Cayos Cochinos marine science report – 2011

Prepared by Operation Wallacea on Behalf of the Honduran Coral Reef Foundation

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Executive Summary

Operation Wallacea operates a multidisciplinary marine research program in Honduras. Investigations are conducted at two research stations, one within the Cayos Cochinos Marine Protected Area (CCMPA) and another on the island of Utila, working in partnership with the Honduran Coral Reef Foundation (HCRF) and the Coral View Research Station respectively. In previous years, a mobile social science team has also operated within the Cayos Cochinos and the communities on the adjacent mainland, but, during the 2011 season no social science team was in operation.

The Cayos Cochinos biodiversity teams focus on both marine and terrestrial research around the islands and associated coral reef systems. These biodiversity teams can be broadly grouped into four specific areas of research, focusing on 1) reef fish, invertebrate and benthic communities, 2) urchin abundance and distribution, 3) Queen Conch population ecology, and 4) population ecology and conservation of endangered island herpetofauna. The work can roughly be divided into two objectives for all groups, the first is to conduct a survey of the populations and densities of the species within the research remit, while the second is to study the ecology, behaviour and interactions of these species with their environment and other species. These two objectives combine to provide an assessment of the current status of the MPA and provide insights into each habitat or species that will be invaluable in their future conservation. Generally the results of the monitoring program have been analysed and reported within this report while the results and data from the more complex second objective is still being processed with the output expected to be produced in individual publications in the near future.

General results of the work indicate that the marine environment is subjected to a variety of different stresses and threats, but that there are some positive results where encouragement can be taken mixed in with some larger areas of concern for the overall health of the system. Certainly the main result from the marine research groups is the dominance of algae over hard corals. Given the common occurrence of phase shifts from corals to algae within the Caribbean this remains a major cause for concern. Results pertaining to the conservation of the herpetofauna of the Cayos Cochinos are also outlined.

Contents

1: **Introduction**

Operation Wallacea has been conducting Marine Research in Honduras since 2003. The program, which started in the Cayos Cochinos archipelago, has now developed to encompass research groups in a number of locations in the Honduran Bay Islands and on the mainland in addition to various associations with local and national groups within Honduras.

The basis of this work is to bring academics and world leaders into the field to run small specialist research groups within the overall project framework. These groups consist of principle researchers, Ph.D. students, dissertation and thesis students and research assistant volunteers. The projects run over a 10 week period every summer (June - August). This format gives many advantages to field research, such as bringing together a wide variety of multidisciplined field scientists with varied backgrounds into the same place with a central organisation coordinating the research. Funding for the research is entirely based on volunteers, this ensures that projects can be run over prolonged periods and datasets can be built up over many years. The research is based on collecting data on the local ecosystems that can be channelled into high quality research publications and grant applications to establish examples of best practice conservation in the local communities.

The marine research and conservation objectives of Operation Wallacea in Honduras often require a multidisciplinary approach, utilising expertise from a variety of research backgrounds or data collection from a variety of locations. To achieve this, Operation Wallacea has established three independent research operations within the marine program. Two of these are based at permanent research centres and concentrate on studying the biodiversity and ecology of the local marine and terrestrial ecosystems. The first of these is run from the Marine Research Station on the island of Cayo Menor within the Cayos Cochinos Marine Protected Area (CCMPA) and the other is based at the Coral View Research Centre on the island of Utila. The third operation, a mobile social development research team, did not operate in 2011 as data from previous years are currently being analysed and results are being made available separately in various reports.

Cayos Cochinos

The Cayos Cochinos Islands were established as a Honduran National Monument in 1993 and given protection through the establishment of the Cayos Cochinos Marine Protected Area (CCMPA) at the same time under the management of the newly established Honduran Coral Reef Foundation (HCRF). Operation Wallacea started working with the HCRF within the CCMPA in 2003 at the Marine Research Station situated on Cayo Menor.

The main objectives of the research within the CCMPA are;

- Yearly monitoring of the status of the reefs within the Marine Protected Area, to determine the health of the reef system and the success of the CCMPA in protecting the reef systems.
- Conducting high quality marine and terrestrial research within the MPA, producing publications suitable for peer review and establish the Cayos Cochinos MPA and Marine Research Station as an internationally recognised centre for quality marine research.
- Assess the population levels, ecology, behaviour and phylogenetic distinctiveness of the Cayos Cochinos Boa Constrictor
- Assess the population levels, ecology and behaviour of the critically endangered Spiny tailed Iguana (*Ctenosaura melanosterna*).

2: Locations of Study

Cayos Cochinos

The Cayos Cochinos Islands are located about 18km off the northern Honduran shore and comprises the very southern end of the Meso-American Barrier Reef System (MBRS), the second largest barrier reef in the world. The islands and surrounding seas were designated as a National Marine Monument by the Honduran Government in 1993 and remains the only such area in Honduras. The islands have been established as a protected area by the Honduran Government under the banner of the Cayos Cochinos Marine Protected Area (CCMPA) and managed by the Honduran Coral Reef Foundation (HCRF). This agreement established specific protection for the reefs and the wildlife on the islands. These included a limit on fishing in volume and species and established protection for the indigenous reptile species on the islands.

The protection of the Cayos Cochinos archipelago was formed in partnership with the World Wildlife Fund (WWF), The Nature Conservancy (TNC) and the HCRF and resulted in the publication of the Cayos Cochinos Management Plan (www.cayoscochinos.org). The main objectives of the Management Plan included the protection of the reefs and terrestrial systems and to ensure that local artisanal fishing communities were appropriately compensated for their loss of income, due to the proposed fishing restrictions, through the establishment of alternative incomes.

The Marine Protected Area

The Cayos Cochinos Marine Protected Area (CCMPA) is based around an archipelago of small islands and shallow seas. There are two main islands within the area, Cayo Mayor and Cayo Menor. Cayo Mayor with a width of 1.8km and length of 1.7km is slightly larger than Cayo Menor that has a width of 1Km and length of 1.3km. In addition to these two main islands there are 14 small Cays within the archipelago. The CCMPA contains a wide variety of marine habitats, with reefs to depths of 30m+, extensive seagrass systems and large areas of bare sediment. These habitats are distributed around the park, often nearer the islands and cays, although several shallow reefs are found in the open sea. Despite this diversity of habitats and islands the reserve does not have a significant mangrove system, with only a few mangrove trees found on Cayo Mayor within the MPA.

Settlements and industry

Cayo Mayor has a resident population year round, with a small artisan fishing community in East End in addition to a small hotel called the Plantation Beach Resort and several private homes. Cayo Menor has no local communities or industry and has been preserved solely for research purposes. This research is based at a small research centre that contains a dive centre, laboratory facilities, several permanent accommodation buildings and catering facilities. The only long term residents on the island are a handful of Navy guards and occasional researchers. The other settlement within the CCMPA is another fishing community on Cayo Chachahuate.

The industry of the area was largely based on artisan fisheries within the two communities of East End and Chachahuate, these have both been impacted by the restricted fishing rights dictated by CCMPA management plan. However, part of this plan established ensures that alternative income sources were established for these communities, and those on the North Honduran Coast, to

compensate for the loss of revenue from fishing activity. This has involved the promotion of ecotourism and the research station in the area. The Islands are now also used as the setting for the Spanish, Italian and Colombian versions of the Survivor television series, generating considerable income for the area.

Areas of research interest

The Cayos Cochinos archipelago offers a wide variety of marine and terrestrial habitats for surveys and experimentation, this allows a number of unique research opportunities within the area. Of particular interest is studying the success of the CCMPA, and the various fishing restrictions that apply to different species and areas of the park. This includes both fish and commercially and ecologically important invertebrate species.

The long term arrangement between Operation Wallacea and the HCRF gives a rare opportunity to collect long term data sets over many years that may reflect the status of the CCMPA or possibly larger scale patterns such as global warming and an impact assessment should the islands be hit by a hurricane.

East End

East End is the sister community of Rio Esteban, situated on the north side of Cayo Menor. The community has an approximate population of 22 residents living in 19 houses along a single 200 m stretch of beach, peaking to a maximum population of 90 during the fishing season (April-September). The community has one primary school with two classrooms that serve all of the MPA. The area also has a Honduran Navy station where the Navy personnel employed to patrol the MPA reside. East End has some tourism development, receiving US AID funding in 2007 to build cabanas and a restaurant. These were ready for use in summer 2008 and were used by both the Operation Wallacea social science and Herpetofauna Research Groups. The immediate marine environment consists of inner reef flats of approximately 3 metres in depth and a gently sloping wall of about 22 metres.

Chachahuate

Chachahuate is the sister community of Nueva Armenia, occupying the largest of the cays within the CCMPA (15° 56' 40N, 86° 28' 43 W). There are approximately 43 households with a maximum population of 200 during the peak fishing season, and an average resident population of 90 people. The island has reduced in size following Hurricane Mitch in 1998, to about 50 metres in length because it is exposed to the prevailing north easterly trade winds. There is some tourism development with US AID and WWF sponsorship of a restaurant and cabanas, and the island is advertised in the national tourism guide 'Honduras Tips'. Commercial sail boats, located in Roatan and Utila advertising day long trips to the Cayos Cochinos islands, use Chachahuate as the island stop off. The reef flat is approximately 1metre deep, sloping gently to a 20 metre wall and sandy bottom.

Part of the recent development of the island was the construction of two environmentally friendly toilets in 2007 to replace the long drop toilets that had been used in the past and were resulting in localised organic pollution. These were not being used as successfully as desired in 2007, however, their use has now become common.

Rio Esteban

Rio Esteban is a small Garifuna settlement furthest from the main city of La Ceiba on the north coast of the sites studied, approximately 12 nautical miles from the Cayos Cochinos. The community has an estimated 630 inhabitants divided into several neighbourhoods (barrios), with one school up to secondary level. The coastal region is an area of estuarine discharge from the River Aguan, mangrove forests and mud flats creating an environment of relatively high deposition with a natural spit. The community has relatively little tourism with four small hotels, five restaurants (including a newly developed beachfront restaurant), and cabanas on the beachfront that are now abandoned as a result of Hurricane Mitch in 1998. It is the least accessible of all the study sites, requiring an off-road vehicle to navigate through a riverbed during the dry season. During the wet season this river bed is prone to flooding and prevents any access to or from the community. The community does have a regular daily bus service to Jutiapa and Trujillo; however, these buses cannot pass through the river when in flood.

In 2008 Operation Wallacea started using Rio Esteban as the entrance point to the CCMPA. This involved volunteers being accommodated in local houses for one night. A cultural event, food and transportation to the CCMPA on fishing boats were also arranged through the village.

3: Fish and Invertebrate Ecology Research Group

Introduction to the group

Fish and invertebrate populations provide an essential source of income for local communities whilst performing vitally important roles in the ecological functioning of a healthy reef system. The reefs around the Cayos Cochinos have huge ecological and economic importance to the area. Assessing the health of the fish and invertebrate populations on these reefs is, therefore, of considerable importance, as is developing an understanding of their ecology, if these resources are to be managed sustainably. By conducting consistent, annual monitoring, Operation Wallacea researchers hope to quantify changes in the reef community over time, and in response to anthropogenic or other environmental factors.

The group focuses on species of fish and invertebrates that are commercially important to the systems, or that can be used as indicators of overall reef health. In 2008 the group conducted basic surveys of the fish populations on 4 reefs around Cayos Cochinos. The monitoring programme was subsequently expanded and 10 reefs were surveyed for fish, coral, invertebrate and benthic structure in both 2009 and 2010. Over the 2011 field season, 8 reef sites were successfully monitored using Underwater Visual Census (UVC) protocols. Unfortunately, it was not possible to survey all 10 reefs that had been monitored in 2009 and 2010 due to the field season being shortened from 8 to 6 weeks. Also, the introduction of additional, complementary survey protocols, in the form of Stereo Video Surveys (SVS), during the 2011 season meant that the time dedicated to UVC transects was reduced (SVS is discussed in detail in section 5).

Methods

Underwater Visual Census (UVC)

Over the 2011 Operation Wallacea field season, 8 reefs were successfully monitored using UVC protocols (Table 3.1). By sending up to 3 teams of 3-4 divers (Operation Wallacea university volunteers and staff) at any one time to the study sites, it was possible to conduct 8 replicate transects at each site at depths of both 8m and 12m at all but two of the study sites. The reefs at the Timon and Peli 3 sites were too shallow to complete transects at a depth of 12m and, thus, transects were only conducted at a depth of 8m for these two sites.

Table 3.1 Record of sites for which 8 replicate UVC transects were completed by Operation Wallacea volunteers over the 6 week field season on Cayo Menor in 2011.

	Arena	El Avion	Jenna's cove	Peli 0	Peli 2	Peli 3	Peli 4	Timon
8m	✓	✓	✓	✓	✓	✓	✓	✓
12m	✓	✓	✓	✓	✓	-	✓	-

On each transect, volunteers recorded fish, invertebrate and benthic data. The transect tape was laid by the transect supervisor, who would first locate a suitable point to anchor the start of transect tape on the reef. Because a specific (non-random) point was being selected for the start of the transect, data collection did not begin until 5m from the start point. The fish surveyor would then conduct a 20 m belt transect, identifying and counting all species (Appendix 1) 2.5 m either side and

5 m above of the transect line, equating to a surveyed volume of 500 m^3. Invertebrates (Appendix 2) were recorded by the invertebrate surveyor 1 m either side of the 20 m transect line, equating to a surveyed area of 100 m^2; and finally the benthic surveyor would record the substrate (Appendix 3) beneath the tape at 50 cm intervals along the 20 m transect line resulting in 40 different measurements from which a percentage cover could be obtained. All data were entered into the Operation Wallacea Microsoft Access Reef Check database, a copy of which was left with HCRF science staff at the end of the season.

Analysis

Diversity (measured through the Shannon diversity index), species number and individual numbers of fish were analysed through univariate statistics, as were populations of individual species of fish. Percentage cover of hard corals and algae is compared between sites and between years.

Results

General patterns of reef fish abundance and diversity across the Cayos Cochinos as observed by UVC

Table 3.2 displays the mean number of fish and mean number of species observed per transect and the Shannon Diversity score for each sampled year between 2008–2011.

Table 3.2 Mean (±1SE) number of species (per transect), individuals (per transect), and Shannon diversity of reef fish populations on the reefs of the Cayos Cochinos determined using UVC.

	2008	2009	2010	2011
Mean no. of individuals (±1SE)	10.50 (2.26)	75.50 (4.93)	67.99 (10.19)	69.69 (6.46)
Mean no. of species (±1SE)	4.29 (0.68)	11.33 (0.62)	12.22 (0.52)	15.08 (0.70)
Shannon Diversity (±1SE)	1.20 (0.17)	1.77 (0.08)	3.19 (0.04)	2.97 (0.01)

Results for the 2008 sampling period were lower than for subsequent years, however, a different sampling protocol was used during the 2008 season and so 2008 data were excluded from subsequent statistical analyses.

Data for the number of fish observed per UVC transect were found to violate the assumption of homogeneity of variance ($p<0.001$) and a Log_{10} transformation of the data was unable to resolve this issue. A Kruskal-Wallis test was performed and resulted in a significant difference in the median number of fish observed per transect between years ($\chi^2 = 14.1$, df = 2, $p = 0.01$). Post-hoc Mann-Whitney U tests performed between pairs of years showed there to be a significant difference in the median number of fish observed per transect between 2009 and 2010 ($p < 0.001$) and between 2009 and 2011 ($p = 0.018$), however, there was no significant difference between 2010 and 2011 ($p = 0.411$).

Data for the number of species observed per UVC transect were also found to violate the assumption of homogeneity of variance ($p<0.001$) and a Log_{10} transformation of the data was unable to resolve this issue. A Kruskal-Wallis test was performed and resulted in a significant difference in the median number of species observed per transect between years ($\chi^2 = 18.37$, df = 2, $p < 0.001$). Post-hoc Mann-Whitney U tests performed between pairs of years showed there to be no significant difference in the median number of species observed per transect between 2009 and 2010 (p =

0.572) but there was a significant increase in the number of species observed per transect from 2009 to 2011 ($p < 0.001$) and from 2010 to 2011 ($p = 0.002$).

Reef fish abundance and species diversity across individual reefs surveyed between 2008-2011

Fish abundance (total number of fish observed) was found to vary between years and also between sites within each year, whereas fish diversity (number of species recorded) was found to be fairly consistent between years and across sites (Figure 3.1).

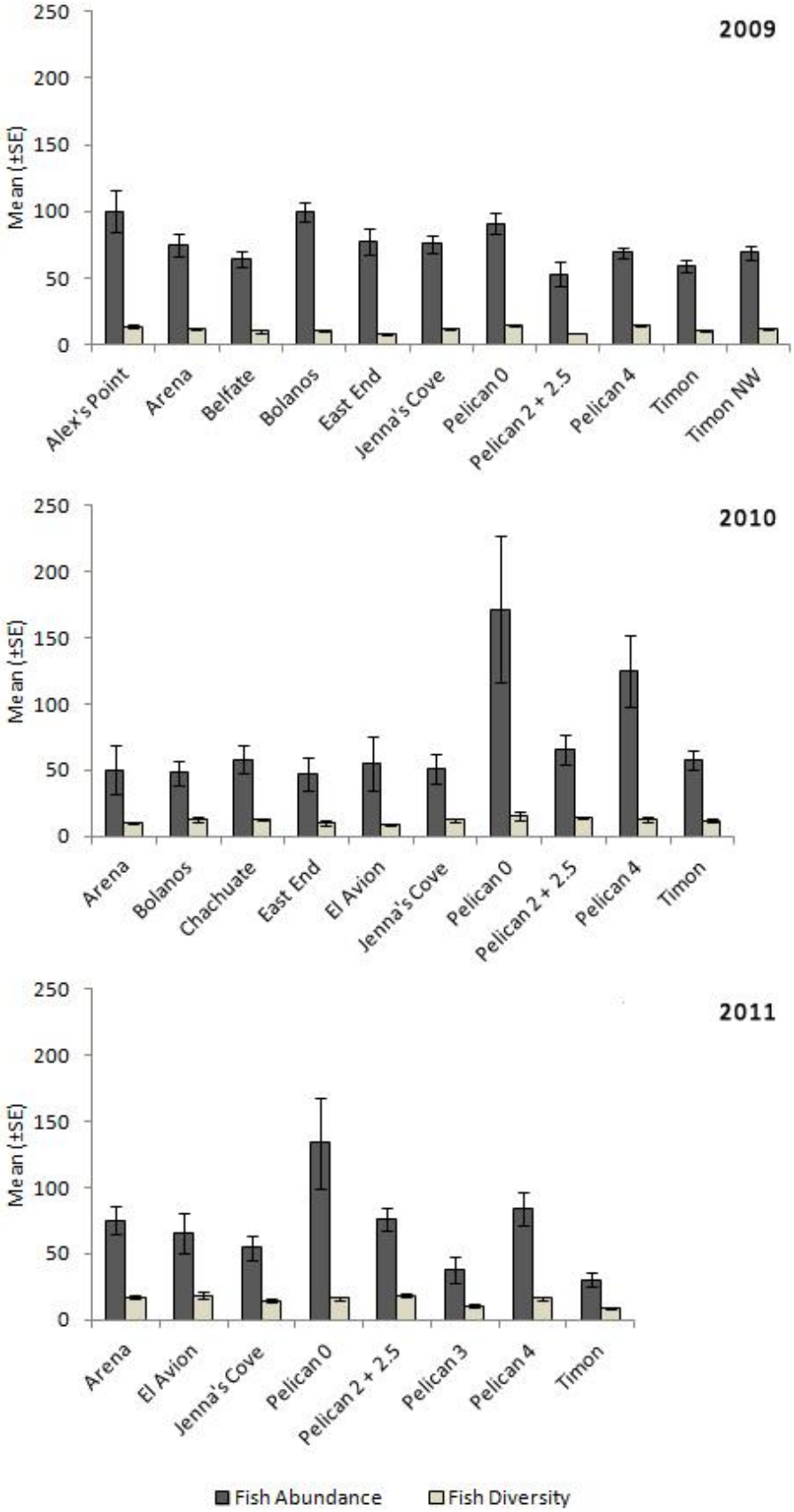

Figure 3.1 Mean fish abundance and diversity per transect at surveyed reef sites between 2009 and 2011. Abundance calculated as total number of fish observed; Diversity calculated as total number of species observed.

A comparison of reef fish abundance across sites that were surveyed in al three years 2009-2011 is given in Figure 3.2. Diversity appears to have, on average, remained reasonably consistent between years. High abundance values for Pelican 0 and Pelican 4 in 2010 are likely the result of outliers in the data, as indicated by the large SE values. However a decline in fish abundance is apparent in 2011 for Timon.

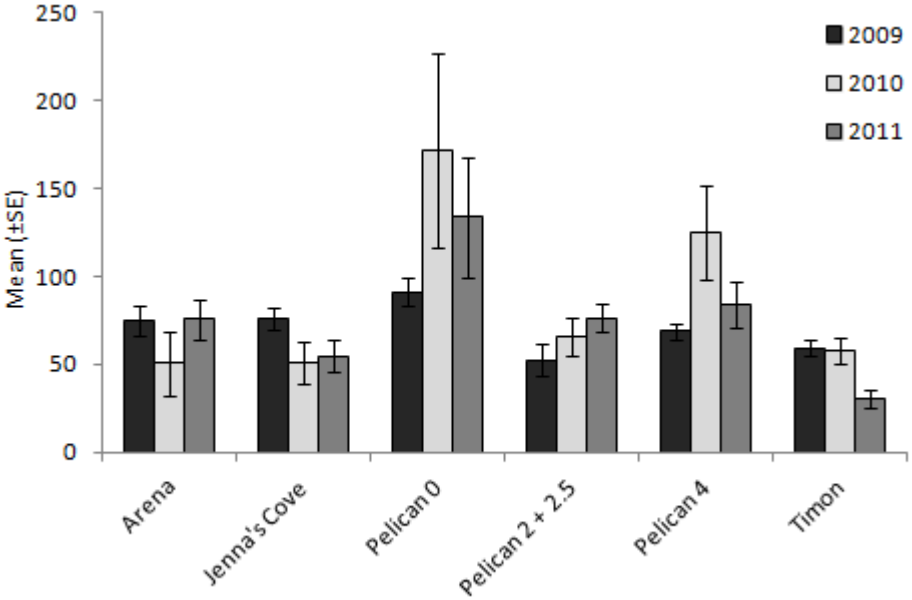

Figure 3.2 Comparison of mean fish a abundance at reef sites surveyed in all three years between 2009 and 2011

A comparison of fish diversity across sites that were surveyed in al three years 2009-2011 is given in Figure 3.3. Arena, Jenna's Cove and Pelican 2+2.5 all appear to show an increase in reef fish diversity between 2009 and 2011. However, no obvious increase in diversity is apparent for Pelican 0 or Pelican 4 and fish diversity at Timon appears to have decreased in 2011.

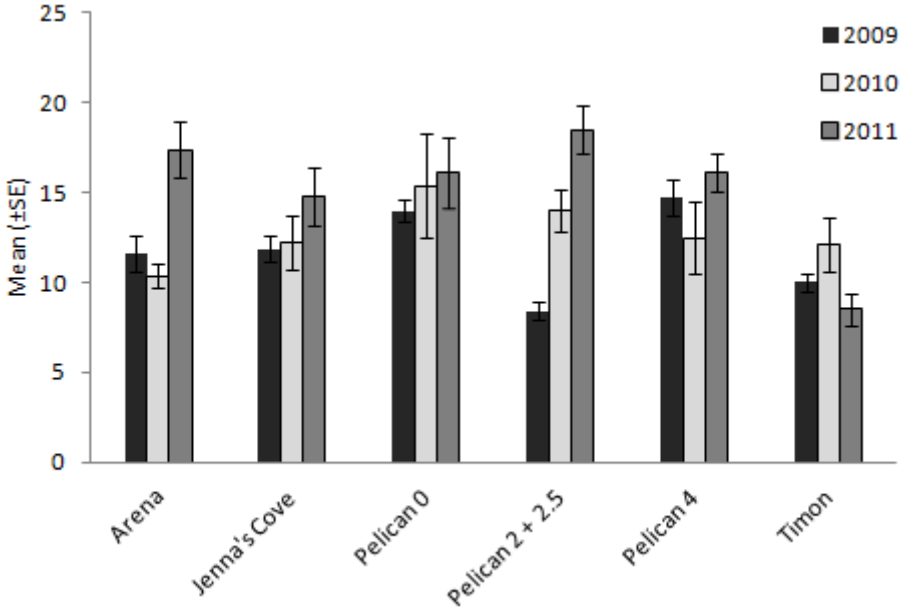

Figure 3.3 Comparison of mean fish diversity at reef sites surveyed in all three years between 2009 and 2011

Abundance of commercially valuable fish species: groupers, snappers and grunts

A large amount of variation was observed in the data for the commercially significant species (grouper, snapper and grunt) between sites and between years, however, encounter rates were relatively low across all sites. Data on the mean number of individual fish observed per transect are displayed in figure 3.4. In general, the numbers of groupers were lower than the numbers of either snappers or grunts. Also, there appears to be a decline in the number of groupers in 2011 across all sites.

The trend in commercial fish species populations over time is better illustrated by comparing those sites that were surveyed in all three years (2009-2011). Figures 3.5, 3.6 and 3.7 display the numbers of groupers, snappers and grunts respectively across all sites surveyed in all three years. It can be seen in Figure 3.5 that grouper populations appear to be showing worrying declines across all sites. Population trends for snappers and grunts are less pronounced than for groupers. Snapper populations appear to show neither a significant decline nor growth within the survey period (Figure 3.6). Grunt populations appear to show declines at Pelican 0 and Pelican 4, but a growth in population size at Timon (Figure 3.7).

Abundance of important herbivorous fish species: parrotfish and surgeonfish

A large decline in the abundance of parrotfish was observed between 2009 and 2010, with parrotfish abundance remaining at low levels in 2011 (Figure 3.8). By comparing parrotfish abundance across those sites that were surveyed in all three years (Figure 3.9), it can be seen that abundance fell sharply between 2009 to 2010 and then continued to fall in 2011 in all but one site, Jennas Cove, however, abundance had already declined more severely at Jenna' Cove between 2009 and 2010 than at any of the other sites.

Abundance of surgeonfish was consistently low across all sites in all years (Figure 3.8) and no obvious trend in decline was observed between years when comparing across sites that had been surveyed in all three years (Figure 3.10). Higher abundance in 2010 is also associated with very large SE and so is better attributed to high variability in the data set rather than a temporary increase in abundance in that year. Overall, abundance of surgeonfish was low across all sites in all years.

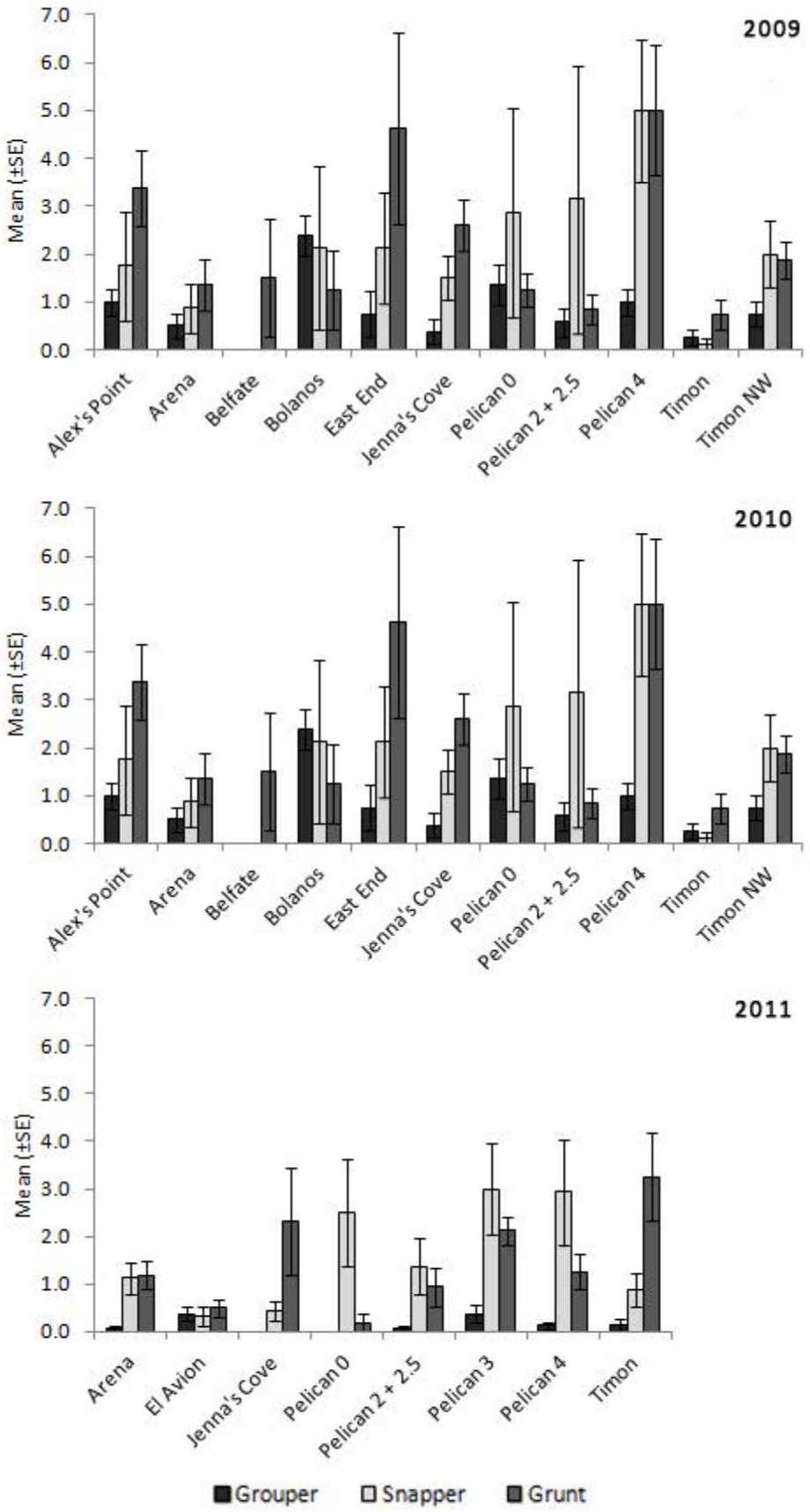

Figure 3.4 Mean number of individuals of commercially important fish species (Groupers, Snappers and Grunts) observed per transect at reef sites surveyed between 2009 and 2011

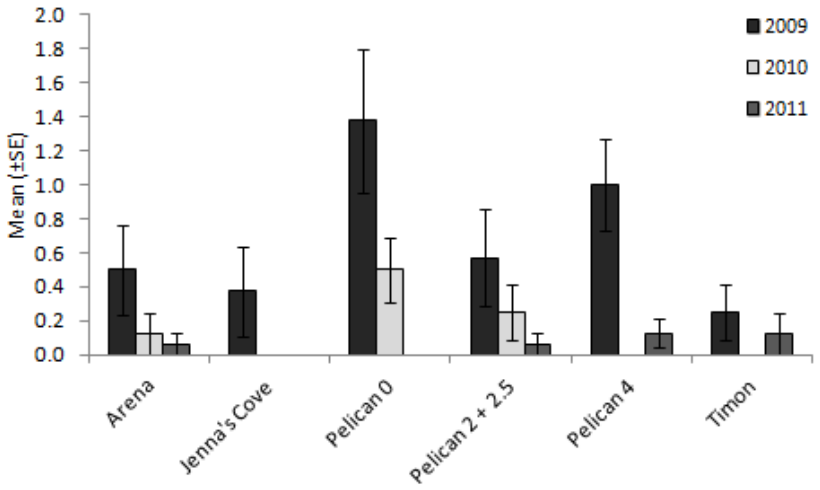

Figure 3.5 Comparison of mean number of groupers observed per transect on reef sites surveyed in all three years between 2009 and 2011

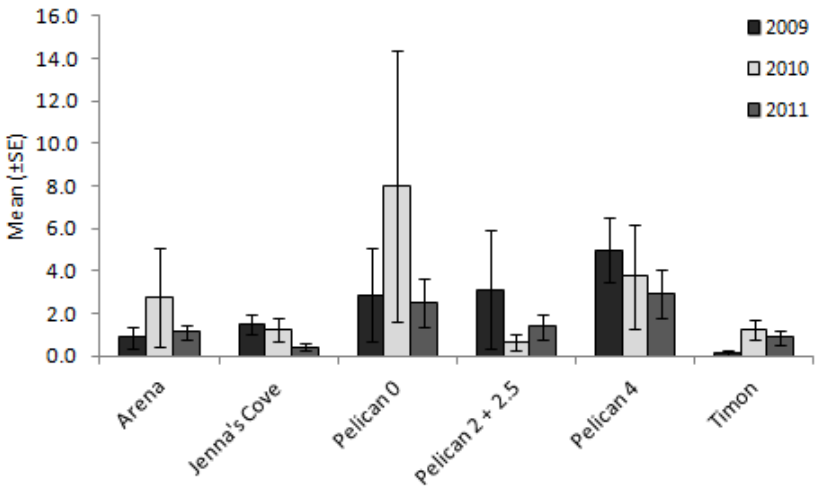

Figure 3.6 Comparison of mean number of snappers observed per transect on reef sites surveyed in all three years between 2009 and 2011

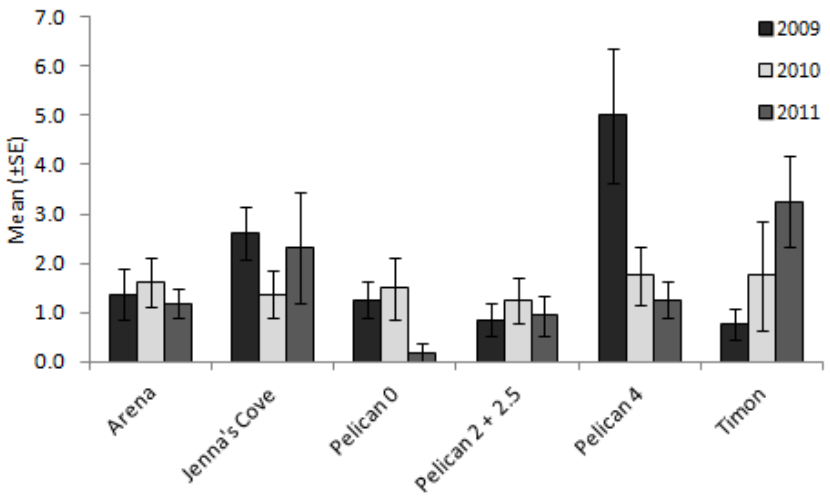

Figure 3.7 Comparison of mean number of grunts observed per transect on reef sites surveyed in all three years between 2009 and 2011

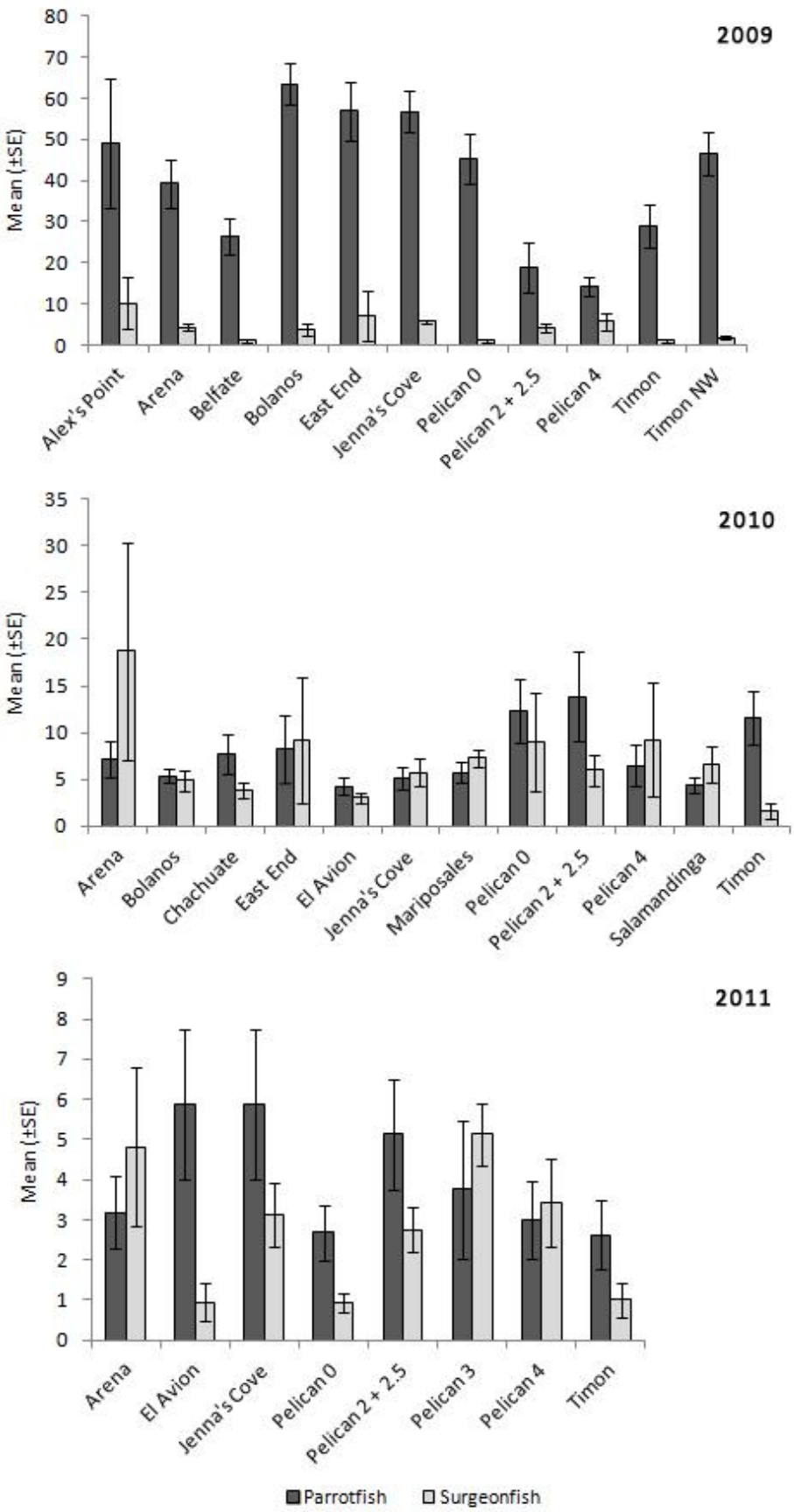

Figure 3.8 Mean number of Parrotfish and Surgeonfish observed per transect at reef sites surveyed between 2009 and 2011. (Note the difference in the scale on the dependent axis between years)

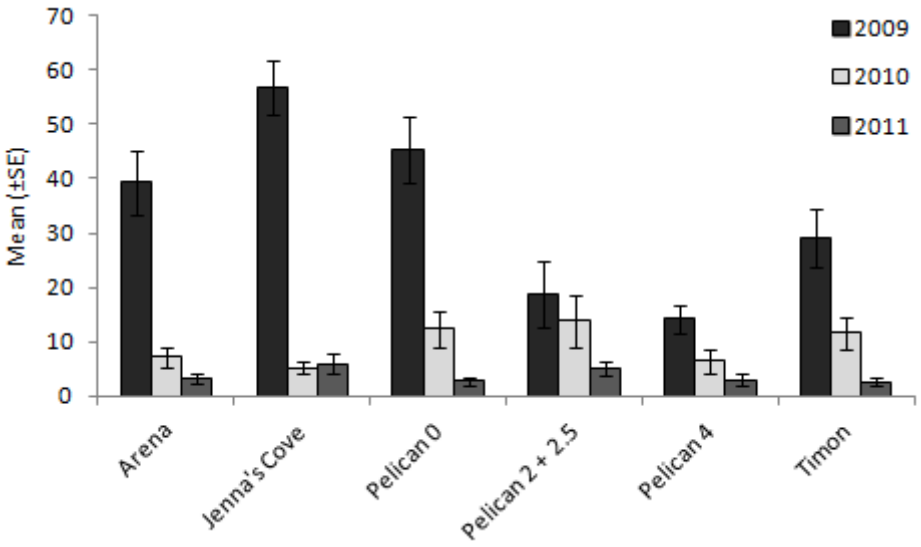

Figure 3.9 Comparison of mean number of Parrotfish observed per transect on reef sites surveyed in all three years between 2009 and 2011

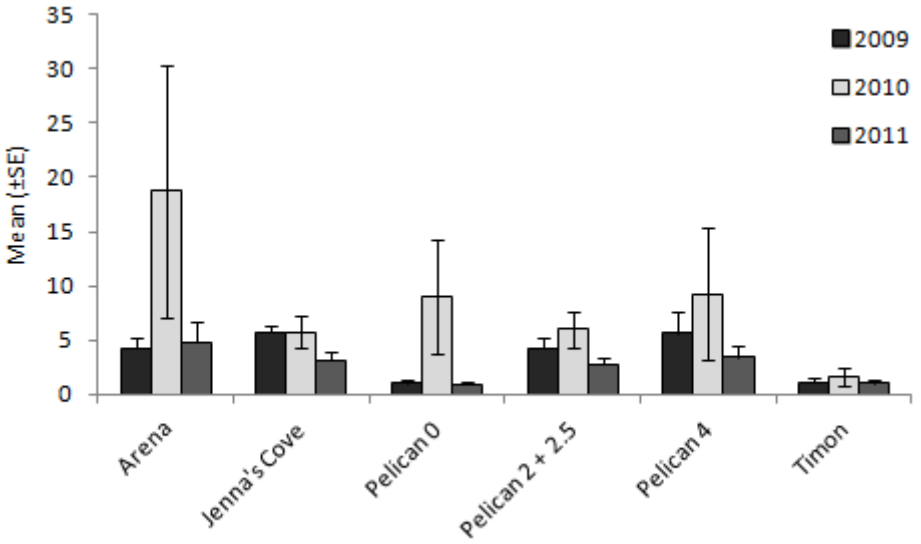

Figure 3.10 Comparison of mean number of Surgeonfish observed per transect on reef sites surveyed in all three years between 2009 and 2011

Percentage cover of hard corals and algae

The mean percentage cover (± SE) of hard coral and algae determined from point intercept line transects data is displayed in Table 3.3.

Percentage coral cover data were found to violate the assumption of homogeneity of variance ($p<0.001$) and a Log_{10} transformation of the data was unable to resolve this issue. A Kruskal-Wallis test was performed and resulted in a significant difference in the median percentage coral cover observed between years ($\chi^2 = 20.2$, df = 2, $p < 0.001$). Post-hoc Mann-Whitney U tests performed between pairs of years determined that percentage coral cover was significantly higher at the sites surveyed in 2010 than in either 2009 ($p = 0.001$) or 2011 ($p < 0.001$), but that there was no significant difference in the percentage coral cover between 2009 and 2011 ($p = 0.249$).

Percentage algal cover data were found to violate the assumption of homogeneity of variance ($p<0.001$) and a Log_{10} transformation of the data was unable to resolve this issue. A Kruskal-Wallis test was performed and resulted in a significant difference in the median percentage algal cover observed between years ($\chi^2 = 16.4$, df = 2, $p < 0.001$). Post-hoc Mann-Whitney U tests performed between pairs of years determined that there was no significant difference in percentage algal cover between 2009 and 2010 ($p = 0.376$), but percentage algal cover was significantly higher in 2011 than in both 2009 ($p = 0.004$) and 2010 ($p < 0.001$).

Table 3.3 Mean percentage cover of hard coral and algae across all survey sites, as determined through point intercept line transects.

	Year		
	2009	2010	2011
Mean % Coral Cover (± SE)	20.43 (1.06)	29.09 (1.79)	19.12 (1.11)
Mean % Algal Cover (± SE)	53.54 (1.77)	50.56 (1.87)	61.35 (1.74)

Mean percentage coral cover varied between sites (Figure 3.11) and, thus, the apparent difference in percentage cover between years is likely to be, in part, due to different sites being sampled in different years. However, a comparison of mean percentage coral cover at only the sites surveyed in all three years also revealed this pattern of higher coral cover during the 2010 sampling period (Figure 3.12).

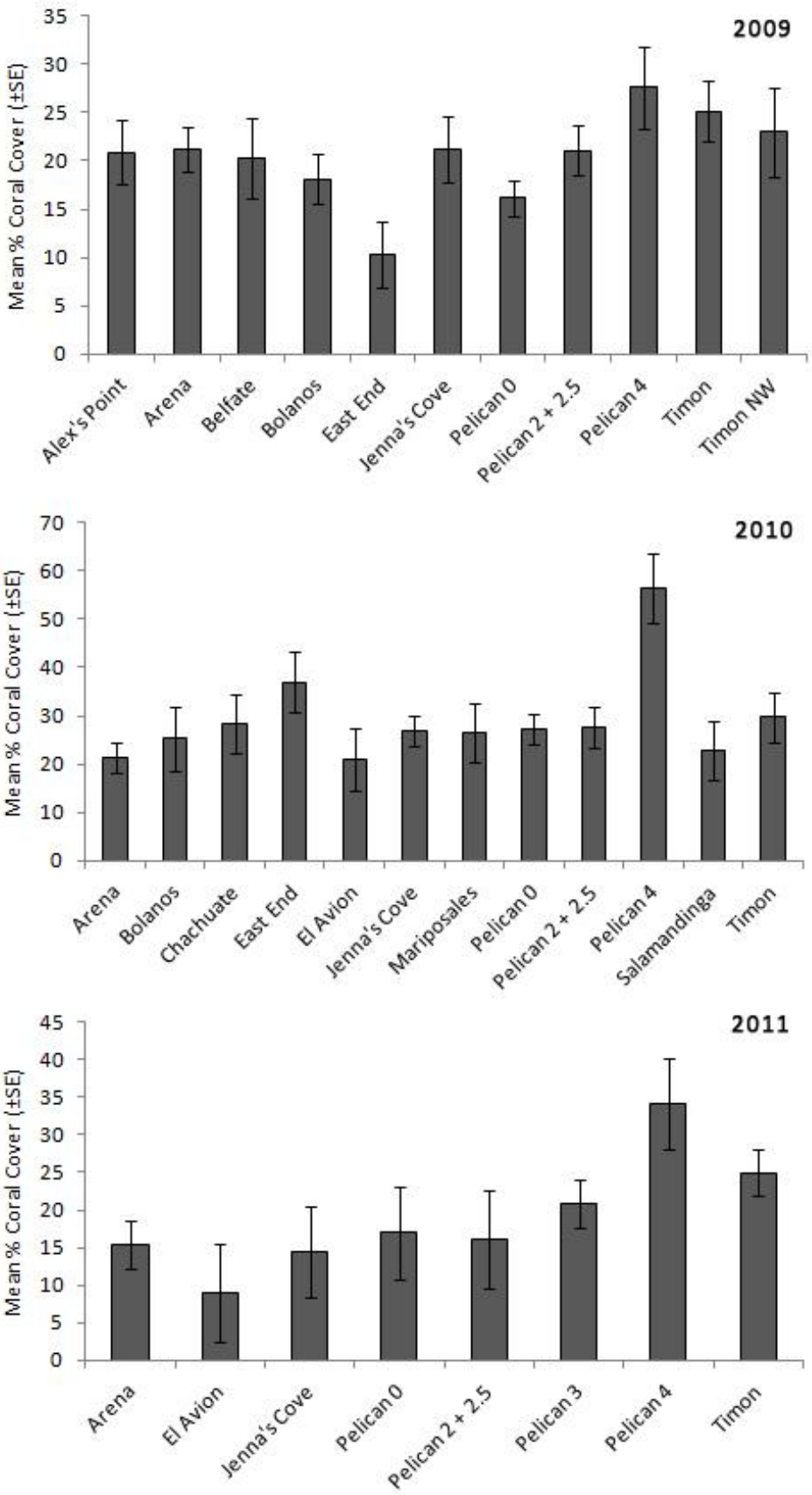

Figure 3.11 Mean percentage coral cover per transect at reef sites surveyed between 2009 and 2011

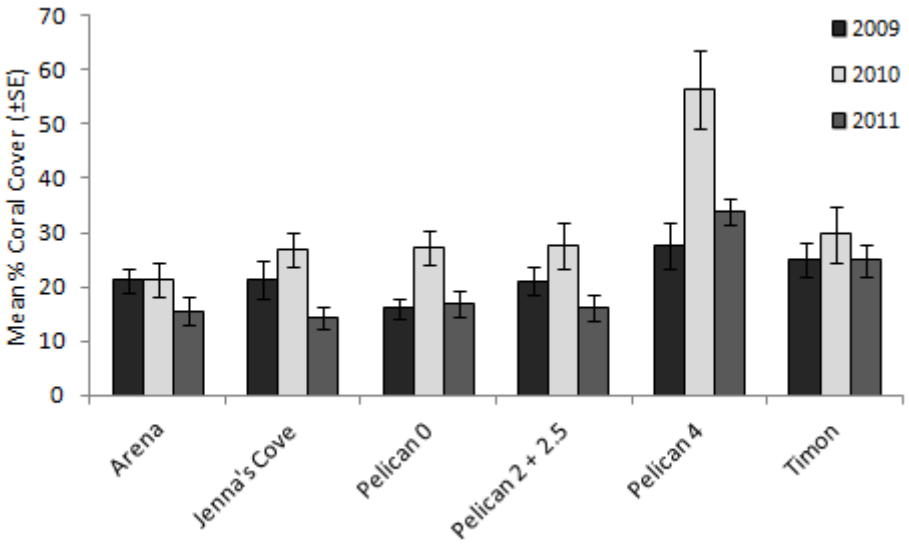

Figure 3.12 Comparison of mean percentage coral cover per transect on reef sites surveyed in all three years between 2009 and 2011

Mean percentage algal cover varied between sites (Figure 3.13). Although overall mean percentage algal cover was found to be higher in 2011 than in the two previous years, a comparison of sites surveyed in all three years revealed this pattern to be true at some sites (e.g. Pelican 0, Pelican2+2.5 and Timon, but not at others (Figure 3.14).

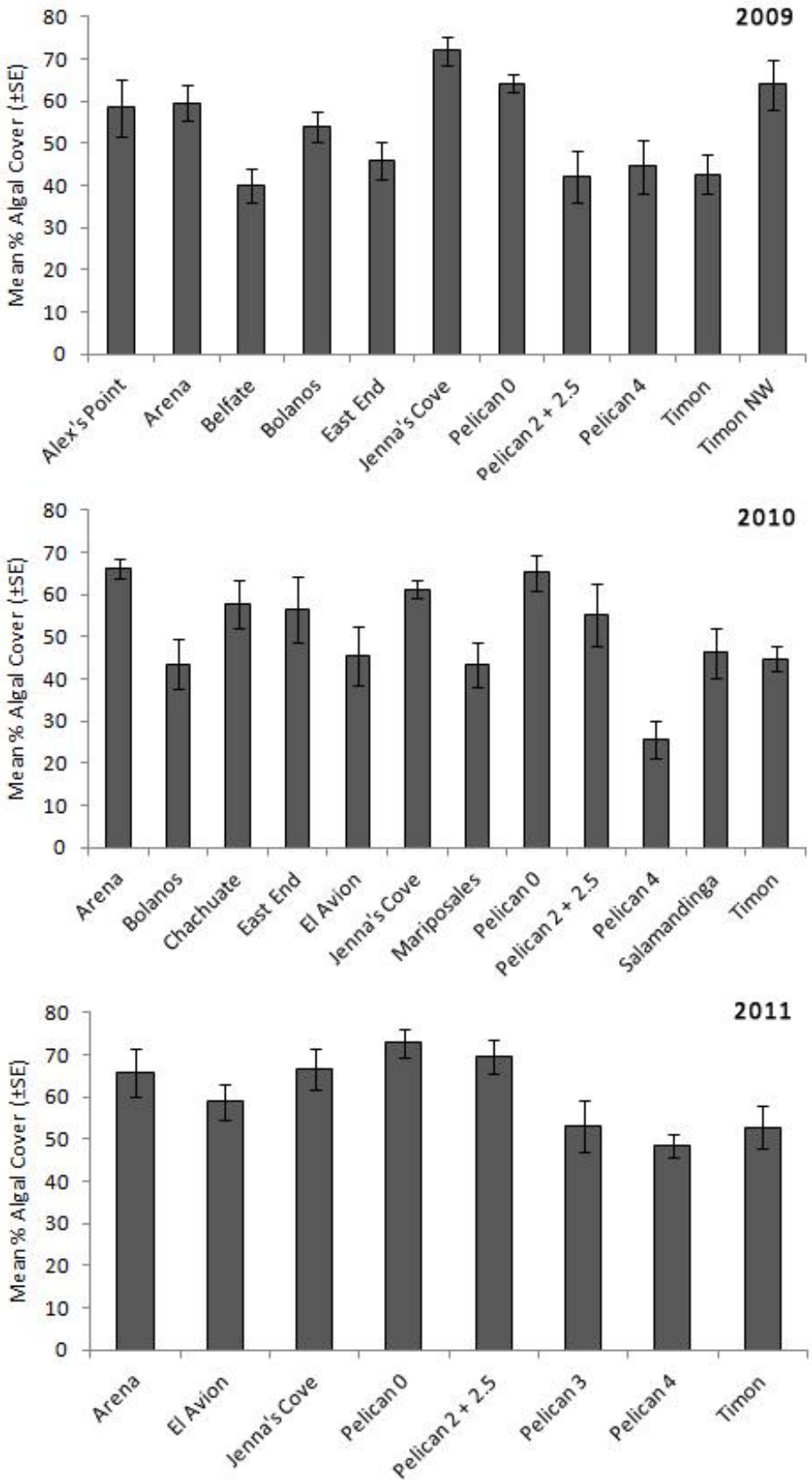

Figure 3.13 Mean percentage algal cover per transect at reef sites surveyed between 2009 and 2011

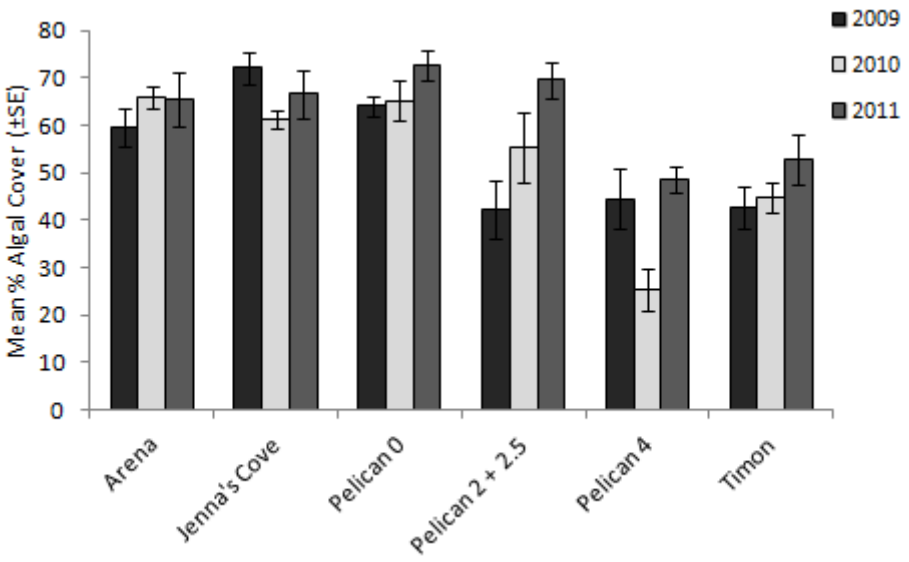

Figure 3.14 Comparison of mean percentage algal cover per transect on reef sites surveyed in all three years between 2009 and 2011

The comparison of algae and hard coral coverage on all the reefs within the Cayos Cochinos is not encoraging. Hard coral coverage is very low while in contrast algal coverage is very high. In 2011, algal coverage was above 30% at all sites. Many reefs have algae coverage 2 to 3 times higher than hard coral coverage. This pattern is suprisingly consistent across all reefs within the MPA.

Conclusions

Low estimates of fish abundance and species diversity in 2008 (as measured as mean number of species per transect and by Shannon Diversity Index) are likely due to a different survey methodology being implemented in that year. Therefore, temporal trends in the dataset should only be investigated using data collected between 2009 – 2011.

A statistically significant drop in the median number of fish encountered per transect was observed between 2009 and 2010 and overall fish abundance did not recover in 2011. This may be indicative of an overall decline of fish abundance on the reefs within the CCMPA and careful attention should be paid to fish abundance estimates generated from future years data in order to identify if this is truly the case. However, by comparing fish abundance on those reefs that were surveyed in all three years (2009-2011), an obvious pattern of declining abundance was not apparent but, instead, appears to vary between sites. It is possible that the apparent decline in overall fish abundance is, at least in part, due to the exclusion of sites in the 2011 survey period that had relatively high fish abundance in previous years. Future surveys should be conducted at the same sites that were visited in 2011 in order to continue to compare trends within individual reefs, but also to improve the utility of the data set for drawing wider conclusions about reef fish populations across the CCMPA.

Encouragingly, the median number of fish species encountered per transect increased between 2009 and 2011. This result may be an indication of a recovering reef system with increasing diversity, however, although Shannon Diversity was higher in 2011 than in 2009 Shannon Diversity declined from 2010 to 2011.

Of the commercially important fish species, Grunts and Snappers were encountered most frequently, but populations appear relatively low and are showing signs of decline at some sites such as Pelican 4. However, for Groupers, the situation is more severe, with extremely low encounter rates and declining population trends across all sites. The situation with Grouper populations must be taken seriously, as in 2011 not a single Grouper was observed at two of the survey sites (Jenna's Cove and Pelican 0).

Analysis of the two major herbivore groups, Parrotfish and Surgeonfish showed that encounter rates were fairly low for both groups. However, Surgeonfish numbers appear to be stable between years, whereas for Parrotfish a worrying declining trend is apparent across all sites. It is unclear why such a rapid decline in parrotfish numbers should be occurring, especially with algal cover being relatively high.

The percentage of coral cover appeared to increase in 2010 but then decline again in 2011 to the base level observed in 2009 of approximately 20%. As it is unlikely that such a fluctuation in coral cover would have taken place over such a short timeframe, this result can most likely be best explained by different dive sites being included in the 2010 survey period. However, even taking into account the influence of including different dive sites in different years, a significant increase in percentage algal cover was observed from 2009 to 2011. Interestingly, this increase in algal cover has occurred during a time where it also appears that parrotfish numbers have declined.

Future directions

Analysis of the UVC data has highlighted some interesting temporal trends within the CCMPA. It is essential that UVC surveys are continued in order to track these trends and to assess the effectiveness of current management practices within the CCMPA. Long-term monitoring is critical, as there will always be some lag phase between implementing management practices and seeing the effects of such actions. This may be particularly true for long-lived species with slow reproductive rates, such as groupers. It is important that future UVC surveys are conducted at the same sites as in 2011 in order to investigate temporal trends in fish populations and the benthic environment.

Publications

Harm JH, Kearns E, Speight MR (submitted) Differences in coral-reef fish assemblages between mangrove-rich and mangrove-poor islands of Honduras Proceedings of the 11th International Coral Reef Symposium, Ft. Lauderdale, Florida, 7-11 July 2008.

Shrives, J.P., Cowie, G.L., Thompson, P.A., Riley, J.S. and Speight, M.R. (2008) Integrating oceanography and marine ecology: What effect does the Río Aguán have upon the benthic reef community of Los Cayos Cochinos, Honduras? Oral Presentation RCUK 2008

Shrives, J.P., Lea, J.S.E. and Speight, M.R. (2008) How Does Black Band Disease Affect The Benthic Ecology of Reefs in Los Cayos Cochinos, Honduras? Poster Presentation RCUK 2008.

Mullier, T.W. and Shrives, J.P. (2008) Ecological distribution, demography and host specificity of cleaner shrimp in the Cayos Cochinos, Honduras. Poster Presentation RCUK 2008.

Shrives, J.P. (2008) Safeguarding the Reefs of Cayos Cochinos, Honduras. ADM, Issue 29

Shrives, J.P., Lea, J.S.E. and Speight, M.R. (in prep) Faunal Associations with Black Band Disease in Cayos Cochinos Honduras. For submission to Coral Reefs

Shrives, J.P., Lea, J.S.E. and Speight, M.R. (in prep) Spatial ecology and succession dynamics of Black Band Disease in Cayos Cochinos Honduras. For submission to Marine Ecology Progress Series or Coral Reefs

4: Stereo Video Surveys (SVS)

Introduction

Traditionally, surveys of reef fish populations, as well as benthic and invertebrate community structure, have been conducted by underwater visual census (UVC) by a team of scuba divers. However, the recent development of stereo video equipment for surveying reef fish communities is allowing large volumes of data to be collected within a single dive and analysed in detail back in the laboratory. An added advantage of stereo video surveys (SVS) over UVC, or even surveys that use a single video camera, is that SVS allows the researcher to accurately measure the size of the fish observed on the transect. The ability to accurately assess fish size makes it possible to estimate and compare biomass of reef fish populations. Although fish size has often been approximated during UVC, it has been shown that these approximations introduce a large degree of error and, thus, it is very difficult to make reliable comparisons of fish biomass between areas using UVC. Thus, SVS provides a sophisticated and novel approach to reef fish surveys that is allowing the first accurate assessment of the biomass of fish populations in Honduran reef systems.

However, the merits of SVS over UVC, like any new survey methodology, must be compared objectively. Therefore, SVS was conducted in addition to traditional UVC surveys in order to compare the results of the two survey methods and also to be able to make direct comparisons to previous year's data where only UVC was conducted.

Methods

This was the first time that SVS had been conducted by Operation Wallacea within the CCMPA. SVS were conducted as outlined in Watson et al. (2010). The equipment consisted of two Canon HD cameras, model VIXIA HFS21, in waterproof housings and mounted on an aluminium bar (Figure 4.1). A diode extends centrally in front of the cameras, which are angled slightly inwards towards the diode, and is used to synchronize the video footage during analysis. Before any surveys were conducted the cameras were calibrated in water using footage of the calibration cube at different orientations, as outlined in the manufacturer's manual. Calibration of the stereo-video cameras is essential to obtain and maintain accurate length measurements of the fish from the stereo-video footage.

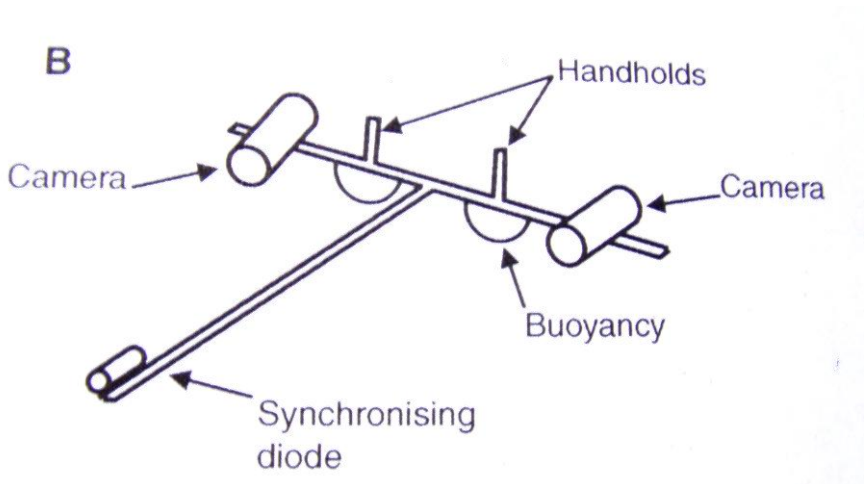

Figure 4.1 The stereo-video apparatus uses two cameras mounted on either side of an aluminium bar with a central diode extending in front of the cameras used to synchronize the video footage during analysis.

The SVS monitoring team consisted of three SCUBA divers: one operating the SVS system, one measuring the length of the transects, using a tape reel, and one to provide additional support to help the team operate safely. The SVS operator swam slowly along each transect with the cameras angled slightly downwards toward the reef, as instructed in the SVS system user manual. Six 25 m transects, separated by 10m gaps, were conducted at depths of both 8 m and 12 m across five sites (El Avion, Jenna's Cove, Peli 0, Peli 2+2.5 and Peli 4).

Analysis

Stereo-video footage was converted from MTS to AVI format using MTS converter and then analyzed using *EventMeasure*. Footage from left and right cameras was synchronized in EventMeasure using the synchronizing diode. Fish were only recorded if they were within the 5 m width of the transect, i.e. 2.5 m to the left and 2.5 m to the right of the cameras. Individual fish were identified by family, genus, and species. Length measurements, snout to base of tail, were computed using *EventMeasure*. A copy of the measurement summaries and the .avi files from both the left and right cameras for each set of transects at each site was left with HCRF science staff at the end of the season.

The current study aimed to compare UVC and SVS on four environmental variables: species richness, species abundance, mean fish lengths, and species diversity, although only the first three are discussed within this preliminary report. Species richness, species abundance, and mean fish length estimates were compared between methods across 3 of the 5 survey sites using two-sample t-tests. Two of the survey sites were excluded from analysis as the video footage was in the process of being analyzed at the time of writing.

Results

Species richness

A two-sample t-test was used to test for a significant difference in the number of species found by each method across 3 of the 5 survey sites. (The image analysis for two of the five survey sites were still being analyzed when this report was written). The t-test found a significant difference in the number of species found between the two methods, showing SVS produced significantly higher species counts than UVC ($p=0.002$) (Figure 4.2).

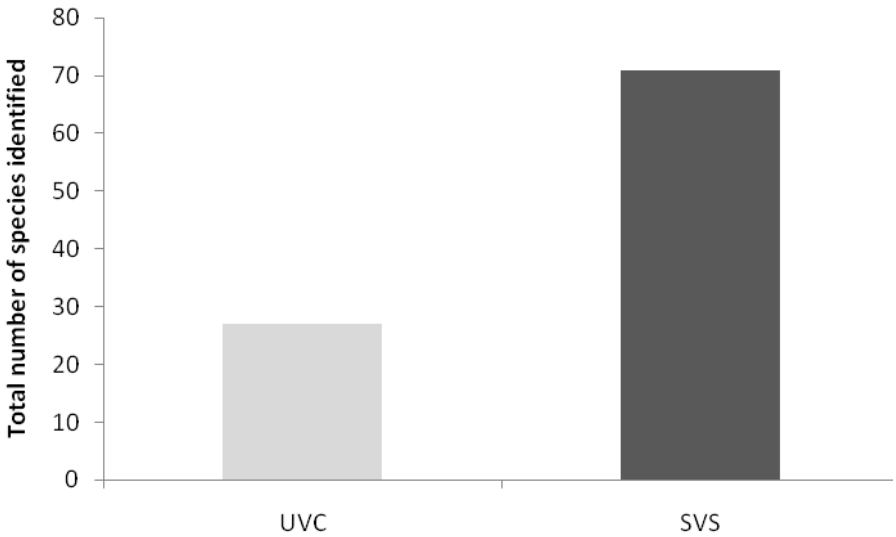

Figure 4.2 Comparison of UVC and SVS on species abundance. SVS produced significantly higher species counts than UVC (p<0.005).

Comparison of mean fish lengths between UVC and SVS

Estimates of mean fish lengths produced by UVC and SVS were compared using a two-sample t-test for 3 of the survey sites. The test showed no significant difference between the methods in mean fish length estimates (p>0.05) (Figure 4.3).

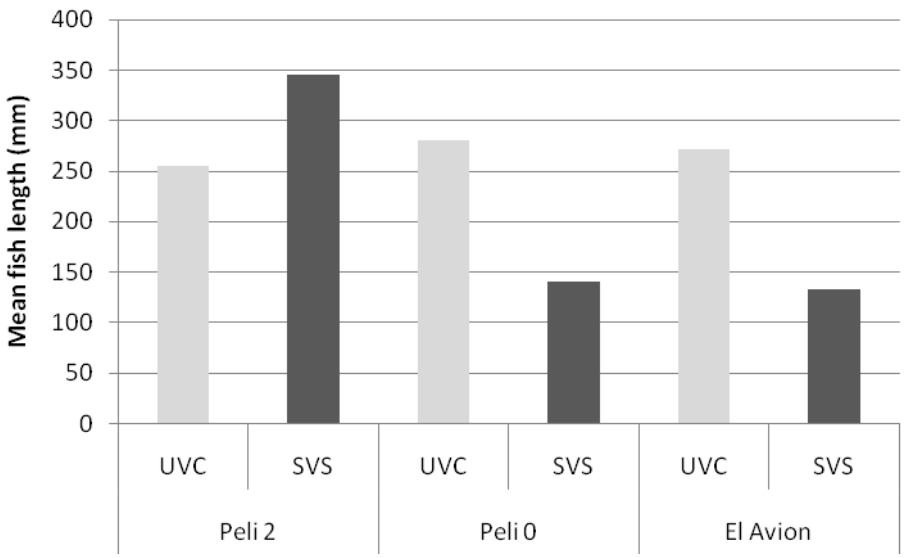

Figure 4.3 Mean fish lengths (mm) were compared between UVC and SVS for 3 survey sites. No significant difference was found between the methods (p>0.05).

Species Abundance

A chi-square test was used to compare abundance estimates produced by UVC and SVS on 8 species of fish (those identified by both methods). Significant differences were found between two species, *Haemulon plumierii* (White Grunt) and *Sparisoma viride* (Stoplight Parrotfish) (Figure 4.4).

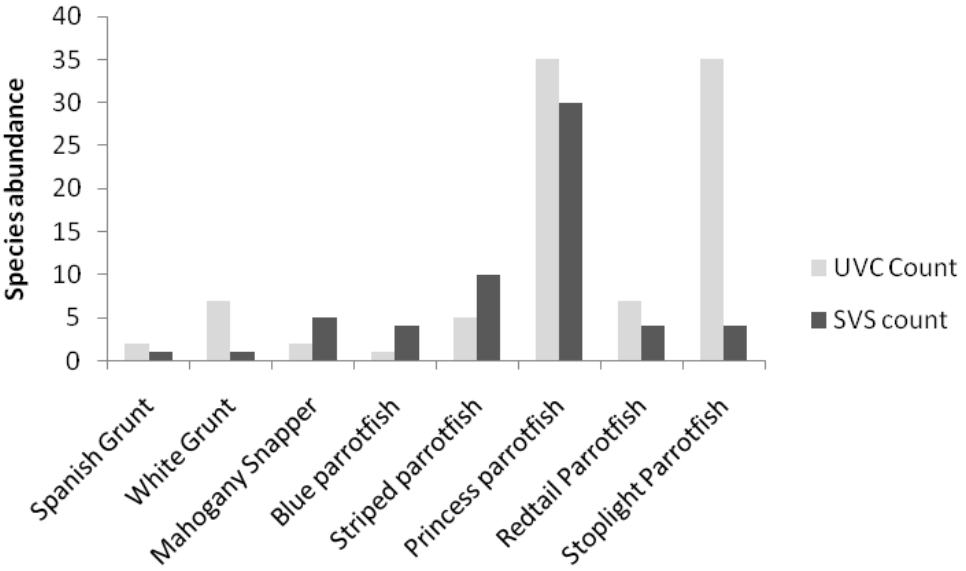

Figure 4.4 Significant differences in abundance estimates were found for 2 of the 8 species compared, White grunt (*Haemulon plumierii*) and Stoplight parrotfish (*Sparisoma viride*) ($p<0.05$, $p<0.05$).

Discussion

This study aimed to directly compare UVC and SVS methodologies for reef fish monitoring. Interestingly, SVS produced higher estimates of species richness over UVC. This may be due to the increased ability of the observer to accurately identify species captured on the video footage, aided by the use of species identification books and increased processing time.

UVC and SVS gave similar estimates for abundance of species ($p<0.05$), with the exception of White Grunts and Stoplight parrotfish. For these two species, UVC gave much higher estimates of abundance than did SVS. Such differences can most likely be attributed to misidentification of those species during UVC surveys and similar species also being 'lumped' into that species category. The misidentification of species during UVC would also explain the lower estimates of species richness produced by this method.

It was predicted that SVS would provide lower estimates of fish length than UVC due to expected overestimation of fish size during UVC. However, preliminary analysis of the data was inconclusive. For two of the three sites (Peli 0 and El Avion) UVC did produce significantly higher estimates of fish length, but for one site (Peli 2) UVC produced significantly lower estimates of fish length than SVS. When all three sites were combined there was no significant difference in fish length between the two methods ($p>0.05$). Further analysis including data from the additional two sites will hopefully help to clarify if a difference does in fact exist between the methods in their ability to estimate fish length.

Conclusions

Comparison of the relative merits of SVS over UVC is a critical in order to evaluate its potential utility as a conservation monitoring tool and this preliminary study has begun to answer some valuable questions. However, due to the vast amounts of data obtained via SVS over the field season and the relatively long time required for analysis, it would be prudent to delay making solid conclusions until the full data set has been analysed. Even so, preliminary analysis of the data has identified some

clear disparities between the two methods, namely that abundance estimates for certain species, that are easily misidentified, can be seriously inflated under UVC. SVS appears to provide a more reliable and systematic method for conducting abundance estimates on reefs and will, no doubt, help to standardise monitoring protocols in the future, thus, aiding the comparison of temporal and spatial datasets.

References

Watson, D. L., Harvey, E. S., Fitzpatrick, B. M., Langlois, T. J. and Shedrawi, G. (2010) Assessing reef fish assemblage structure: how do different stereo-video techniques compare? *Marine Biology* 157:6 page 1237-1250

5: Sea Urchin Ecology Research Group

Project introduction and rational

Sea urchins hold a very highly important role within the ecology of coral reefs, acting as one of the main grazers of algae and ensuring that a low algal density is maintained on the reefs and corals dominate. Without such grazing the potential is for algae species to grow unregulated and smother and eventually kill corals, leading to a phase shift from coral dominated to algal dominated reefs.

There are four dominant species of urchins on Caribbean reefs: the long spined, reef, rock boring and pencil. Of these, the long spined urchin (*Diademia antillarum*) was the largest and most significant grazer. However, in 1983 an as yet unidentified disease decimated the populations of the long spined urchin throughout the Caribbean. Mortality rates were between 95-99%, with 99.9% recorded in some locations. Urchins can live for 100+ years and so recovery of the populations has been slow and indeed absent in some areas. The consequences of this mass mortality (the largest ever recorded in the marine environment) are varied. The majority of reefs have maintained coral dominance, due in part to increased grazing by other urchins and herbivorous fish species compensating for the loss in the long spined urchin. However, many reefs have undergone a phase shift and are now dominated by algae with severe ecological and economic consequences. The main example of this being the Jamaican coral reefs that are now nearly all algal dominated and as a result both the dive tourist and fishing industries have been heavily impacted.

The reefs around the Cayos Cochinos have so far maintained coral dominance, but long spined urchin populations remain low and, therefore, the reefs remain highly vulnerable to a future phase shift to algal dominance. This program aims to assess if and how urchin populations are changing, determine the rate of increase of the long spined urchin population and potentially offer a warning sign if populations of any urchin species start to drop.

Methods

Urchin Surveys

Reefs were surveyed around the CCMPA between 2009 and 2011 for the four most common species of urchin. Reefs were surveyed by snorkelling at depths between 1-2m and counting all urchins within a 15m x 2m transect ($30m^2$). Various sites were surveyed in each year, however, here we focus on those sites that were surveyed in all three years of the study period (El Avion, Jenna's Cove, Pelican 4 and Timon).

Results

Full results of the urchin surveys conducted at all of the sites are provided in appendix 4 of this report. Figure 5.1 displays the mean number of each urchin species that were observed per transect for each of the sites surveyed in all three years (2009-2011). Numbers of the ecologically important long spined urchin were found to be highest at El Avion, but were low at the other three sites.

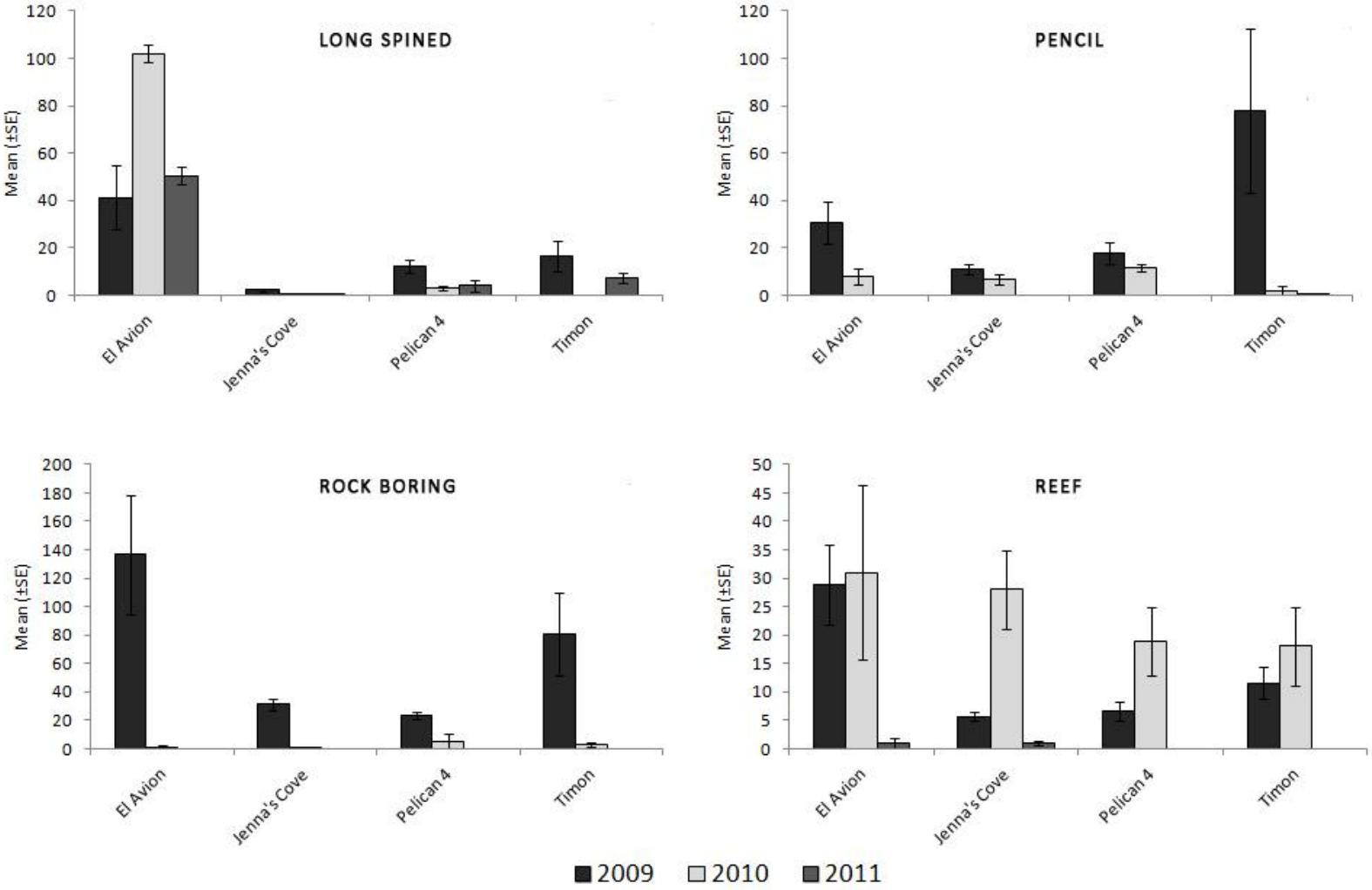

Figure 5.1 Mean number of long spined, pencil, rock boring and reef urchins encountered per 15m transect across reefs surveyed 2009-2011

Worryingly, a significant decline in all four species seems to have occurred over the study period (Figure 5.2) and numbers of all four species were low at sites visited in 2011.

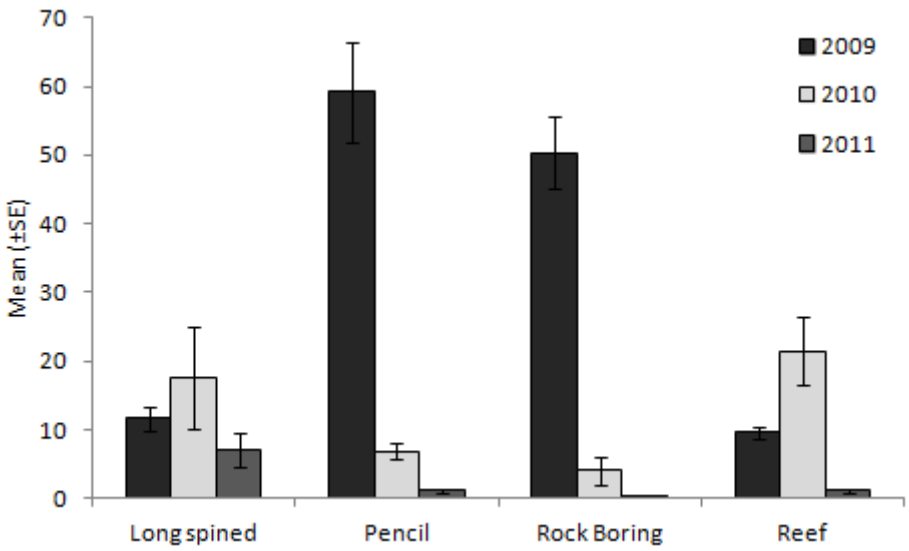

Figure 5.2 Mean number of long spined, pencil, rock boring and reef urchins encountered per 15 m transect (combining data from the 4 sites surveyed in all 3 years) between 2009-2011

Conclusion

A decline in all four of the target species was observed during the study period and most significantly for pencil and rock-boring urchins. If this is a true reflection of the urchin populations within the Cayos Cochinos then it is a serious cause for concern. However, care should be taken when interpreting the results, as only 4 survey sites were visited in all three years and the number of repeat transects conducted was lower in 2010 and 2011 than in 2009.

Future urchin surveys should attempt to focus on a small number of sites (including those that have so far been surveyed in all three years) that can be sampled more frequently during the Operation Wallacea season. An increased effort in annual monitoring at pre-determined sites, rather than the rather ad-hoc approach that has been implemented in previous years will generate a reliable data set from which firm conclusions can be drawn.

Publications

Bologna, P., Webb-Wilson, L., Connelly, P., Saunders, J.E. (2012) A new baseline for *Diadema antillarum*, *Echinometra viridis*, *E. lucunter*, and *Eucidaris tribuloides* populations within the Cayos Cohcinos MPA, Honduras. *Gulf and Caribbean Research* 28

Hall, C. M., Shrives, J.P., Speight, M.R and Saunders, J. (2008) Sea urchin ecology on the shallow reefs of Cayos Cochinos, Honduras, with particular focus on Diadema antillarum recovery. Poster Presentation RCUK 2008.

Shrives, J.P. and Speight, M.R. (in prep) Assessment of the 2007 post-Hurricane Dean stranding of sea urchins upon the beaches of Los Cayos Cochinos, Honduras. For submission to Coral Reefs

6: Conch Ecology and Research Group

Introduction

The Conch Ecology and Research Group is based solely on the Cayos Cochinos and is studying the ecology and populations of the highly important mollusc the Queen Conch *Strombus gigas*. Research is aimed at firstly determining an accurate population estimate for conch within the MPA and developing a monitoring program that allows this to be tracked each year. Secondly the group studies elements of the ecology and life cycles of the conch, with the intention of better understanding the species and, therefore, instigating effective conservation.

The Queen Conch is a large mollusc found throughout the Caribbean. It holds a highly important position with both the ecology and economy of the area. Ecologically the Conch is a deposit feeder, grazing off algae, detritus and epiphytic algae, this is coupled with a large level of sediment disruption and turnover during feeding and movement. This feeding and continued cycling of the sediment prevents it becoming stagnant and a potential source of disease and pollutants on the reef. Conch are also a major food source for some large species of fish on the reefs including sharks and eagle rays. Between these roles they hold a vital position in the functioning of a healthy and diverse reef system. Unfortunately the Conch is also commercially important as a food source for the local communities or more often a catch that is sold by local fishermen nationally and internationally. This means that Conch populations have become very low on many reefs and despite many areas where they are protected or their populations managed, their general abundance remains low.

The reefs around the Cayos Cochinos archipelago are no exception to this, with historically large fishing pressure on the Conch resulting in low numbers on the reefs. However, with the establishment of the Cayos Cochinos Marine Protected Area (CCMPA) the fishing of Conch was made illegal in the area. It is one of the few species to receive complete projection in this manner, with most other fish and invertebrate species only receiving partial spatial or temporal protection. This then provides a unique opportunity to assess the regrowth of a Conch population after the removal of fishing pressures in the area, potentially a prime indicator of the success of the CCMPA. The project aims to tag individual conch so that the sites can be returned to in future years and measurements of population structure, movement and growth rate can be made.

Methods

The Cayos Cochinos MPA contains many different reef, seagrass and sediment areas and survey sites were selected to incorporate these three habitat types. Sites were surveyed by conducting three 50m transects and recording all conch visible from the transect line. Transects were either carried out by snorkelling or SCUBA diving, depending on the depth of the site. All conch encountered were measured and categorised according to the methodology set out in Tewfik *et al.* (1998).

Results

Full results of conch surveys at all survey sites between 2009-2011 are provided in appendix 5. During the study period, the highest abundance of conch was consistently found at Arena. Figure 6.1

displays the mean number of conch encountered at those sites surveyed in all three years (2009-2011). Numbers of conch encountered varied between sites and years.

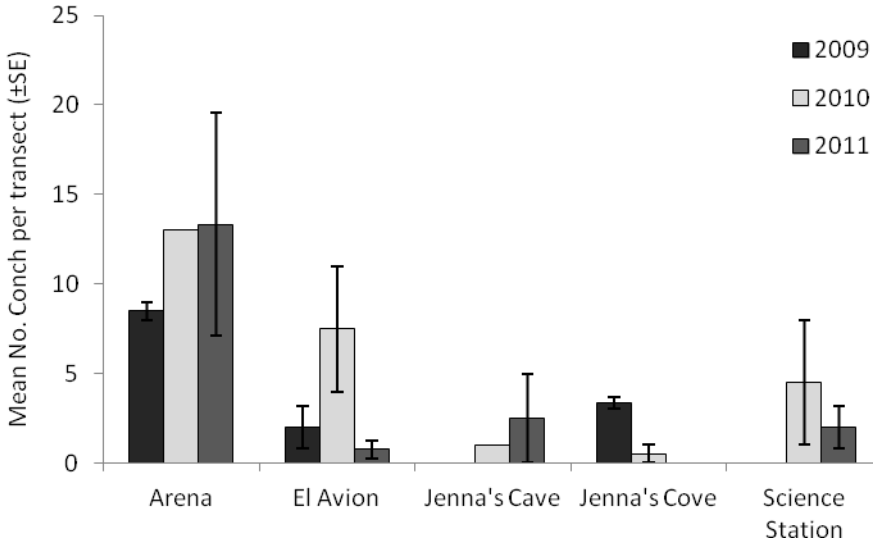

Figure 6.1 Mean number of conch encountered per transect for those sites surveyed in all three years (2009-2011). Error bars are the standard error of the mean.

The mean length of conch encountered at each site is provided in appendix 5. Mean length of Conch at the most densely populated site, Arena, was 26.0 cm (±0.29) in 2011 and size class frequency data is displayed in figure 6.2.

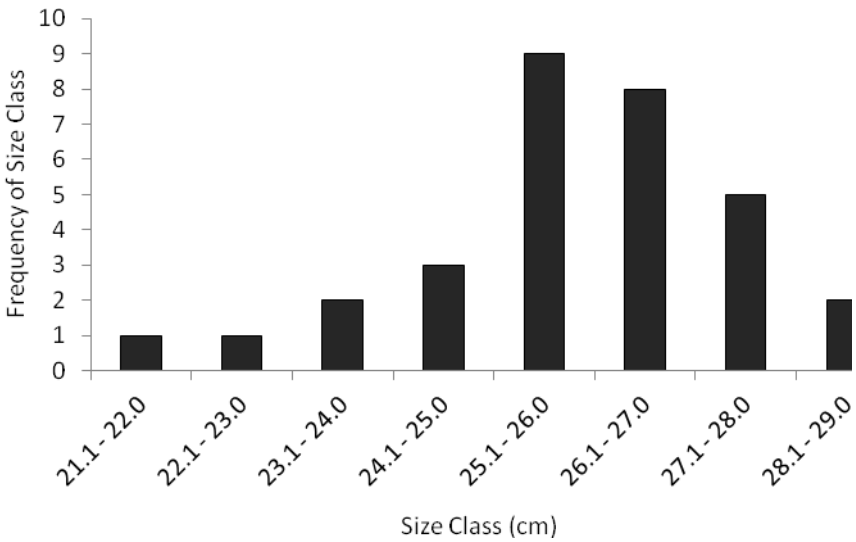

Figure 6.2 Size class frequency distribution histogram for conch encountered at Arena in 2011.

Discussion

This report provides only a preliminary look at the data collected by the conch research team. Mark-recapture data are still to be analysed to attempt to provide estimates of population size rather than relative abundance between sites. However, it is clear that the conch population is very varied

within the MPA and although some sites have large populations, others contain few if any conch. It may be worthwhile concentrating future efforts of the mark-recapture study on only those sites that have reasonably high encounter rates. This will allow better estimates of demographic parameters, such as population size, growth rates and mortality, to be calculated for the Cayos Cochinos conch populations.

References

Tewfik, A., Guzmán, H. and Jácome, G., (1998). Assessment of the Queen conch *Strombus gigas* (Gastropoda: Strombidae) Population in Cayos Cochinos, Honduras. *Revista de biologia tropical* 46:4 Pages 137-150.

7: Herpetofauna Research Group

The Herpetofauna Research Group is subdivided into three core areas of research: 1) Boa constrictor conservation and ecology, 2) Spiney-tailed Iguana (*Ctenosaura melanosterna*) conservation and ecology, and 3) predicting the response of *Anolis* ('anole') lizard communities to climate change in the Bay Islands and Cayos Cochinos. This report outlines the current progress and results of each of these core areas of research

Boa constrictor conservation and ecology

The *Boa constrictor* research group was initiated in 2004 to determine the extent to which the Cayos Cochinos *B. constrictor* population had been impacted by illegal poaching for the pet trade, and to collect ecological and genetic data essential for implementing appropriate management strategies to ensure the successful conservation of the population. Eight years on, a huge wealth of data has been generated and is currently being analysed by teams of researchers at Universities in the UK and USA. Dr. Stephen Green has recently completed his Ph.D. at the University of Kent, UK, in collaboration with Operation Wallacea and his Ph.D. thesis, entitled "Evolutionary Biology and Conservation of the Hog Island Boa constrictor" is available on request from the University of Kent Library, or direct from Dr. Green (Steve.green@opwall.com). The following is a summary of the major findings of this research project to date.

Introduction

The Hog Island Boa constrictor is a dwarfed insular race of *Boa constrictor imperator* endemic to two small islands (Cayo Mayor and Cayo Menor) in the Cayos Cochinos archipelago, Honduras. The population reportedly experienced severe decline as a result of intense collection for the pet trade throughout the late 1970s and 1980s, during which time thousands of snakes were removed from the islands (Porras 1999; Reed et al. 2007). Just a decade after collection began, a herpetological expedition to the Cayos Cochinos was unable to find a single specimen of this previously abundant snake, and local residents involved in the trade confirmed that as of 1988 virtually all adult boas had been removed from the islands (Wilson & Cruz Diaz 1993). In 1993 the Cayos Cochinos was declared a protected area and in 1994 the Honduran Coral Reef Foundation (HCRF) was established to facilitate the protection, restoration and sustainable management of the area under the legislative decree 1928-93 (HCRF & TNC 2008). Since this time, the removal of boas has been dramatically reduced and the population appears to be recovering, although illegal poaching of boas from the islands remains problematic.

Capture-mark-recapture analysis of the Cayo Menor population estimates current adult census size to be in the region of 700 individuals, with genetic analysis suggesting the Cayo Mayor population to be of a similar size. Although evidence of a recent genetic bottleneck was detected in both populations, the rapid rate at which the populations recovered from the demographic bottlenecking event may have prevented the loss of substantial genetic diversity. Phylogenetic analysis reveals that populations of *B. c. imperator* in the Cayos Cochinos and on the nearby Bay Islands form a monophyletic group that likely diverged from the mainland approximately 2 million years ago.

Dwarfism has subsequently evolved rapidly in the Cayos Cochinos since the islands were last isolated from the mainland by rising sea levels at the end of the last ice age. Thus, the Cayos Cochinos and Bay Island populations represent an Evolutionary Significant Unit of high conservation priority representing both historical and recent evolutionary divergence of the species on islands. Conservation management strategies should focus on conserving this important historical genetic diversity while maintaining the ecological processes responsible for evolutionary variation in the Cayos Cochinos.

Research objectives

This study attempts to address a number of research objectives in relation to *B. constrictor* populations in Central America and in particular the insular dwarfed populations of the Cayos Cochinos, Honduras. The primary objectives are:

i. To determine the phylogenetic relationships of populations of *B. constrictor* in the Cayos Cochinos and Bay Islands with mainland populations in Honduras and Central America as a whole, and the evolutionary timescale over which observed levels of diversity have evolved.
ii. Investigate the level of population structure and gene flow that exists between island populations and the mainland, specifically between dwarfed and non-dwarfed populations.
iii. Assess the impact that unsustainable collection for the pet trade has had on the Cayos Cochinos population and the severity of the potential genetic bottleneck through which the population has passed.
iv. Assess the level of population recovery in the Cayos Cochinos, measured through current adult census and effective population sizes.

Methodology

Visual Encounter Surveys (Cayos Cochinos)

Visual encounter surveys (VES) for boas were conducted on Cayo Menor (CMN), and to a lesser extent on Cayo Mayor (CMY), during and outside of the Operation Wallacea field season between July 2004 and August 2011. The number of participants varied between VES depending on the number of volunteers present, however, the number of participants and the time spent searching were recorded for each VES in an attempt to quantify search effort. Experience of participants was also variable, ranging from completely naive to experienced field herpetologists. Average duration of each VES was approximately 1-2 hours. Search effort was estimated as being the time spent searching multiplied by the number of participants. In addition to boas caught during VES, boas were also captured opportunistically at other times while moving around the islands.

All boas encountered were captured by hand and either placed in a cloth snake bag and taken back to the field station for processing, or processed in situ and released immediately. Boas that were taken to the field station for processing were released at the exact point of capture within 48 hours. The Universal Transverse Mercator (UTM) coordinates of the exact capture site were obtained using a hand-held Global Positioning System (GPS) in order to plot capture locations across the island and estimate local abundance and density.

Processing boas

Snout-vent length (SVL) and tail length (TL) were measured by stretching the snake along a tape measure fixed to the laboratory bench, or if processing in the field, by stretching the tape measure along the snake. Sex was determined by observing the size of the cloacal spurs and the relative length of TL to SVL (males having enlarged spurs compared to females and relatively longer tails). If sex could not be determined with confidence, sex was confirmed by the use of hemipenial probes. All new captures were implanted with a Passive Integrated Transponder (PIT) tag (11 x 3 mm) and the unique ten digit identification code recorded. Subsequent recaptures were identified by scanning boas using a Biomark PIT tag reader. A tissue sample was then taken in the form of 1-3 ventral scale clips from every new snake captured and retained for genetic analysis. Tissue samples were stored in >75% ethanol in screw top plastic sample tubes.

During the study, each of the Bay Islands (Utila, Roatan and Guanaja) and Mainland Honduras were also visited to collect tissue samples from *B. constrictor* for genetic comparison with the Cayos Cochinos populations. Tissue samples were also kindly collected opportunistically by various collaborators on the Bay Islands and across mainland Central America.

Genetics Methods and data analysis

Phylogenetic analyses were carried out using mitochondrial DNA genes *ND4* and *cytochrome-b*. Population genetic analyses were performed using eight polymorphic microsatellite loci. Detailed descriptions of the genetic methodologies used can be found in Green (2011).

Results

During the study period more than 1017 captures were made of 715 individual snakes (302 recaptures) on CMN and 106 captures of 106 individual snakes (7 recaptures) on CMY. Encounter and capture rates of *B. constrictor* were not consistent over CMN, with some areas resulting in a much greater number of captures than others (Figure 7.1). Population estimates for CMN based on mark-recapture data suggest the adult population to be 698 (401-1389 95% C.I.). Detailed results of the demographic population models and genetic analyses are provided in Dr. Green's Ph.D. thesis available on request from the University of Kent or direct from Dr. Green (steve.green@opwall.com).

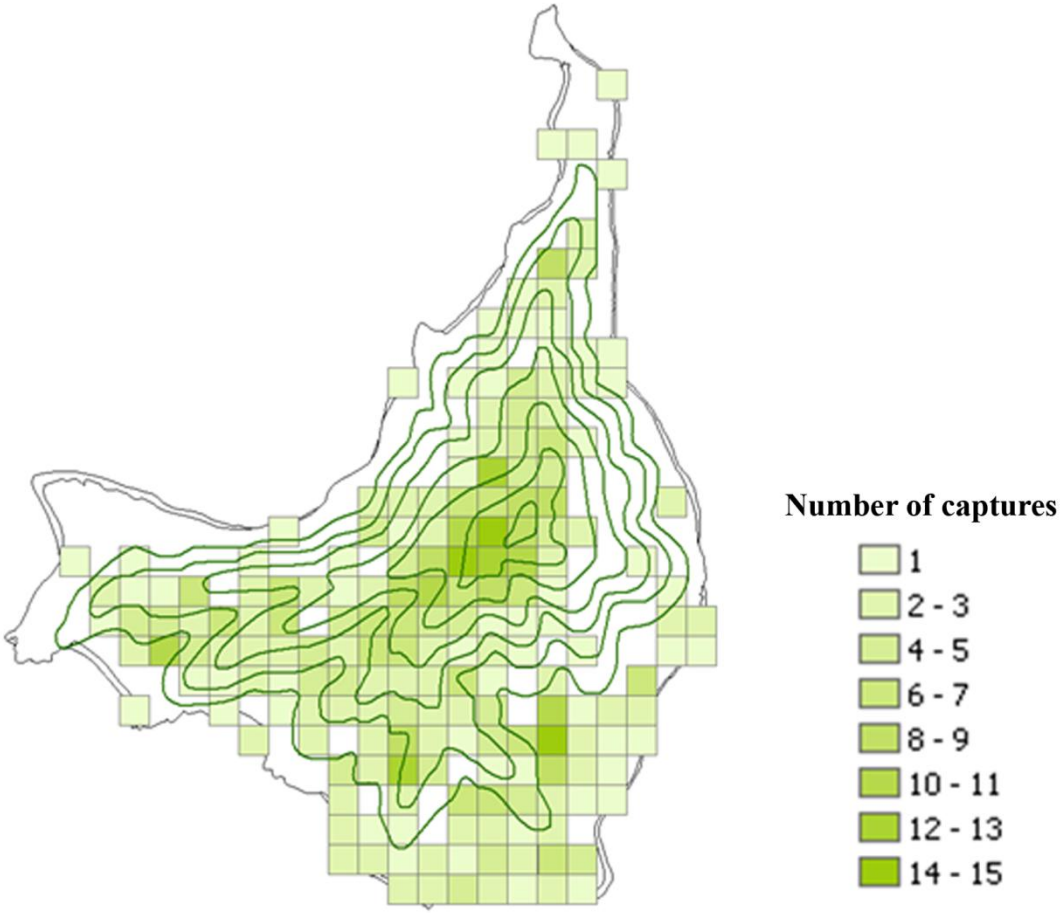

Figure 7.1 Density plot of capture locations on Cayo Menor. Grid squares represent 50m^2 and colours relate to the number of individual boas encountered within that grid square during the study period (excluding recaptures). Darker colours represent areas of higher encounter rates and thus assumed higher relative population density. Areas that are not covered by a grid square indicate no boas were encountered in that area.

Conclusions

The findings of this study suggest that: (i) the Islas de la Bahía represent a monophyletic clade that diverged from the mainland at the start of the Pleistocene and may qualify for recognition as an ESU, but Utila appears to have also been subsequently colonised from the mainland; and (ii) dwarfism appears to have evolved rapidly in the Cayos Cochinos only relatively recently, most likely in response to the reduced availability of large prey items on the islands.

Low levels of population structure were detected within the Cayos Cochinos, but the two islands can clearly be distinguished from one another based on allele frequencies. The two populations have been diverging from one another for approximately 5,000 years and most likely colonised the islands after their most recent isolation from the mainland by rising sea levels at the end of the last ice age. Cayo Menor and Cayo Mayor can thus be considered as distinct populations with high historical but low contemporary levels of gene flow rather than as a single freely interbreeding population. It is important that conservation planners recognise this level of population structure when developing appropriate management strategies for the Cayos Cochinos. Further sampling of the other islands

and mainland populations will be necessary to fully assess migration and population structure between the island and mainland populations.

Although the genetic signature of a bottleneck was detected in the Cayos Cochinos populations, general levels of genetic diversity do not appear to be of immediate conservation concern. Loss of genetic diversity may have been minimized by the rapid recovery of the population in response to increased protection provided after the creation of the Cayos Cochinos Protected Area. However, it may be too soon to determine the full extent and long-term impact on the overall fitness and survival of the population. Given sufficient time and the continued enforcement of anti-poaching legislation, the population may well make a full recovery. However, as is always the case with small populations, stochastic processes will have a large part to play in that recovery. The illegal removal of boas from the Cayos Cochinos continues to be problematic and a significant concern for the long-term conservation of the wild Hog Island Boa population.

Initial analysis of the mark-recapture data set has provided the first detailed quantifiable insights into the recovery of the Cayo Menor Boa constrictor population. Despite some potential limitations of the study, it is clear that the population has recovered substantially since anti-poaching legislation was first enforced in 1993. The continued collection of data over future years and a greater attempt to standardize search effort between samples will hopefully result in greater confidence in parameter estimates and provide further insights into the demography of the population. This information will hopefully help to inform future conservation management plans to ensure the long-term survival of this unique dwarf insular *Boa constrictor* in the wild.

Spiney-tailed Iguana (*Ctenosaura melanosterna*) Conservation and Ecology

Introduction

The Black-Chested, Spiney-tailed Iguana (*Ctenosaura melanosterna*) is listed as critically endangered by the IUCN red list with its distribution restricted to the Rio Aguan Valley in northern Honduras and the Cayos Cochinos. The Rio Aguan Valley population is in decline and its future uncertain. The Cayos Cochinos population, therefore, represents an important refuge for this species and its protection is a conservation priority for the area. However, small insular populations are vulnerable to extinction as a consequence of stochastic events. Therefore, it is crucial that the parameters acting on this population are understood in order to implement appropriate management strategies.

Ongoing research is being carried out to monitor the population size of *C. melanosterna* on Cayo Menor through capture-mark-recapture methods. However, during the 2011 field season, in addition to the long-term objectives of the population estimate, the general health of individuals captured was assessed. Body Condition Index (BCI) and parasite load were used as general indicators of health, and blood hematocrit levels as an indicator of stress. These indicators of health and stress were then used to compare animals living in close proximity to the research centre, where anthropogenic disturbance is higher, with those animals living in more 'natural' conditions away from the research centre. A key objective was to determine whether the presence of researchers on the island was having a negative impact on the health of animals around the research facility on the south of the island.

Methods

Individual Ctenosaurs were captured either using a noose and pole or by strategically placed nets which were checked on a regular basis. The majority of individuals were capture on Cayo Menor, however, some individuals were also captured on Cayo Mayor for comparison. On Cayo Menor, capture effort was focussed around the research station on the south of the island as well as the more remote palm forest habitat on the northern tip of the island. On Cayo Mayor, capture effort was focussed close to the village of East End.

For each individual, snout-vent length (SVL), tail length and mass were recorded. Sex was determined by inspection of the femoral pores on the underside of the hind legs of individuals. The number of external parasites (ticks) was recorded and a blood sample was drawn from either the caudal vein (using a syringe) or from the postorbital sinus (using a capillary tube), for subsequent analyses. Finally, individuals were marked using a PIT (Passive Integrated Transponder) tag for long term identification and also given an external marking with white paint to allow easy identification of recently captured individuals, so as to avoid wasting effort capturing the same individuals. This type of marking is only short-lived as Ctenosaurs shed their skin regularly. Hatchlings were not PIT tagged, but were instead toe clipped for future identification and will be PIT tagged at the point of future recapture once they are sufficiently large to do so.

Determining hematocrit level and making blood smears for blood parasite identification requires further treatments. The hematocrit level (which consists of the portion of the blood accounting for red blood cells only) is read with a microhematocrit capillary tube reader after spinning the capillary tube for five minutes. To make a blood smear, the blood is spread over a microscope slide. The blood is then dyed with Gemsa stain, and is left to dry for 20 minutes.

Results

During the 2011 field season, a total of 153 individuals were captured on Cayo Menor, as outlined in table 7.1. The number of captures from the north of the island was lower than from around the research centre due to lower population density and increased difficulty of capture.

	Research Station	North Palm Forest
Male	19	2
Female	38	6
Hatchling	73	15
Total	130	23

Table 7.1 Numbers of Ctenosaurs captured on Cayo Menor during the 2011 field season.

A regression analysis was conducted on log(SVL) and log(mass) and the residual scores used to calculate the BCI for each individual. An ANOVA test performed on BCI for males, females and hatchlings found no significant difference between these three groups (P=0.9754), suggesting that, based on this health indicator (BCI), there is no difference in health conditions between males, females and hatchlings. Therefore, data were pooled for these three groups in order to test if a significant difference in BCI existed between animals at the research centre and those found at the more isolated northern site. A t-test found there to be no significant difference in BCI between animals captured at the two sites (p=0.99)

Animals captured around the research centre and those captured from the north of the island were also compared for number of ticks. A t-test showed there to be no significant difference in tick number for individuals found at each site (P=0.1863). Analysis of hematocrit levels showed there to be no significant difference between males, females and hatchlings (ANOVA, p = 0.43). Similarly, no significant difference was found between the hematocrit levels of individuals captured at the research centre and the north of the island (p = 0.74).

Discussion

Analysis of the long-term capture-mark-recapture data set is not discussed in this report as this work is ongoing. However, in order to justify the continuation of such an investigation it was desirable to first establish that the investigation itself was not having a negative impact on the population. The population of *C. melanosterna* around the research centre on the southern side of the island is exposed to higher levels of human disturbance and has been studied more intensively over recent years than those animals found further away, such as those found in the northern palm habitat included in this study. We attempted to determine whether increased levels of human disturbance and direct study could be resulting in decreased health in the animals around the research centre.

Analysis of the data obtained in the 2011 field season suggests that there is in fact no difference in those health indicators selected for comparison. BCI, external parasite load (ticks) and hematocrit levels were found not to differ significantly between animals around the research centre and those found in the more isolated, and presumably more 'natural', habitat on the northern side of the island.

Predicting the response of *Anolis* ('anole') lizard communities to climate change

Introduction

To date, the Anole Project has focused on predicting the response to climate change among *Anolis* ('anole') lizard communities in the Bay Islands and Cayos Cochinos of Honduras. Since **2008,** numerous high-profile studies (two in Proc. Roy. Soc., three in PNAS, two in Science, and one in Nature) have all forecasted massive extinctions in tropical lizards driven by anthropogenic climate change. However, the majority of these studies have used coarse measures of physiology (e.g. critical thermal maxima or field-active body temperature). Moreover, they have attempted to make global predictions, which by necessity require global temperature data sets. Unfortunately, global temperature data are only available at a maximum resolution of about 1 km^2, when the vast majority of tropical ectotherms experience temperature at a much smaller scale (usually less than 1 m^2). To examine the generality of these predictions, the Anole Project has employed a novel approach, integrating **holistic** measures of thermal physiology (the entire thermal reaction norm of each species) with fine-scale measurements of contemporary thermal heterogeneity. By projecting climate change at a fine spatial scale onto thermal reaction norms, we are beginning to understand how lizard populations will respond in years to come. And as it turns out, not all species are doomed, and the predicted outcome of climate change differs among islands. The project summary below is organized into two parts, 1) New analyses and results from data collected in 2010, and 2) Analyses and results from data collected in 2011.

New Analyses of data from 2010: Comparison between forest anoles on Utila and Cayo Menor

The anoles *A. bicaorum* and *A. lemurinus* are ecologically-similar forest species that occur on the islands of Utila and Cayo Menor, respectively. They are sister species, and share a common anscestor as recently as 8,000 years ago. While Utila and Cayo Menor are less than 50 km distant from one another, these islands differ dramatically in topographical complexity. Utila is uniformly flat, while Cayo Menor is composed of numerous ridges and valleys, the only flat land represented by beaches around the perimeter of the island. Despite forest habitat being broadly similar among islands, we hypothesized that differences in landscape structure would translate to differences in the thermal environment as experienced by lizards. We measured the thermal environment at high-resolution on both islands using operative temperature models (OTMs) that mimic the thermal properties of each species. In 2010, OTMs were deployed at 60 randomized locations within typical forest environments on each island. Figure 7.2 shows that forest habitat, from the perspective of a lizard, is generally warmer on Utila than on Cayo Menor. Figure 7.3 shows that the spatial variance in temperature is generally lower on Utila. In the face of climate change, the relatively high spatial variation in forest operative temperatures on Cayo Menor suggests the existence of thermal refugia for *A. lemurinus*.

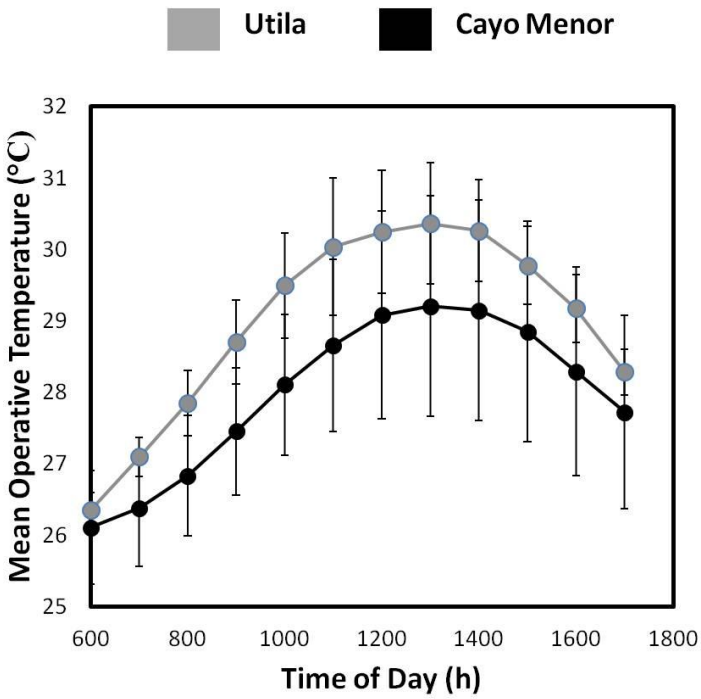

Figure 7.2 Mean operative temperature across a typical day during the dry season on Utila and Cayo Menor (OTM measurements were averaged for every hour across the entire study period). Forest on Utila is warmer during most of the day. Error bars represent standard deviation.

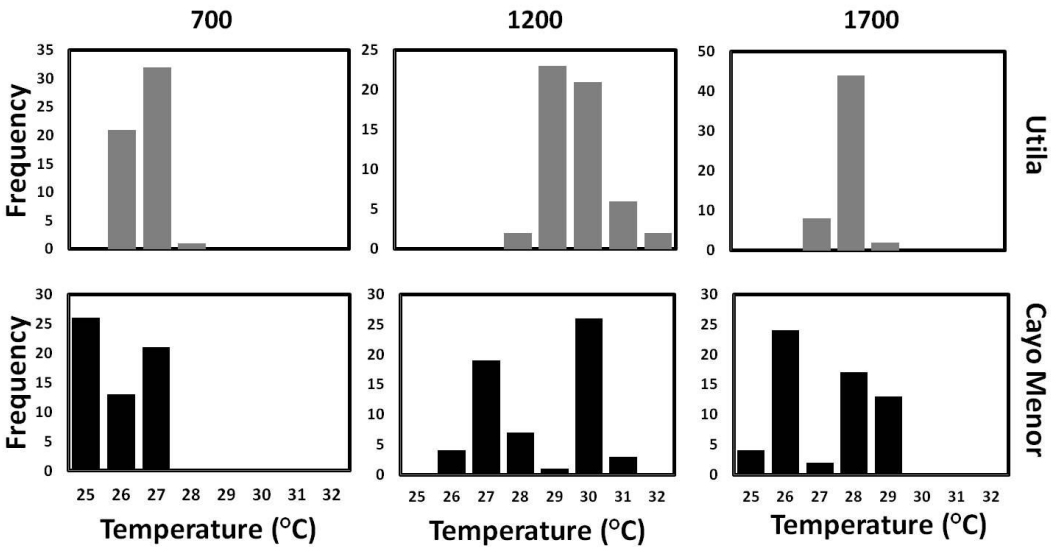

Figure 7.3 Frequency distributions of forest operative temperatures for Utila (first row) and Cayo Menor (second row) at three times of the day: 700 (first column), 1200 (second column), and 1700 (third column). Despite broad similarity between forest habitat on both islands, the spatial variation in operative temperature on Cayo Menor is much higher, suggesting the existence of thermal refugia permitting A. lemurinus to escape the effects of climate change.

When we integrate the average diel varation in forest operative temperature on each island with the thermal reaction norm (running speed as a function of temperature) for each species, we see that the predicted response to climate change differs dramatically between *A. bicaorum* and *A. lemurinus*. By the year 2100, *A. bicaorum* should be experiencing a significant drop in performance, while *A. lemurinus* should experience no net change (Figure 7.4). Two important observations immerge from this. First, these two closely related tropical forest species differ dramatically in their succeptibility to climate change, contrasting greatly with the sweeping predictions for tropical forest lizards made in recent high-profile publications. Second, the species that is more vulnerable to climate change (*A. bicaorum*), is the species with the *higher* optimal temperature for performance (Table 7.2), and is endemic to the island of Utila. The loss of the Utila population would therefore mean the death of the entire species.

Table 7.2. The characterstic values that describe the shape of the thermal reaction norms for the forest species *A. bicaorum* and *A. lemurinus*, on Utila and Cayo Menor, respectively. T_o is the optimal temperature for performance, P_{max} is the maximal performance, B_{80} and B_{95} are the 80% and 95% performance breadths, respectively, and CT_{max} is the critical thermal maximum. These values were extracted after fitting the raw performance data with 25 asymmetrical peak functions built into the statistical program TableCurve 2.0. The best fit for each species was chosen using Akaikes Information Criterion (AIC).

Species (island)	T_o (°C)	P_{max} (m/s)	B_{80} (°C)	B_{95} (°C)	CT_{max} (°C)
A. bicaorum (Utila)	31.6	0.77	27.8 – 32.6	30.3 – 32.2	33.2
A. lemurinus (Cayo Menor)	29.6	0.72	24.8 – 31.7	27.7 – 30.9	33.6

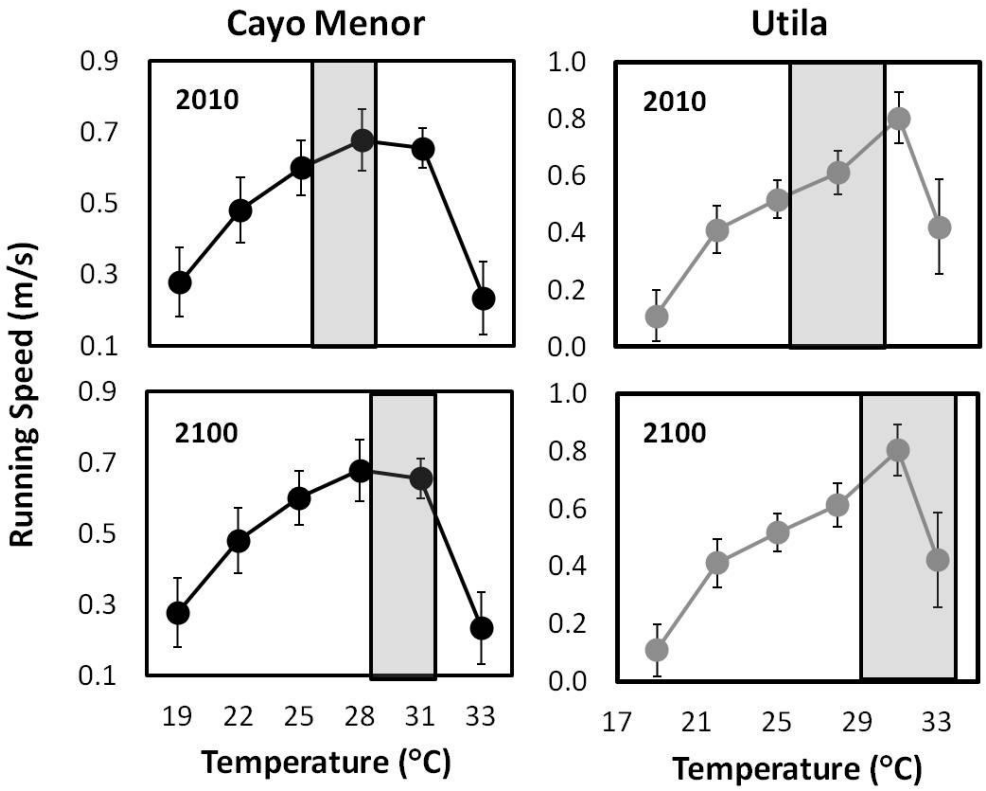

Figure 7.4 The projected effect of climate change on running performance in the forest species *A. lemurinus* (first column) and *A. bicaorum* (second column). Shaded regions show the mean diel operative temperature range in 2010 (first row) and after 90 years of climate change (second row). Error bars represent standard error. Despite the higher T_o of *A. bicaorum*, this species appears to be more vulnerable to climate change as performance will be depressed during a significant portion of the day by the year 2100.

Analyses from 2011: The effects of operative temperature, humidity, and wind speed on the abundance of A. lemurinus and A. allisoni on Cayo Menor.

One way in which climate change is hypothesized to effect the viability of species is through reduced activity time driven by increasing temperatures, which in turn means reduced foraging opportunity. However, the interacting effects of other abiotic variables (like humidity and wind) on lizard activity patterns remain relatively unexplored. In 2011, we conducted daily abundance surveys (using

distance sampling techniques) along transects where OTMs were deployed simultaneously. At the start and end of each transect we also recorded absolute humidity and wind speed. We conducted surveys along two transects per species, and at various times of the day in order to attempt to tease apart the effects of these abiotic variables on lizard abundance (or 'activity'). We asked two primary questions: 1) Are patterns of abundance consistent with the thermal physiology of each species? 2) Is temperature the most imporant abiotic variable that drives daily patterns in lizard activity?

First, patterns of abundance on Cayo Menor were broadly consistent with the thermal reaction norms of *A. lemurinus* and *A. allisoni*. The highest abundance for both species occurred when mean operative temperatures were within 0.5°C of their T_o (Figure 7.5). Nevertheless, when data from both transects were combined, only a small proportion of variance in abundance was explained by temperature in *A. allisoni* (the open-habitat species on Cayo Menor). When transects were examined independently, temperature explained a large proportion of variance in abundance for transect 1, but not transect 2 (Figure 7.6). After further analysis, it was determined that wind speed was much higher on transect 2, and *A. allisoni* abundance drops significantly when wind speeds increase above approximately 0.2 m/s (Figure 7.7). We surmise that this pattern is driven by high winds causing water loss due to convection, likely a strong constraint on the ability of lizards to remain active. These results suggest that, while temperature is an important variable driving lizard abundance on Cayo Menor, lizards can only stay active (and achieve optimal body temperatures) when wind speeds are low enough for them to avoid water loss.

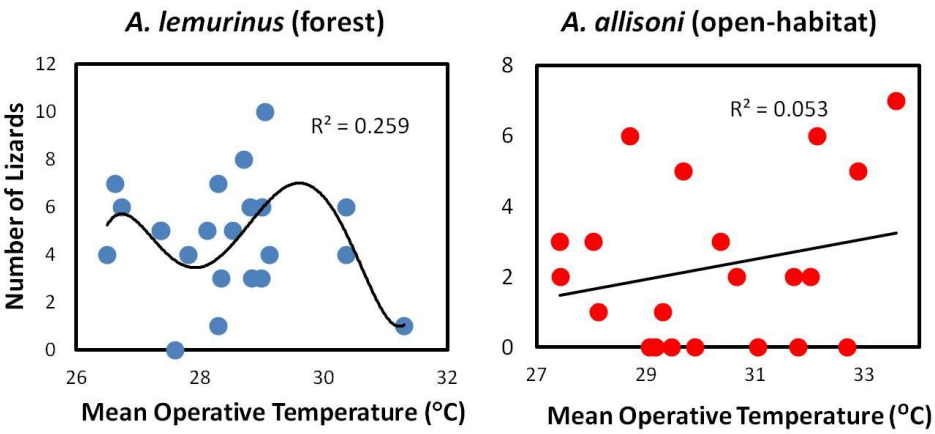

Figure 7.5 Total lizards observed along transects as a function of mean operative temperature in *A. lemurinus* (blue) and *A. allisoni* (red). More of the variance in abundance was explained by temperature in *A. lemurinus* than in *A. allisoni* when wind speed was not taken into consideration.

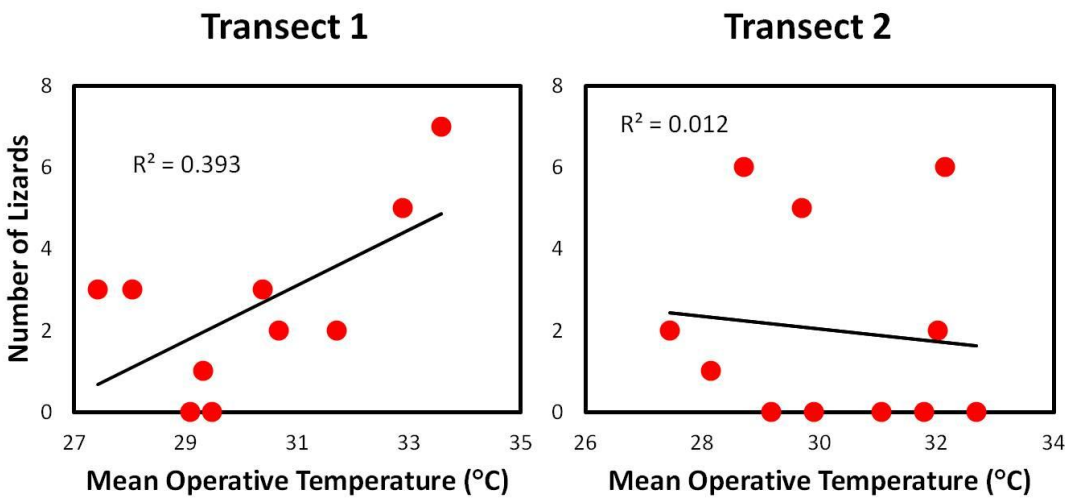

Figure 7.6 Total lizards observed along transects 1 and 2 as a function of mean operative temperature in *A. allisoni* (the open-habitat species). More of the variance in abundance was explained by temperature on transect 1.

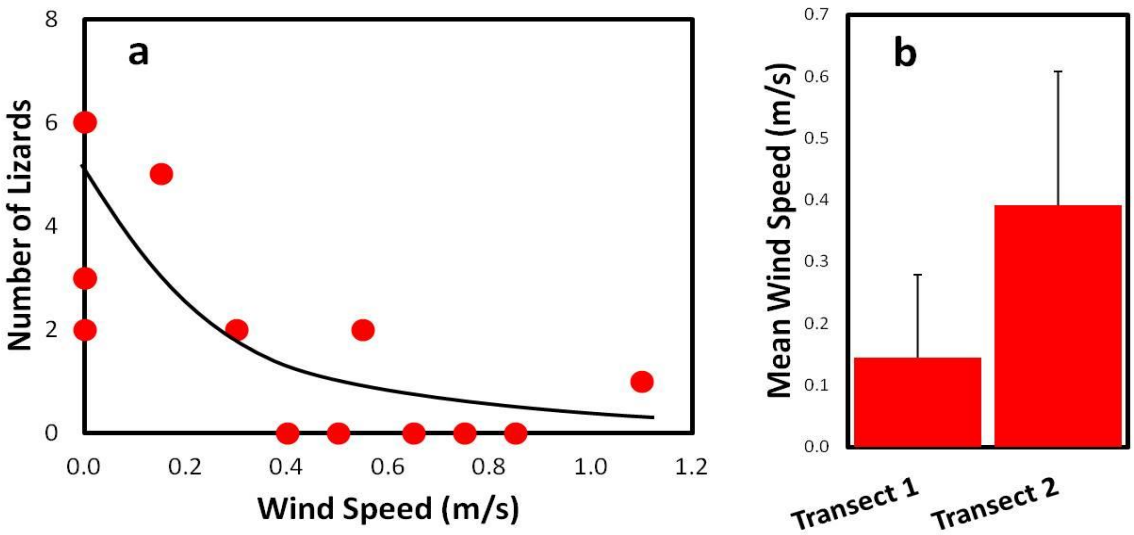

Figure 7.7 a) Total number of *A. allisoni* (the open-habitat species) observed on both transects as a function of wind speed, and b) the difference in mean wind speed between *A. allisoni* transects. The lack of variance in abundance explained by temperature in transect 2 is due to higher wind speed on that transect. This pattern suggests that a trade-off exists between the ability of lizards to achieve optimal body temperatures through basking and water loss due to convection.

Research Group Publication Output

Boback, S.M., C.E. Montgomery, R.N. Reed, and S. Green. 2006. *Oxybelis aeneus* (Brown Vinesnake). *Herpetological Review* 37(2):242.

Boback, S.M., C.E. Montgomery, R.N. Reed, and S. Green. 2006. *Kinosternon leucostomum* (Mud Turtle). *Herpetological Review* 37(2):239.

Frazier, J.A., C.E. Montgomery, S.M. Boback, and R.N. Reed. 2007. *Coniophanes imperialis* (Black–striped Snake) Diet. *Herpetological Review* 38(1):86.

Frazier, J.A., C.E. Montgomery, S.M. Boback, and R.N. Reed. In Press. *Leptophis mexicanus* (Mexican Vinesnake). *Herpetological Review*.

Green, S., Vuong H., Kundu S., Griffiths R. A., Montgomery C. E., Boback S. M., Reed R. N. and Groombridge J. J. **Phylogeography and the origins of dwarfism in a giant snake. (In prep.)**

Green, S., Montgomery C. E., Boback S. M., Reed R. N., Frazier J., Kundu S., Griffiths R. A. and Groombridge J. J. Genetic consequences of the pet trade induced decline of a dwarfed insular race of snake; the Hog Island Boa (*Boa constrictor imperator*). (In prep.)

Green, S., Montgomery C. E., Boback S. M., Reed R. N., Frazier J., Kundu S., Griffiths R. A. and Groombridge J. J. Population structure and gene flow of *Boa constrictor imperator* in the Cayos Cochinos and Bay Islands, Honduras. (In prep.)

Green, S., Montgomery C. E., Boback S. M., Reed R. N., Frazier J., Griffiths R. A. and Groombridge J. J. Assessing population recovery of a critically exploited insular *Boa constrictor* in the Cayos Cochinos, Honduras. (In prep.)

Logan, M. L., Montgomery, C. E., Boback, S. M., Reed, R. N., and Campbell, J. A. 2012. Divergence in morphology, but not habitat use, despite low genetic differentiation among insular populations of the lizard *Anolis lemurinus* in Honduras. *Journal of Tropical Ecology* 28:215-222.

Logan, M. L., Huynh, R. K., Precious, R., Calsbeek, R. Measuring the impact of climate change at relevant spatial scales: new hope for tropical lizards. In prep.

Montgomery, C.E., R.N. Reed, H.J. Shaw, S.M. Boback, and J.M. Walker. 2007. Distribution, habitat, size, and color pattern of *Cnemidophorus lemniscatus* (Sauria: Teiidae) on Cayo Cochino Pequéno, Honduras. *Southwestern Naturalist* 52(1):38-45.

Reed, R.N., S. Green, S.M. Boback, and C.E. Montgomery. 2006. *Ctenosaura melanosterna* (Black-chested Ctenosaur). Predation. *Herpetologcial Review* 37(1):84.

Reed, R.N., S.M. Boback, C.E. Montgomery, S. Green, Z. Stephens, and D. Watson. 2007. Ecology and conservation of an exploited insular-endemic population of *Boa constrictor* (Squamata: Boidae) in the Cayos Cochinos, Honduras. In: Biology of the Boas and Pythons. R.W. Henderson, R. Powell, G.W. Schuett, and M.E. Douglas eds., Eagle Mountain Publishing.

Walker, J. M. Boback, S.M., C.E. Montgomery, R.N. Reed, and S. Green. In Press. Morphology, diet and habitat use of *Cnemidophorus lemniscatus* in the Cayos Cochinos, Honduras. Herpetological Conservation and Biology.

References

Green, S. E. W. 2011 Evolutionary biology and conservation of the Hog Island Boa constrictor. Ph.D. thesis, University of Kent, UK.

HCRF & TNC. 2008 *Plan de Manejo del Monumento Natural Marino Archipiélago Cayos Cochinos Honduras 2008 – 2012*. Comité para la Restauración, Protección y Manejo Sostenible del Monumento Natural Marino Cayos Cochinos: TNC-Programa Honduras Nicaragua, TNC-Programa MAR.

Porras, L. W. 1999 Island Boa Constrictors (*Boa constrictor*). *Reptiles Magazine* **7**, 48-61.

Reed, R. N., Boback, S. M., Montgomery, C. E., Green, S., Stevens, Z. & Watson, D. 2007 Ecology and conservation of an exploited insular population of *Boa constrictor* (Squamata: Boidae) on the Cayos Cochinos, Honduras. In *Biology of the Boas and Pythons* (ed. R. W. Henderson & R. Powell). Utah: Eagle Mountain Publishing.

Wilson, L. D. & Cruz Diaz, G. 1993 The herpetofauna of the Cayos Cochinos, Honduras. *Herpetological Natural History* **1**, 13-23.

8: Overall Conclusions

The reefs around the Cayos Cochinos are protected from a variety of fishing pressures through the CCMPA, monitoring of the status of the fish populations with the area is, therefore, essential to gauge how successful the CCMPA is being in developing the fish populations within the park. The large scale monitoring of many reefs was conducted, assessing fish, coral, algae and invertebrate populations within the MPA. The introduction of Stereo Video Surveys into the protocol was an important advancement to the overall monitoring protocol and will produce valuable fish biomass data. Although much of the stereo video data is still to be analysed, preliminary analysis has highlighted some interesting differences between UVC and SVS, notably the apparent misidentification of some common species during UVC that cause abundance estimates to be artificially inflated. The continuation of SVS alongside UVC will provide standardised monitoring and high quality data that can not only be used to compare the CCMPA with the reefs of Utila, but with the reefs at the other Operation Wallacea marine research sites where SVS are being conducted (Cuba, Indonesia and Mozambique). This new technology provides a powerful tool for standardised comparisons of reef systems at the global scale.

Conch and urchin analysis has provided some useful baseline data to build on and future research should look to increase sampling effort at key sites. Apparent declines in urchin numbers since 2009 should be treated as a potentially worrying result, however, increased attention should be paid to standardising survey efforts in order to test to see if this is a real phenomena or a stochastic result due to low sampling effort. Future conch surveys should focus on those sites that support the largest numbers of conch in order to complete mark-recapture work that will generate reliable estimates of population demographic parameters.

The terrestrial herpetofauna research continues to provide valuable data that should be incorporated into the islands' management plan. For example, the identification of the Bay Island Boa constrictor as a potential Evolutionary Significant Unit (ESU), based on mitochondrial DNA, should be acknowledged and care taken to maintain the environmental conditions that have contributed to the Cayos Cochinos Boa's unique phenotype. Results from the anolis research project provide reason to be optimistic about the response of *Anolis lemurinus* to climate change. It suggests that the thermal environment of the forest is sufficiently heterogeneous for *A. lemurinus* to cope with the predicted temperature increase. However, this does not appear to also be true *for A. bicaorum* on Utila. Although the continuous monitoring results for the Ctenosaur melanosterna project are not discussed in this report, it was encouraging to show that our presence on the island and our study itself does not appear to be negatively affecting the health of the population.

The intention of this report is to provide an overview of the general monitoring work by Operation Wallacea volunteers. Data are constantly being analysed by research team leaders with the intention for peer-reviewed publication. For further information on any of the research teams please contact academics@opwall.com.

9: Appendices

Appendix 1 Fish species categories

Family	Common name	Latin name
Butterfly	Foureye butterflyfish	Chaetodon capistratus
	Spotfin butterflyfish	Chaetodon ocellatus
	Banded butterflyfish	Chaetodon striatus
Damsel	Sergeant Major	Abudefduf saxatilis
Grouper	Graysby	Cephalopholis cruentatus
	Rock hind	Epinephelus adscensionis
	Nassau Grouper	Epinephelus striatus
	Black Grouper	Mycteroperca bonaci
	Scamp	Mycteroperca phenax
	Tiger grouper	Mycteroperca tigris
Grunt	Caesar grunt	Haemulon carbonarium
	French Grunt	Haemulon flavolineatum
	Spanish Grunt	Haemulon macrostomum
	White Grunt	Haemulon plumieri
	Blue Stripped Grunt	Haemulon sciurus
	Black margate	Anistremus surinamensis
	White margate	Haemulon album
	Porkfish	Anistremus virginicus
Parrotfish	Redband Parrotfish	Sparisoma aurofrenatum
	Redtail Parrotfish	Sparisoma chrysopterum
	Redfin Parrotfish	Sparisoma rubripinne
	Stoplight Parrotfish	Sparisoma viride
Snapper	School master	Lutjanus apodus
	Yellowtail Snapper	Ocyurus chrysurus
Surgeon	Ocean Surgeon	Acanthurus bahianus
	Blue Tang	Acanthurus coeruleus

Appendix 2 Invertebrate categories

Sponge	Barrel Sponge
	Ball Sponge
	Encrusting Sponge
	Fire Sponge
	Rope Sponge
	Vase Sponge
	Tube Sponge
Anemone	Giant Anemone
	Corkscrew Anemone
	Branching Anemone
	Knobby Anemone
Zooanthid	Zooanthid
Gorgonian	Encrusting
	Sea Whip
	Sea Plume
	Sea Fan
	Black Coral
Mollusc	Triton
	Conch
	Flamingo Tongue
	Nudibranch
	Bivalve
	Squid
	Octopus
Crustacean	Pederson Shrimp
	Banded Shrimp
	Arrow Crab
	Spiney Lobster
Enchinoderm	Diadema Urchin
	Pencil Urchin
	Rock Bornig Urchin
	Sea Cucumber
	Brittle Star
	Feather Star
Other	Christmas Tree Worm
	Feather Worm
	Blue Bell Tunicate

Appendix 3 Bethic Cover Categories

CoverType

Algae	Amphiroa
	Caulerpa
	Corraline crustose algae
	Cyanophyta (fuzzball)
	Dictyota
	Halimeda
	Lobophora
	Padina
	Sargassum
	Udotea
	Valonia
	Coraline Crustose Algae
Coral	Acropora cervicornis
	Acropora palmate
	Agaricia agaricites
	Agaricia lamarcki
	Agaricia tenufolia
	Colpophyllia natans
	Diploria labyrinthiformis
	Diploria strigosa
	Eusmilia fagistiana
	Favia fragrum
	Madracis mirablis
	Meandrina meandrites
	Millepora sp.
	Montastrea annularis
	Montastrea cavernosa
	Montastrea faveolata
	Montastrea franksi
	Mycetophyllia sp
	Porites asteroides
	Porites porites
	Sidastrea sidereal
	Stephanocoenia intersepta
	Dichocoenia Stokesii
Sponge	Ball Sponge
	Barrel Sponge
	Fire Sponge
	Rope Sponge
	Vase Sponge
	Encrusting Sponge
	Tube Sponge
Misc. Inverts	Anemones
	Gorgonians
	Hydroids
	Polychaete
	Zooanthids
	Sea Fan
	Sea Plume
Substrate	Rock
	Recently killed coral
	Rubble
	Silt
	Sand

Appendix 4 Mean numbers of urchins encountered per transect across sites and years

Year	Site	N	Mean Long Spined (±SE)	Mean Pencil (±SE)	Mean Rock Boring (±SE)	Mean Reef (±SE)
2009	Alex's Point	6	3.50 (1.95)	151.83 (54.36)	14.83 (5.12)	3.17 (0.87)
	Arebaa Inside	6	31.67 (9.47)	81.83 (24.30)	114.17 (18.44)	30.00 (5.24)
	Arebaa Outside	6	1.83 (1.22)	51.50 (19.50)	69.67 (14.59)	4.17 (0.87)
	Arena	6	9.17 (4.94)	72.33 (18.65)	24.33 (6.16)	7.17 (1.89)
	Belfate	6	1.00 (0.63)	208.83 (63.49)	7.50 (3.67)	5.17 (1.30)
	Cayo Paloma	6	30.17 (8.65)	29.00 (9.02)	29.50 (14.90)	8.67 (1.80)
	Chachuate	6	4.67 (1.33)	54.50 (15.80)	60.50 (23.74)	13.67 (4.59)
	El Avignon	6	41.33 (13.32)	30.67 (8.87)	136.50 (42.06)	28.83 (7.01)
	Hotel Bay	6	0.33 (0.33)	8.50 (3.20)	32.67 (4.18)	3.83 (0.40)
	Jenna's Cove	7	2.17 (0.77)	11.00 (2.06)	31.33 (4.25)	5.67 (0.82)
	Menor East	6	0.50 (0.50)	9.00 (3.75)	28.50 (6.89)	7.50 (1.63)
	Menor West	6	5.33 (2.38)	75.83 (17.53)	29.17 (6.77)	11.17 (1.54)
	Pelican 0	6	4.33 (1.31)	70.17 (23.51)	69.50 (20.16)	13.83 (2.33)
	Pelican 2 + 2.5	5	1.00 (0.63)	49.40 (9.27)	15.20 (5.94)	2.00 (0.63)
	Pelican 3	6	5.17 (2.40)	58.67 (13.45)	26.33 (14.04)	3.50 (0.99)
	Pelican 4	6	12.17 (2.86)	17.67 (4.60)	23.17 (2.50)	6.50 (1.67)
	Pier West	6	39.17 (5.79)	8.83 (2.94)	113.50 (15.57)	5.83 (0.60)
	Timon	6	16.33 (6.46)	77.83 (34.91)	80.50 (29.43)	11.50 (2.83)

Year	Site	N	Mean Long Spined (±SE)	Mean Pencil (±SE)	Mean Rock Boring (±SE)	Mean Reef (±SE)
2010	Alex's Point	2	9.00 (2.00)	2.00 (1.00)	2.50 (0.50)	10.50 (3.50)
	El Avion	3	102.00 (3.46)	8.00 (3.46)	1.33 (0.88)	31.00 (15.39)
	Jenna's Cove	3	0.33 (0.33)	6.67 (2.03)	0.33 (0.33)	28.00 (6.93)
	Menor West	5	8.00 (4.35)	5.20 (2.40)	11.20 (8.49)	26.00 (19.44)
	Pelican 0	2	1.50 (1.50)	14.00 (1.00)	1.00 (1.00)	26.00 (8.00)
	Pelican 2 + 2.5	3	3.67 (0.33)	7.00 (5.51)	1.33 (0.88)	5.67 (2.73)
	Pelican 4	2	3.00 (1.00)	11.50 (1.50)	5.50 (5.50)	19.00 (6.00)
	Timon	2	0.00 (0.00)	2.00 (2.00)	2.50 (1.50)	18.00 (7.00)

Year	Site	N	Mean Long Spined (±SE)	Mean Pencil (±SE)	Mean Rock Boring (±SE)	Mean Reef (±SE)
2011	El Avion	2	50.50 (3.50)	0.00 (0.00)	0.00 (0.00)	1.00 (1.00)
	Jenna's Cove	5	0.60 (0.24)	0.00 (0.00)	0.00 (0.00)	1.00 (0.45)
	Pier West	4	2.00 (0.71)	2.25 (1.60)	0.00 (0.00)	2.25 (1.93)
	Cayo Menor (North	6	10.33 (4.81)	3.17 (0.79)	0.83 (0.54)	2.67 (1.15)
	Pelican 3	6	0.50 (0.50)	0.83 (0.83)	0.00 (0.00)	0.33 (0.21)
	Pelican 4	5	4.00 (2.53)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)
	Timon	3	7.33 (2.40)	0.67 (0.33)	0.00 (0.00)	0.00 (0.00)

Appendix 5 Mean number of conch encountered and mean conch size in the Cayos Cochinos 2009-2011

Year	Site	No. Transects	Mean Abundance per transect	SE
2009	Arena	2	8.50	0.50
2009	Arriba East	3	0.00	0.00
2009	Balfate West	3	0.00	0.00
2009	Balonos	3	0.33	0.33
2009	Cayo Gallo (East)	3	2.67	1.45
2009	Chachahuate	3	0.00	0.00
2009	El Avion	3	2.00	1.15
2009	Hotel Bay	4	0.25	0.25
2009	Jenna´s Cave	3	0.00	0.00
2009	Jenna´s Cove 1.5	2	9.00	1.00
2009	Jennas Cove 1	3	3.33	0.33
2009	Largos Arriba West	3	0.00	0.00
2009	Menor West	3	0.33	0.33
2009	Paloma	3	0.00	0.00
2009	Paloma (East)	3	0.00	0.00
2009	Peli 2.5	3	0.33	0.33
2009	Peli 3	3	0.00	0.00
2009	Science Station	4	0.00	0.00
2009	Timon	3	2.00	1.15
2010	Arena	1	13	-
2010	Bolanos	1	0	-
2010	Cayo Balfate (West)	1	2	-
2010	Cayo Gallo (East)	1	3	-
2010	Cayo Largo Arriba (E)	1	0	-
2010	Cayo Largo Arriba (W)	1	1	-
2010	Cayo Paloma	2	0	-
2010	Cayo Timon	1	5	-
2010	Chachahuate	1	1	-
2010	East End	1	0	-
2010	El Avion	2	7.5	3.5
2010	Hotel Bay	1	4	-
2010	Jenna’s Caves	1	1	-
2010	Jenna’s Cove	2	0.5	0.5
2010	Menor West	1	0	-
2010	Paloma (East)	3	0	-
2010	Pelicano 2.5	1	1	-
2010	Pelicano 3	2	3.5	0.5
2010	Science Station	2	4.5	3.5
2010	The Rock	1	0	-

2011	Arena (3)	3	13.33	6.23
2011	El Avion (4)	4	0.75	0.48
2011	Jenna´s Cave (2)	2	2.50	2.50
2011	Jenna´s Cove (4)	4	0.00	0.00
2011	Science Station (3)	3	2.00	1.15
2011	Timon (4)	4	0.50	0.29

Number of individual conch (excluding within season recaptures) and mean length of conch encountered at each site in each year. SE is the standard error of mean conch length.

Year	Site	N	Mean Conch Length (cm)	SE
2009	Arena	29	24.8	0.80
2009	El Avion	8	26.1	0.66
2009	Hotel Bay	7	20.6	1.23
2009	Jenna's Caves	4	25.2	0.69
2009	Jenna's Cove	28	24.3	0.22
2009	Timon	11	25.6	0.47
2010	Arena	43	24.4	0.25
2010	Arriba West	4	16.0	2.80
2010	Belfate	7	20.1	2.84
2010	Bolanos	7	11.9	1.53
2010	Cayo Gallo	12	24.3	0.48
2010	Cayo Largo Arriba West	1	8.7	0
2010	Chachuate	7	12.4	2.47
2010	East End	1	21.1	0
2010	El Avion	15	18.6	5.47
2010	Hotel Bay	20	15.5	0.92
2010	Jenna's Caves	3	13.4	4.07
2010	Jenna's Cove	1	9.0	0
2010	Menor South West	20	24.0	0.33
2010	Menor West	42	24.1	0.21
2010	Paloma East	1	30.0	0
2010	Paloma West	1	24.3	0
2010	Pelican 2 + 2.5	2	24.9	0.40
2010	Pelican 3	24	24.9	0.34
2010	Science Station	18	17.4	1.61
2010	Timon	14	10.6	1.30
2011	Arena	31	26.0	0.29
2011	El Avion	3	24.8	0.76
2011	Jenna's Caves	5	26.6	0.73
2011	Science Station	6	25.6	0.67
2011	Timon	2	26.7	0.30