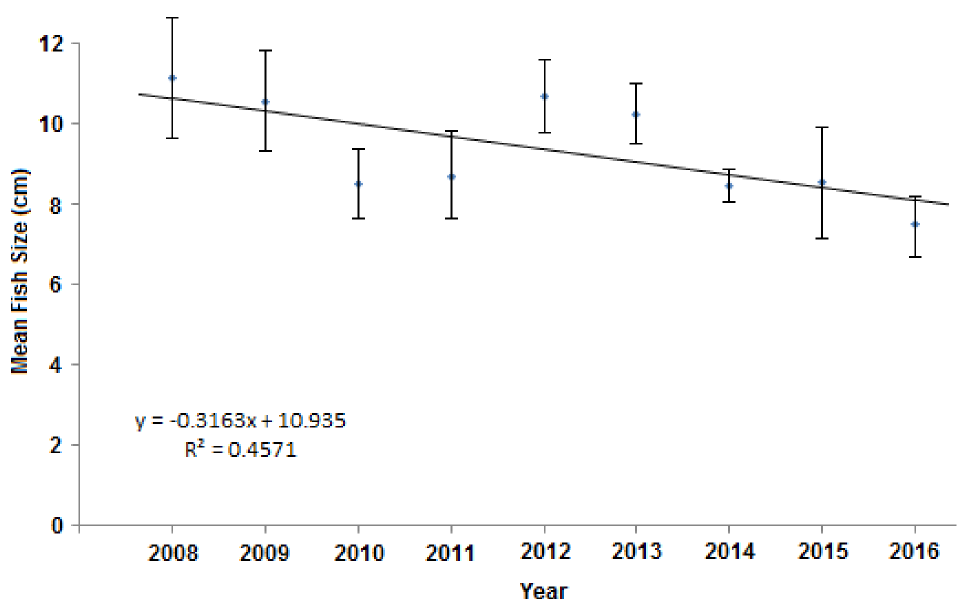

Figure 8 – Mean fish size (commercial and ecological importance) for all sites combined across the study period using belt transect methodology, showing a 24% drop in mean fish size, from 10.6 cm to 8.1 cm. Regression equation is presented where x = year series number (2008 = 1, 2009 = 2, etc.).

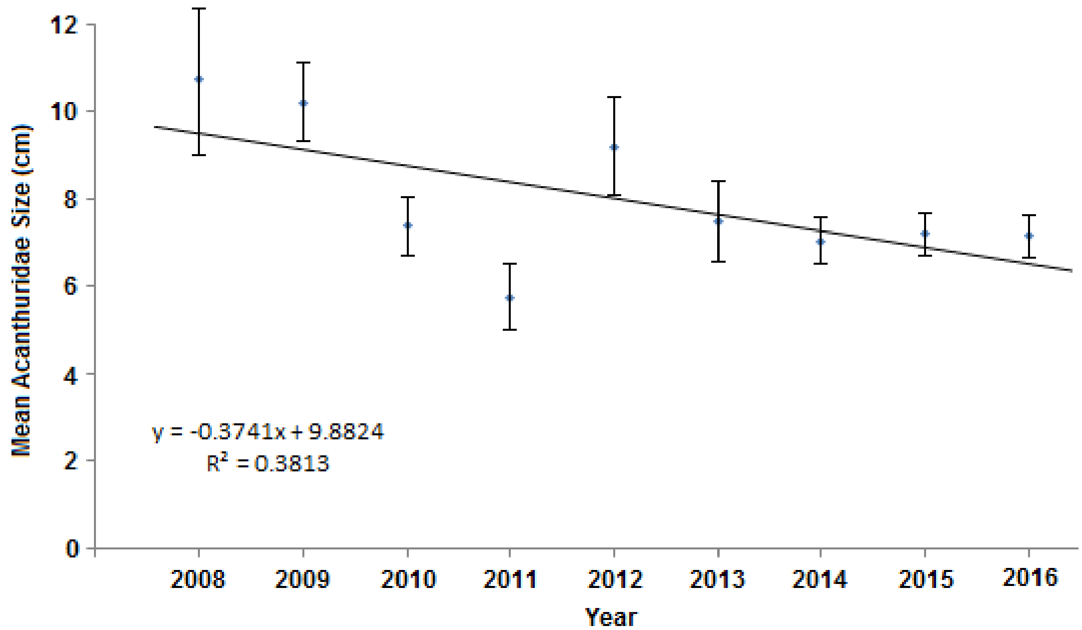

Figure 9 – Mean Acanthuridae (Surgeonfish) size for all sites combined across the study period using belt transect methodology, showing a 31.5% drop in mean size, from 9.5 cm to 6.5 cm. Regression equation is presented where x = year series number (2008 = 1, 2009 = 2, etc.).

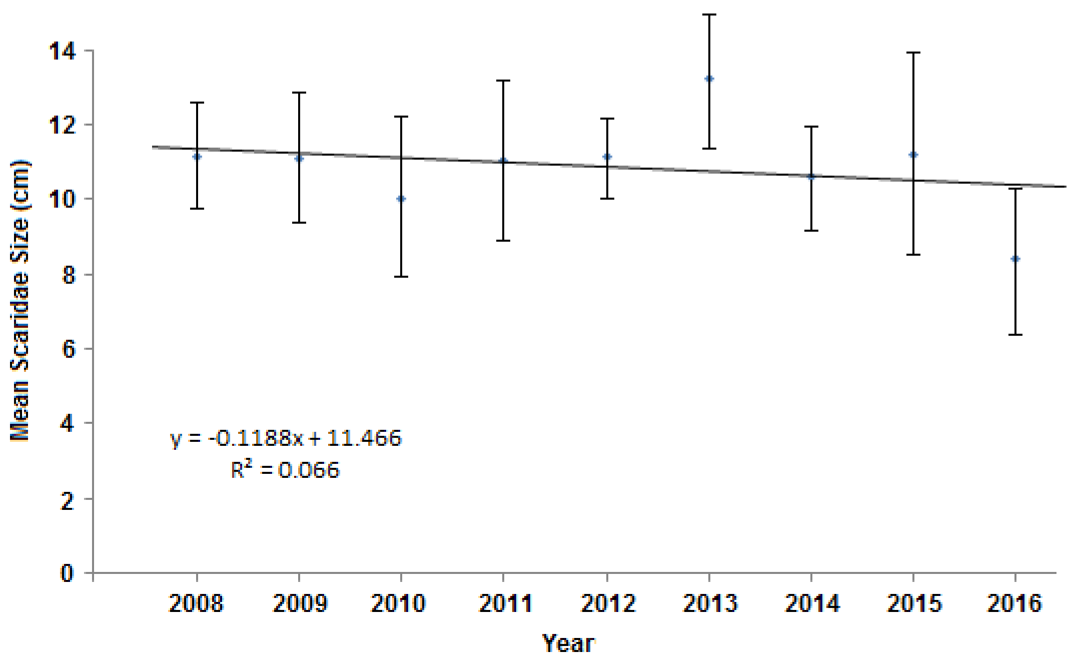

Figure 10 – Mean Scaridae (Parrotfish) size for all sites combined across the study period using belt transect methodology, showing a 9% drop in mean size, from 11.5 cm to 10.4 cm. Regression equation is presented where x = year series number (2008 = 1, 2009 = 2, etc.).

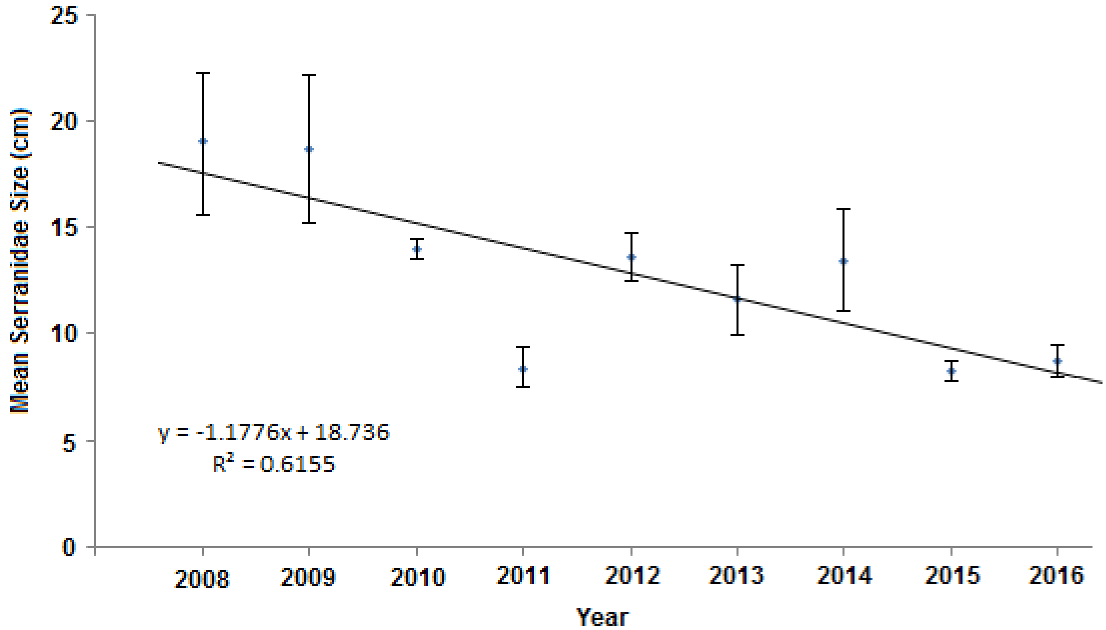

Figure 11 – Mean Serranidae (Grouper) size for all sites combined across the study period using belt transect methodology, showing a 54% drop in mean size, from 17.6 cm to 8.1 cm. Regression equation is presented where x = year series number (2008 = 1, 2009 = 2, etc.).

Seagrass site plant and algae cover

Key results from the seagrass monitoring sites are presented in Figures 12-14. Total plant and algae cover (including all seagrasses, *Halimeda sp*. and other algae types except cyanobacteria) was seen to increase over the study period (Figure 12), although when breaking this down into components an overall decline in *Thalassia testudinum* (turtle grass) was observable (Figure 13). The increased percentage cover seen in figure 12 was actually due to increasing cover of ‘other’ plants and algae as seen in Figure 14, which represents algal species other than *T. testudinum*, *Syringodium filiforme* and *Halimeda sp*. In particular, the overall increases were seen to be caused by changing levels of *Dasycladus vermicularis* (Fuzzy Finger Alga) and the invasive seagrass species *Halophila stipulacea*.

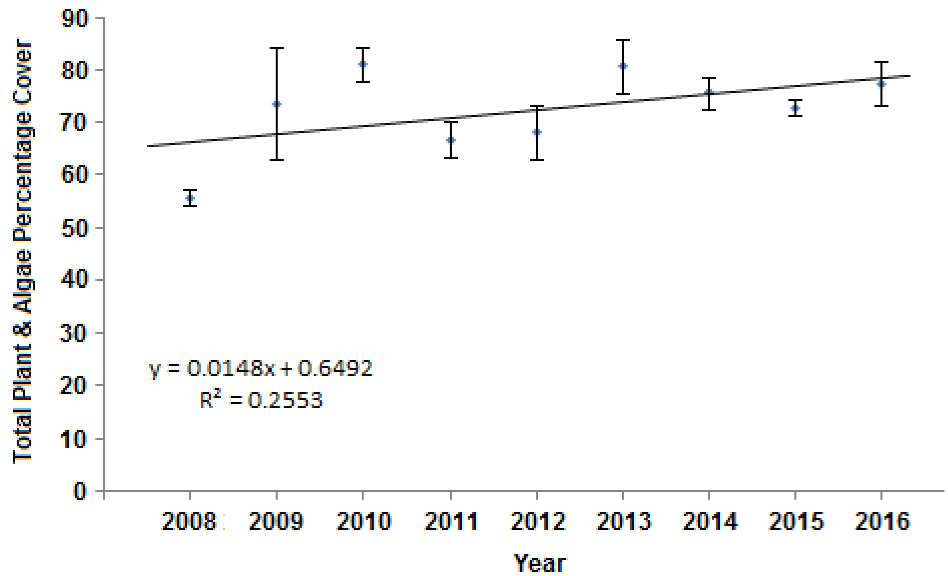

Figure 12 – Mean total plant and algae percentage cover, for all sites combined across study period using habitat quadrat methodology, showing a 15% increase in cover, from 66.4% to 78.2%. Regression equation is presented where x = year series number (2008 = 1, 2009 = 2, etc.).

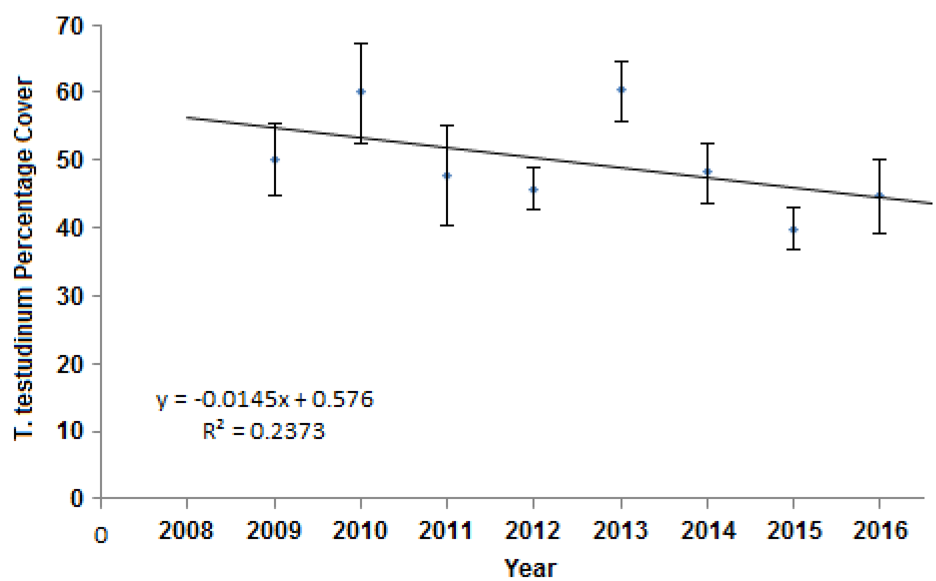

Figure 13 – Mean total *T. testudinum* percentage cover, for all sites combined across study period using habitat quadrat methodology, showing a 21% decrease in cover, from 56.2% to 44.6%. Regression equation is presented where x = year series number (2008 = 1, 2009 = 2, etc.).

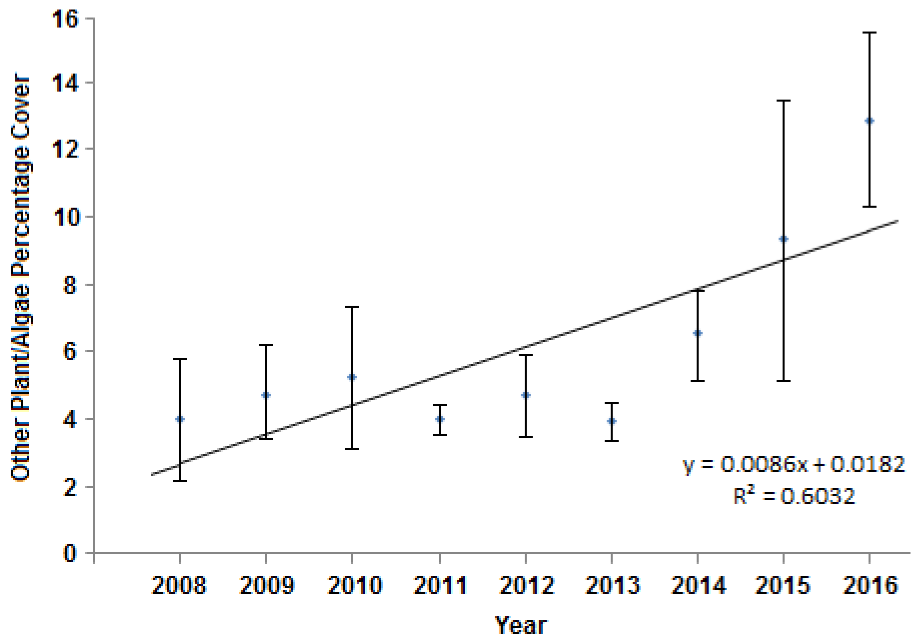

Figure 14 – Mean total 'other' plant and algae percentage cover (excludes *T. testudinum*, *S. filiforme* and *Halimeda sp*.), for all sites combined across study period using habitat quadrat methodology, showing a 72% increase in cover, from 2.7% to 9.5%. Regression equation is presented where x = year series number (2008 = 1, 2009 = 2, etc.).

Disease Occurrence

Coral diseases have been present in Anguillian waters since this study began. Images, with close comparisons of reefs as they were in Anguilla back in 1994 can be found in the Appendix (Plates 1-10). White band disease (Plate 6), yellow blotch disease (Plate 11), black band disease (Plate 12), and other various cyanobacterial infections (Plate 13-15) are the most common. White band disease has been noted only on *Acropora sp*. with yellow blotch disease primarily observed on *Orbicella sp*. Black band disease and cyanobacteria infections have been observed on a wide range of species, but primarily appear to affect *Siderastrea sp*. and *Diploria sp*. *Porites porites* is often seen fragmented and dying (Plates 8-10) but no evidence has yet been found that this is directly caused by disease or cyanobacteria, instead it often appears to be due to the smothering of small colonies by macroalgae, or overgrowing of sponges/zoanthids. The larger colonies of *P. porites* that remain appear to be gradually breaking down, but precise reasons for this remain unclear. It does not appear to be due to physical damage, but possibly more an overall weakening of their skeletal structure. Worryingly, the hardy species *P. astreoides*, the most common coral species in Anguilla has started showing signs of disease or cyanobacteria infections over recent years of monitoring. This either takes the form of pale or dark blotches (Plate 14) appearing on healthy tissue (in a similar way to that seen on *Siderastrea siderea*) or a fuzzy overgrowth of cyanobacteria filaments (Plate 15); fully recent dead *P. astreoides* colonies are now more frequently sighted than before.

D. antillarum Observations

No discernible trend in numbers of *D. antillarum* was identified over the study period at the monitoring sites, with some sites almost completely devoid of the species, other sites with stable but generally low populations, and others showing a mixture of increasing or decreasing numbers. Outside of the study sites, in areas where one might not expect to find them in such numbers due to a general low topography, very dense populations exist in orders of magnitude higher than that found on any of the monitoring sites. Such areas are usually close to the coastline and include: Little Bay; Isaacs Cliffs; Maundays Bay; Shoal Bay West; and Barnes Bay.

P. volitans Observations

Numbers of *P. volitans* were low across all monitoring sites, with sightings only taking place at Shoal Bay East and Anguillita. During extended swims away from the central site marker, and while exploring other areas around the island, *P. volitans* are generally uncommon and currently only found in high numbers at a small number of identified hotspots. Full results from this population study can be found in Wynne (2016b).

Water Quality

Water visibility during the 2008-2016 study period varied greatly, ranging between 5 m and 30 m. This clarity appeared to be influenced by weather conditions, with no discernible trends. Anecdotal reports, however, suggest that turbidity may have increased over recent years, and 'green water events' (as described in Wynne 2016), have occurred in Anguilla during this monitoring programme. While the most notable was in 2010 when bright green water appeared around the western end of the island (photographed in Wynne, 2010), other smaller algal blooms have been observed since then (Plate 16), although they are usually short in duration. Also, green tinged murky water is a frequent occurrence when conducting monitoring, especially on the south coast where conditions have become increasingly rough (Plate 17). A water quality pilot study was conducted to compliment the AMMP programme in 2008 (Wynne, 2009) but financial restrictions led to its discontinuation. This study concluded that the waters around Anguilla were showing signs of eutrophication, with variable turbidity. The signs of decreasing water quality also include thick coverings of cyanobacteria and/or fine filamentous algae on rocky reef areas, sand/rubble flats, and over seagrass beds (Plate 18). No other variables tested appeared a cause for concern.

Discussion

The first extensive ecological study conducted in Anguilla during 1990 by the Bellairs Institute, interestingly concluded that "Anguilla has a variety of diverse and attractive marine habitats which are in relatively good condition, with little apparent impact from human activities……the assertion that there has been limited deterioration of marine habitats in Anguilla is supported by comparing observations made in the present study with the *qualitative* descriptions given by Salm (1980)" (Oxenford & Hunte, 1990, page 162). This statement is in contradiction to the reports of white band disease sweeping through the region in the 1970's, which although not thought to be directly relating to human activities, decimated most of the *Acropora palmata* reefs that dominated at the time. This contradiction is likely due to the fact that the Bellairs study focused on areas where massive reef builders did not or had not in recent history been present. Based on the photographs presented in their report it appears that the sites classified as 'hard coral sites' were mixed reef communities with many gorgonians and ahermatypic hard coral species. If massive reef builders had existed at the Oxenford & Hunte (1990) study sites, reports of skeletal remnants would have presumably been made. Also, as the conclusions drawn in the Bellairs report were based upon comparisons with qualitative reports made by Salm (1980) rather than on quantitative ecological data, their margin for error increases and overall reliability is drawn into question.

It is far more likely that the areas studied in 1990 had been in gradual decline for a number of years but without ecological data as a baseline such a decline could not be identified. Such a gradual change would follow with the results of the comparison study conducted in 2009 (Wynne, 2010) that found coral cover had declined by between over 50-90% at most of sites studied by Oxenford & Hunte (1990), with the biggest decrease noted on the south coast at Forest Bay where a 99% reduction had taken place. This gradual decline appears to be continuing to this day, with an 11% decrease in coral cover noted between 2008 and 2016, from a 5.6% mean across all study sites to 5.1%. While this may only be an overall drop in cover of 0.5%, but it is very concerning when considering how low the mean percentage was to begin with.

From the photographic evidence it is clear that stands of *Acropora palmata* (and other reef builders) still existed when the 1990 Bellairs study took place, and could also have contributed to the conclusion that the marine habitats around Anguilla were in relatively good condition. However, the photographs were taken by a researcher who had seen the reef areas prior to 1990, and recognized that these were just remaining pockets of presently unaffected corals and overall corals were in severe decline (Bythell, pers. comm.). Indeed, by the time AMMP began, virtually all the massive reef builders had already disappeared. Today, *Acropora sp* now accounts for less than 2% of colonies recorded, which is extremely low when considering that at least five of the monitoring sites (Long Reef, Island Harbour, Shoal Bay, Forest Bay and Sile Bay) historically would have had very high coral covers, based on the skeletal remains that are still present there. It seems no recovery has taken place since the 1970 white band disease outbreak, and although recruitment of this genera is still taking place, and the occasional large healthy colony observable (Plate 2), the disease can still be seen affecting colonies. Other illnesses also appear to be affecting healthy colonies (white syndrome; cyanobacteria growth), with approximately 50% exhibiting some kind of mortality. Mortality has also now spread to hardier species such as *Porites astreoides* (Plates 14 & 15), which is especially worrying as such species were not seen to be affected by coral syndromes when monitoring began.

Mortality of corals in general over the last decade, for the majority of cases, seem related to water quality issues (i.e. nutrient enrichment, sedimentation) rather than pathogens (white band disease) or physical impacts (sea surface temperature causing bleaching events). A bleaching event for example was reported regionally in the Caribbean in 2005. Although no data were being collected in Anguilla at the time, and some incidences of bleaching were reported, Anguilla generally did not appear to have been as badly affected as other parts of the region. Conversely, cyanobacteria have been on the increase in Anguilla's waters (Gumbs, 2012) and can be seen to directly influence coral health – either through infection or smothering. Cyanobacteria benefits from increased nutrients and as such is related to water quality (Teta *el al*., 2017). Other increasing effects that can be potentially related to water quality are manifesting around the island and range from high turbidity and increasing sedimentation that can smother corals, to nutrient enrichment causing phytoplankton blooms (Plate 16) and facilitating macroalgae growth which can also have a smothering effect (Plate 17-18). Sources of sediment are generally seen to be from local sources (rain water runoff, salt pond breaches, coastal sediment disturbance from ground seas) and so mitigation measures may be applicable. Nutrient enrichment does not seem to have such clear relationships. Certain bays with relatively heavy small vessel traffic and close to the beach development (i.e. Road Bay and Island Harbour) appear more severely affected with increased filamentous algae growths (plate 18). This may be caused by increasing organics being introduced to the water though waste products (cleaning vessels and fish directly into the water) or through old and leaching septic tanks. Indeed, the latter may be an issue in many coastal areas, but development is still relatively limited in Anguilla as a whole and so cannot explain the apparent reduction in water quality taking place. Alternatively it has been suggested that nutrient-rich waters emanating from St Martin may be responsible for the situation (C.A. Samuel, pers.comm.), but while it is recognized that this may make a contribution, prevailing current directions mean it cannot have a particularly noticeable effect. An alternative theory put forward (Wynne, 2016) is that nutrient rich water is in fact arriving via North Brazilian current rings (Fratantoni & Richardson, 2006) that also go on to explain how Sargassum and the recent Caribbean inundations may have originated from the Atlantic and multiplied rapidly after hitting this nutrient rich water (Gower *et al*., 2013). It is also theorized that this water may have been responsible for the green water event recorded in Anguilla in 2010 (Wynne, 2010) and may also influence fish assemblages (Johns *et al*., 2014). Anguilla, with its small population size, small scale developments, and limited agriculture makes an interesting case study that adds credence to this regional eutrophication theory.

The small overall increase in macroalgae recorded over the monitoring period also suggests a potential increase in available organic nutrients (Reopanichkul *et al*., 2009) although some studies have questioned the validity of this assumption (Sotka & Hay, 2009) despite others critiquing such manipulative experiments (Thacker, *et al*., 2001). A number of studies have however failed to identify the link between nutrients and macroalgae (Littler et al., 2006; Burkepile & Hay, 2009), which could explain the minimal increase trend even though eutrophic conditions are present (Wynne, 2008b). It is also interesting to note that this trend does not appear to be related to *D. antillarum* recovery (or lack thereof) as monitoring sites that did show a decrease in macroalgae (for example Limestone Bay) actually suffered a decrease in *D. antillarum*. Furthermore, the patchy nature of current *D. antillarum* populations in low relief non-reef areas make such inferences problematic. It is also not necessarily clear as to whether this macroalgal trend (although not overly strong) has been influenced by grazing fish trends. Overall, although total fish abundance, number of species present, and mean fish sizes have decreased over time, a more significant effect on macroalgae coverage would be expected (Angelo & Wiedenmann 2014). Reasons for this remain unclear, and warrant more research. Recent publications have however found similar anomalies (Suchley *et al*., 2016) and so the situation found in Anguilla is not necessarily unique.

It is clear, however, that fish diversity, abundance and sizes are all decreasing and, in all likelihood, this is due to a combination of factors, primarily habitat degradation and extractive fishing practices. Interestingly, the trends found with regard to Scaridae (parrotfish) species were much weaker than expected which is encouraging, as although they still show an overall size reduction, it means even with larger size classes disappearing recruitment is still taking place at a sufficient level for recovery to happen if stressors were addressed. In all likelihood it is spearfishing that is causing this pressure, especially at monitoring sites such as Limestone Bay, Shoal Bay East, and Island Harbour. The lowering mean size class of Serranidae (grouper) species is however more of a cause for concern, as it is a popular target group and a favourite within local markets. Probably for this reason, smaller sizes are increasingly seen for sale at retail outlets, suggesting management measures are needed if this important fishery resource is to be sustainable into the future. Acanthuridae (surgeonfish) sizes were also similarly in decline, although they are still often seen in large schools, even in the more ‘unhealthy’ areas (for example Forest Bay). Again, in a similar respect as parrotfish it is hoped that if management measures were introduced this negative trend could still be reversed. To reverse these trends habitat degradation would also need to cease, and this unfortunately seems unlikely unless dramatic region-wide legislative changes take place.

The declines/changes seen are not just limited to hard bottom/reef areas, as the illustrated trends in Figures 12 to 14 show for seagrass monitoring sites. Here, an overall decline in the important native seagrass species (*Thalassia testudinum*) was recorded. Gumbs (2012) noted this decline in Crocus Bay, but it now also shows as an overall decline when working out means for all sites combined. This is a cause for concern as the plant stabilizes sand, provides shelter for juvenile fish and also acts as food for foraging sea turtles. When looking at means of total plant/algae cover however, the trend becomes an increasing one. By separating plant/algae groups out slightly, this increase can be seen to be largely caused by the invasive species of seagrass *Halophila stipulacea*, and members from the category ‘other algae', especially that of *Dasycladus vermicularis* (Fuzzy Finger Alga). *D. vermicularis* is filamentous in appearance and generally appears to increase in abundance in areas suspected to have poor water quality (i.e. Sandy Ground), whereas *H. stipulacea* is now present in and around almost all seagrass study sites. The species is today considered common whereas when monitoring started it was not present, and even its Caribbean relative *Halophila baillonis* (Midrib Seagrass) was only rarely sighted. Even though considered invasive because it has the potential to outcompete *T. testudinum,* recent studies suggest *H. baillonis* may benefit areas by being more virulent with a greater potential to spread into areas otherwise devoid of seagrass (aiding sand stabilization). This goes on to increase the amount of food available to species such as *Chelonia mydas* (green turtles) and potentially to support a larger more species-rich fish population (Rogers *et al*., 2014).

As well as the five key indicators primarily explored within this report (coral cover, algae cover, fish diversity, fish abundance, fish size), other potential indicators exist to ascertain habitat health. One proposed by Wynne (2016b) is the presence of *Pterois volitans* (lionfish). The 2015-2016 study noted that *P. volitans* had a tendency to be higher in density in 'healthier' areas which, based on the five key indicators, would include areas with more juvenile fish. The presence of juvenile fish adds to ecological complexity and itself requires a certain level of habitat health (habitat complexity and food items, for example). Throughout monitoring, no lionfish were sighted on the south coast, and were only ever recorded within the Anguillita and Shoal Bay East monitoring sites, although sightings were also made close to the Limestone Bay and Scrub Island locations. This fits well with the overall relative health index in Wynne (2016b) despite the fact that site scores will have changed from this work if the analysis were to be re-run. Another relatively easy variable to measure that might be used as a future indicator is water turbidity. High turbidity is often associated with lower water quality (Reopanichkul *et al*., 2009) and therefore could be used as an indicator of poor habitat health (Fabricius, 2005). Once again, this fits with the findings here where the sites located along the south coast, which are often very turbid, are in a far more degraded state than most north coast sites where turbidity is usually lower. Furthermore, the most degraded north coast site (Island Harbour) is the most likely to be suffering turbid conditions. Worryingly, although no direct trends were uncovered, turbid conditions appear to be increasingly common around the whole island, which if used as an indicator, does not bode well for future habitat health and reversing the trends recorded over the last ten years.

Conclusions

Overall, ten years of monitoring has confirmed that Anguilla's coastal marine habitat health is in general decline. Trends identified include: a slow but steady decrease in coral cover; an increase in macroalgae cover; a reduction of fish species richness and abundance; and a decline in mean fish size of commercially and ecologically important species. These trends, if they continue, will lead to significant problems for the economic future of Anguilla due to its dependence on coastal areas (both marine and terrestrial) for much of the island's income. Indeed, such impacts already appear to be manifesting with changes in erosional regimes noted in some important tourist areas, some of which are thought to be directly related to habitat degradation of important near-shore reef areas (Wynne *et al*., 2016).

This degradation trend, although somewhat variable, is generally considered to be Caribbean-wide, and thought to have begun with white band disease decimating extensive *Acropora palmata* reefs in the 1970s. This event was not caused by local anthropogenic factors, but is thought to have originated either via African dust, through the opening of the Panama Canal, or ships ballast water. The *D. antillarum* mortality event followed this a decade later, with the pathogen origin again unknown, but thought to have arrived from the same source as white band disease. Low coral recruitment, mainly due to low remnant population size and algal overgrowth, inhibited the massive reef builders recovery, and throughout the 1990's coral diseases continued to proliferate and algae continued to grow, mainly driven by low *D. antillarum* numbers, overfishing, and nutrient enhanced fertilization. By the early 2000's coral cover had become so low, with such a paucity of massive reef builders, that continued decline was difficult to discern and shifting baselines meant that sighting a large *Acropora palmata* colony led to great rejoicing and the conclusion that corals may be recovering. With the initiation of AMMP in 2007 these trends began to be quantified and, after ten years, the continued overall decline of habitat health around Anguilla has now been quantified and confirmed.

Although many of the reasons for this continued decline are regionally based, there are some local management measures that should be introduced immediately in an attempt to mitigate the situation. If Anguilla continues under the 'business as usual' scenario marine habitats will continue to decline, erosion will increase, coastal fisheries will collapse, and almost all livelihoods around the island will be impacted. While these management measures will not solve the situation and, in general, without regional/global change the future of coral reefs around the entire Caribbean is bleak, this is not a reason for inaction or complacency. For example, by focusing on existing Marine Protected Areas in Anguilla, and trying to maintain relic populations, one gives nature the best chance possible to bounce back if the necessary regional changes happen. Also, by minimizing runoff and leaching, even if regional nutrient enrichment is an over-riding force, it may be sufficient to level off and halt continued habitat degradation. From this, the following management recommendations can be made[2].

[2] This list does not include the ongoing need for DFMR to continued public awareness and education on marine and coastal environmental issues, and also continue the push for legislative change

Recommendations

- Marine Parks should be offered the highest level of protection possible with fishing limited to hook and line, pollution fines established, and all legislation (anchoring etc) strictly enforced.

- Minimum sizes should be introduced for all key reef fish species. Full protection should be given to rare species that have a commercial tourist value (i.e. *Epinephelus itajara* and *Epinephelus striatus*).

- Artificial reef projects should be encouraged as both species aggregating devices for fishery seeding purposes and tourist attractions. Careful consideration is needed if such structures are to be used as erosional barriers, as they have a low potential to positively affect erosional regimes. Coral nursery projects are not recommended until the reason behind continued coral mortality is correctly identified and reversed.

- Runoff from developments need careful and strict regulation and all septic tanks should be routinely inspected and leaching threats identified. Any coastal properties with septic tanks older than twenty years need special monitoring. A coastal setback should be introduced for both general construction of buildings and the location of such septic systems.

- A full comprehensive sea water quality monitoring programme should be established and funded by the Government of Anguilla so as to better understand reasons behind habitat health decline as well as in the interests of public health and tourism.

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Appendix

Photographic Plates 1 – 18

Plate 1 – Taken in 1994 (J. Bythell) in an offshore area of the fore-reef northwest of Seal Island. It illustrates expansive, healthy, *Acropora palmata* stands and shows that pockets still existed after the white band disease outbreak took place in the 1970's. No such pristine areas exist today anywhere in Anguilla waters.

Plate 2 – Taken in 2015 (S. Wynne) in approximately the same location as that pictured in Plate 1. This is probably one of the 'healthiest' areas of *Acropora palmata* reefs remaining today, comprised mostly of dead skeletal remains although scattered live colonies are present: some diseased or degrading (bottom left) while others remain intact (middle). The following (Plate 3) illustrates the situation on most of the old *Acropora palmata* reefs where dead skeletal remains are covered in a thick layer of macroalgae.

Plate 3 – Taken in 2007 (S.Wynne) in an area in the Shoal Bay - Island Harbour Marine Park. This is the typical state of many of the old *Acropora palmata* reefs found around Anguilla. Long dead coral skeletons gradually erode under the forces of nature, and with few new colonies growing to repopulate their future as a storm defence appears limited.

Plate 4 – Taken in 1994 (J. Bythell) in an unidentified area possibly somewhere in the region of Sandy Island. It illustrates expansive, healthy, *Acropora cervicornis* thickets. Today not even the remnants of these areas are identifiable, with the species now reduced to a few rare small scattered colonies (Plate 5 & Plate 6).

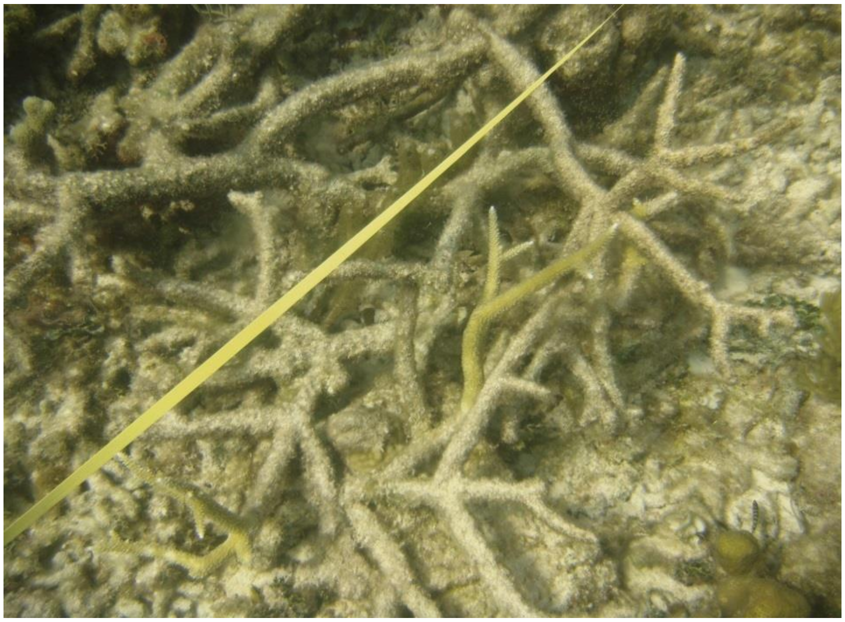

Plate 5 – Taken in 2015 (S. Wynne) within the Sandy Island monitoring site (transect tape is illustrated in photograph). *Acropora cervicornis* no longer forms dense thickets as pictured in Plate 4, but instead is only seen on rare occasions as small colonies, and often diseased (Plate 6) or dying as seen here.

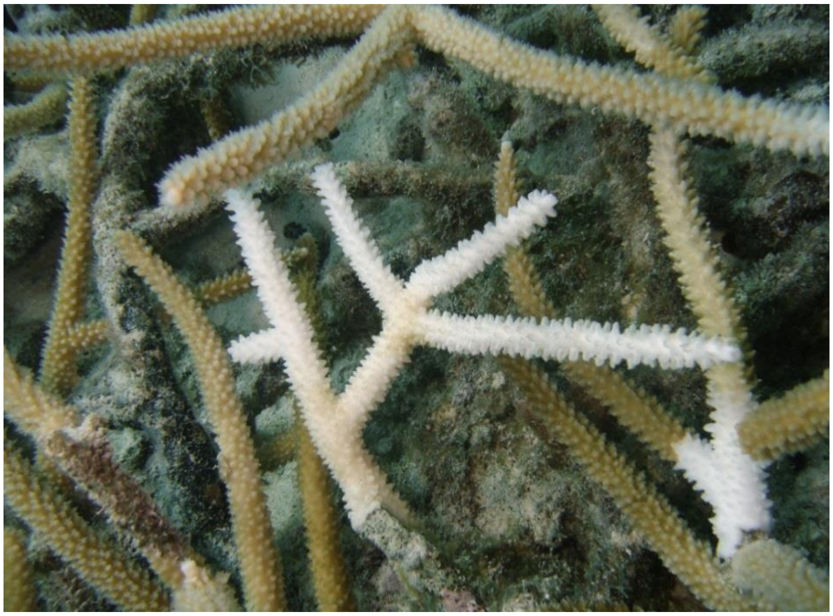

Plate 6 – Taken in 2007 (S. Wynne) within the Sandy Island monitoring site and illustrating White Band disease on *Acropora cervicornis*. When monitoring began, this species was fairly common around Sandy Island (relative to the rest of Anguilla), but today colonies are less frequently sighted and more likely to be sick looking colony remnants (Plate 5).

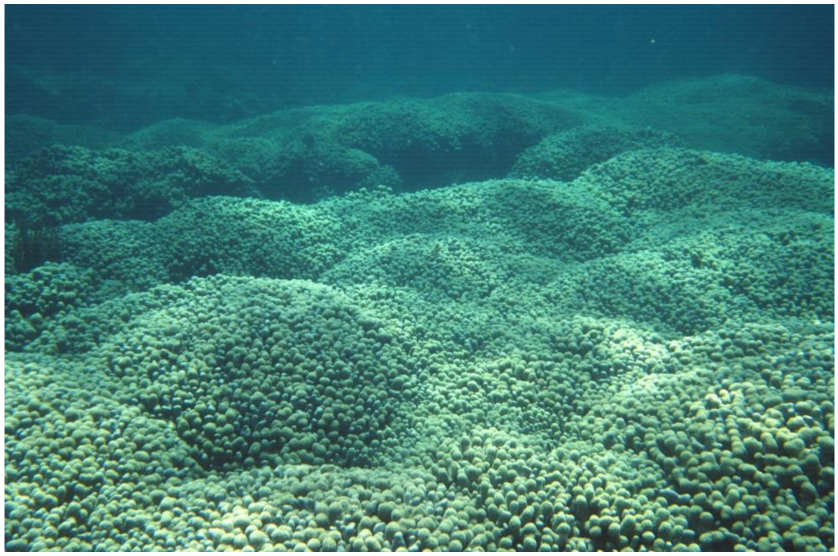

Plate 7 - Taken in 1994 (J. Bythell) in an offshore area east of Seal Island and illustrating an expansive and healthy *Porites porites* bed. This is estimated to be in the same area as the Long Reef monitoring site, as with the following image below taken in 2009.

Plate 8 - Taken in 2009 (S. Wynne) in the same offshore area pictured in the previous image. It illustrates an expansive, but slowly degrading, *Porites porites* colony. Compared to other large colonies today, this is probably the most intact example yet encountered, with most now consisting almost entirely of rubble.

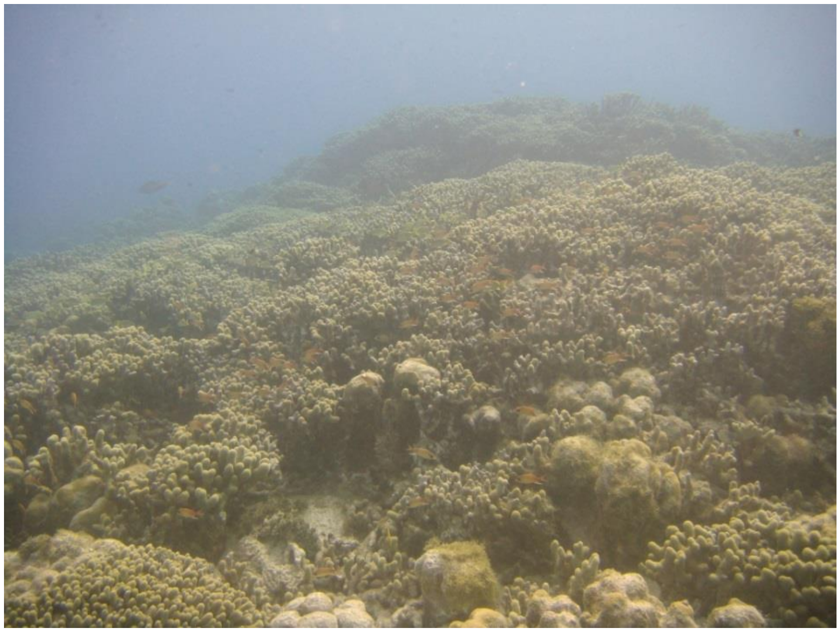

Plate 9 - Taken in 2008 (S. Wynne) close to the Sandy Island monitoring site. Illustrated is an expansive, but slowly degrading, *Porites porites* colony. This colony has continued to degrade over the study period as can be seen (from a different angle) in plate 10.

Plate 10 - Taken in 2015 (S. Wynne) close to the Sandy Island monitoring site. Illustrating the degradation of an expansive *Porites porites* colony previously pictured in plate 9. Although the photograph is taken from a different angle, it clearly shows how the colonies underlying skeletal structure appears to be weakening and collapsing over time.

Plate 11 – Taken in 2007 (S.Wynne) illustrating yellow blotch disease on *Orbicella annularis* in Shoal Bay East – Island Harbour Marine Park. This disease continues to be prevalent today, and primarily affects this coral genera only.

Plate 12 – Taken in 2009 (S. Wynne) illustrating black band disease on *Siderastrea siderea* in rocky area close to Crocus Bay. This disease isn't particularly common in Anguilla, but a cause for concern is its ability to attack and kill this otherwise hardy coral species.

Plate 13 – Taken in 2009 (S. Wynne) illustrating cyanobacterial overgrowth on *Colpophyllia natas.* Over recent years conditions such as this have been seen in seemingly increasing frequency. It ultimately results in the death of the colony.

Plate 14 – Taken in 2010 (S. Wynne) illustrating pale blotches on *Porites astreoides*, a usually hardy species. Note how cyanobacteria has started to grow in the center of some of the larger blotches.

Plate 15 – Taken in 2015 (S. Wynne) illustrating diseased colony of *Porites astreoides*, usually considered a hardy species. Note how filaments of cyanobacteria are growing over live discoloured tissue, with recently dead areas having thicker coverage.

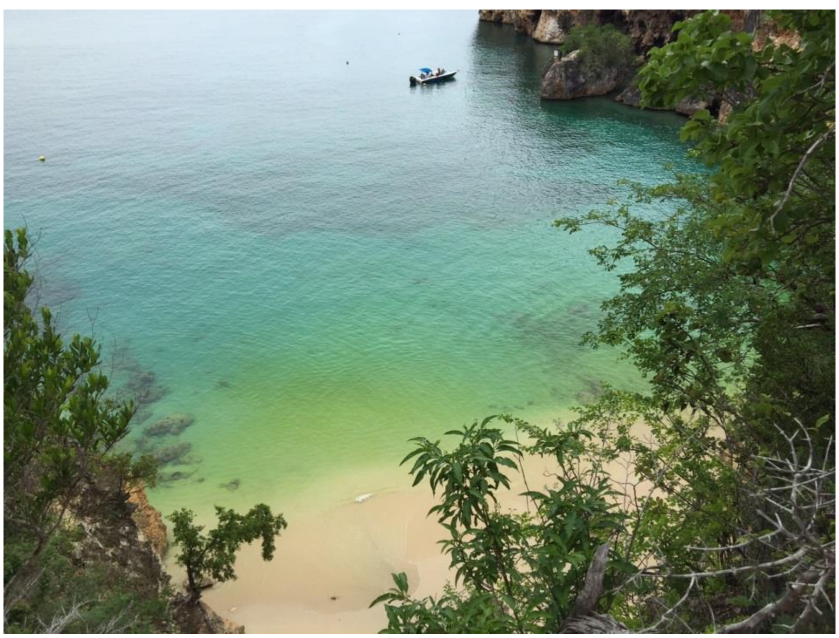

Plate 16 – Taken in 2015 (S.Wynne) illustrating small localized algal blooms that appear to be becoming more frequent around the island. This one was observed in Little Bay and lasted approximately two days.

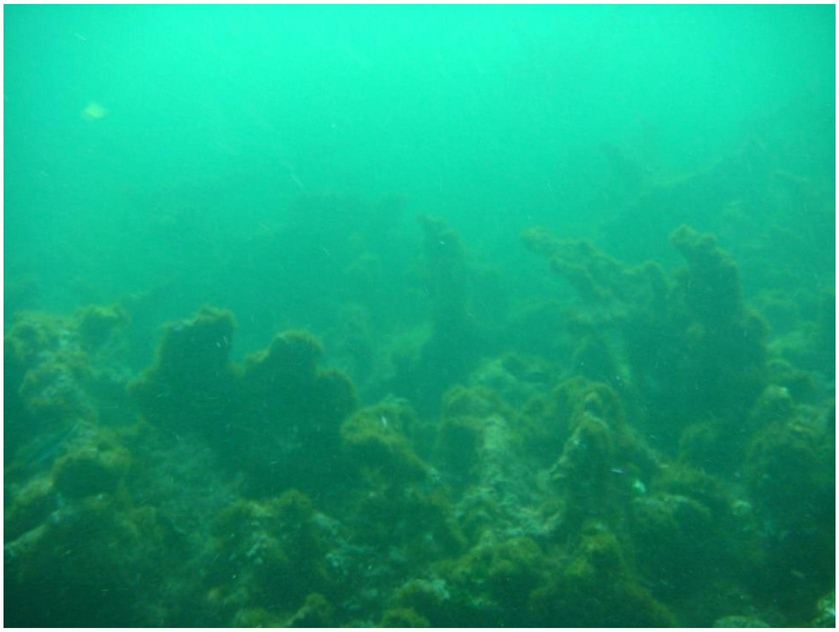

Plate 17 – Taken in 2015 (S.Wynne) illustrating murky green tinged water observed on the south coast while conducting surveys at Sile Bay monitoring site, together with a thick cover of macroalgae over an old *Acropora palmata* reef.

Plate 18 – Taken in 2010 (S.Wynne) illustrating fuzzy filamentous algal growths over the seagrass bed in Island Harbour.