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GVI Seychelles – Curieuse

Island Conservation Expedition

Annual Report

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Seychelles National Parks Authority (SNPA)

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Special thanks

To all volunteers and staff from January 2018 – December 2018 for assisting with data collection.

GVI Seychelles – Curieuse Island Conservation Expedition

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Executive Summary

This report summarises the science programs conducted by the Global Vision International (GVI) Seychelles, Island Conservation Expedition on Curieuse Island, between January 2018 and December 2018.

The total rainfall for this time period was 2222.3 mm (compared to 2017 and 2016 rainfalls of 1867.5 and 1757.1mm, respectively).

The sixth year of the annual Aldabra giant tortoise census was completed in September 2018, with a total of 122 individuals being successfully located throughout the island. The majority of tortoises were located at the Ranger Station, with the remainder dispersed throughout the island. With no known adult mortality, the population appears to be stable. One additional free ranging sub-adult individual was located and added to the known population of free ranging individuals. The tortoise nursery on Curieuse now houses 74 juvenile tortoises. With increased security measures, the nursery not only offers protection from poachers, but also reduces the risk from introduced predators on the island, such as feral Black rats (*Rattus rattus*). Captive tortoise hatchlings will continue to be measured and weighed biannually, and new individuals electronically tagged.

Baited Remote Underwater Video (BRUV) surveys were conducted off the north coast of Curieuse for the second time, in order to assess the relative abundance and diversity of predatory and scavenging species. Four survey sites were selected, at each of which two deployments were carried out per replicate, one shallow (≤10m) and one deep (≥15m). A total of 103 target species were determined for monitoring and 71 were positively identified. Shallow deployments were observed to have a significantly higher level of fish diversity, although substrate type was not evenly distributed by depth. Deployments conducted to date have provided baseline data on fish abundance and diversity along the north shore of Curieuse Island, and continued BRUV surveys are expected to yield valuable data which can be used in the adaptive management of the MPA.

Beach Profiling continued on six beaches split into two sections, with each section being profiled every two months. Each year, substantial changes to beaches have been observed between the Northwest and Southeast monsoon seasons. Some trends are beginning to emerge, strongest on the beaches along the southern coast of Curieuse (Anse St. Jose, Anse Cimitiere, and Anse Caiman) with sediment movement in respect to both beach area and width observed to be shifting north-westerly

during the Southeast monsoon season and south-easterly in the Northwest monsoon. This is observed to a lesser extent on the eastern beaches (Anse Laraie, Anse Papaie, and Grand Anse). With the exception of Anse Caiman and Anse Cimitiere, there was a general downward trend in mean beach area and width from 2016-2018, with a large amount of variation between transects and months. As such, these results should be interpreted with caution and long-term trends should continue to be monitored through the existing research program.

The Coco de Mer growth survey has now produced 57 months of growth and reproduction data, which has elucidated growth rates for all life stages/sex classes. While there are no consistent patterns in male catkin production or flowering, the mean number of nuts per female tree has been displaying a steady increase throughout the duration of the study. This is promising in terms of the reproductive output of the Curieuse population and its capacity to survive and recover to historical levels, and it is planned that in the near future more focus will be placed on data related to female reproduction in order to better understand sustainable harvest levels of nuts within the park.

Permanent quadrats were monitored at eight locations within the Baie Laraie mangrove forest with the aim of investigating seedling recruitment and mortality and further determining species distribution. The two most dominant species in the mangrove forest are *Rhizophora mucronata* and *Bruguiera gymnorrhiza*. Considering the value that mangrove forests provide in terms of ecosystem services and the potential to improve its state, it is vital that mangrove monitoring continues in order to better understand, protect and rehabilitate the area.

The 2017-2018 Hawksbill turtle nesting season began in August 2017 and lasted until the end of March 2018. Nesting activities peaked in December, and Grand Anse remained the most heavily utilised nesting beach. Encounter rates also peaked in December; however, there were considerably less encounters and an overall smaller population estimate than the previous three seasons. Excavations showed high reproductive success for both Hawksbill and Green sea turtles, in line with previous seasons. Photo identification and metal flipper tagging are being continued in the 2018-2019 season. The number of Green turtle activities and nests has increased compared to previous seasons.

September 2018 marked the end of season four and the beginning of season five of juvenile Sicklefin lemon shark monitoring. The project has been very successful in providing baseline data to inform management of this important marine predator. Since the onset of the project until December 2018, a total of 521 individuals have been captured and tagged, and a total of 160 recaptures have been made, which has provided a wealth of information on changes in size, weight, and body condition. Population estimates each year have greatly exceeded preliminary estimates, and population size appears to be stable. We hope to further the tracking program through an acoustic array to help inform management regarding critical habitats for this species within Curieuse Marine National Park.

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Introduction

Global Vision International (GVI) Seychelles comprises two expeditions based on separate granitic islands. The Island Conservation Expedition is based on the small granitic island of Curieuse, located approximately 1km north of Praslin. Base camp is located at Anse St. Jose within the Curieuse Marine National Park (CMNP). This marine national park was established in 1979, and covers an area of 14.7km^2.

All of GVI's scientific work in Seychelles is conducted on behalf of and at the request of local partners, using their chosen methodology. GVI supplies experienced staff, trained volunteers, and equipment to conduct research in support of their on-going work. GVI's key partner in Seychelles is the Seychelles National Parks Authority (SNPA).

Seychelles National Parks Authority (SNPA): A parastatal organisation partly funded by the government, responsible for management and research relevant to the protection of the national parks within Seychelles.

Seychelles Islands Foundation (SIF): This organisation manages and protects the UNESCO World Heritage Sites of Aldabra and Vallée de Mai. Their work with the Coco de Mer forests on Praslin and other endemic plants and animals is closely linked with the flora, fauna and Coco de Mer population on Curieuse.

Island Conservation Expedition

The Seychelles archipelago represents the only mid-oceanic granitic islands on earth. Isolated for 75 million years, Seychelles now hosts a unique assemblage of flora and fauna, many of them extremely primitive. Such ancient species include endemic palm trees such as the Coco de Mer (*Lodoicea maldivica*) and Aldabra giant tortoises (*Aldabrachelys gigantea*). However, 200 years of human settlement has exerted a negative influence on the native biota of these islands. Habitat loss and fragmentation, as well as invasive species, have resulted in several extinctions and reduced populations of many species to perilous levels. Natural resource exploitation continues to pose a serious threat to Seychelles' native flora and fauna (Hill 2002).

Curieuse Island is a small granitic island (2.86km^2) in Seychelles, approximately 1km north of the island of Praslin. Curieuse is notable for its bare red earth intermingled with the unique Coco de Mer palms, one of the cultural icons of Seychelles - present only in three main populations on Praslin and Curieuse.

In 1979, Curieuse and its surrounding waters were declared Curieuse Marine National Park in order to protect the native wildlife. Today, it is home to approximately 126 free ranging Aldabra giant tortoises (*Aldabrachelys gigantea*), found primarily at the Ranger Station but also in smaller numbers throughout the island. Sea turtles are often found in the surrounding seagrass and reef habitats, and several of Curieuse's beaches represent important nesting sites for female Green and Hawksbill turtles, particularly during their nesting season (between October and February). Another key component of the Curieuse marine ecosystem is the mangrove forest. Mangrove trees are found most extensively around the lagoon area at Baie Laraie, and bridge the gap between the marine and terrestrial environments, playing a key role in maintaining optimum reef building conditions for corals (Obura and Abdulla 2005) as well as providing a vital habitat for birds and fish, including the Sicklefin lemon shark.

The objectives of the Island Conservation Expedition on Curieuse for 2018 focused on the continuation of the Coco de Mer growth survey, mangrove monitoring project, annual giant tortoise census, sea turtle monitoring, beach profiling, Sicklefin lemon shark monitoring, and the recently started Baited Remote Underwater Video surveys. The fundamental goal behind all fieldwork is to ensure data collected is relevant and valuable to our project partners. The information collected by GVI Seychelles is available through SNPA to help inform management decisions and for use as a baseline for future study.

Training

Island Conservation Health and Safety

All Expedition Members on the Island Conservation expedition are educated through safety inductions to work in all survey areas and walk off-path to study sites. Risk assessments have been carried out for all surveys undertaken. Volunteers are provided with first aid training through the Emergency First Response course, which is taught on-site.

Terrestrial & Marine Species identification and Field Techniques

GVI relies heavily on volunteers to carry out all of its fieldwork. These volunteers stay for periods of between two and 12 weeks. To ensure precision and continuity, all volunteers are intensively trained and have a fully trained staff member or experienced intern accompany them on all field surveys. All expedition volunteers are required to undergo training in any surveys they will be participating in during their stay, e.g. identify the various life stages of Coco de Mer palms, understand appropriate handling and measurements for giant tortoises, sea turtles and lemon shark pups and learn the six species of mangrove tree present on Curieuse. They are also trained in how to operate equipment used for each survey, which includes a GPS, PIT tag scanner, Abney level and shark capture equipment. Training is initially provided in the form of presentations, classroom sessions and informal discussions with the expedition staff, followed by in-field training in practical field techniques. Self study materials are also available in the form of textbooks, field guides, journal articles and flashcards. Volunteer progress is monitored and staff supervision remains vigilant until each volunteer demonstrates a grasp of all procedures and is able to identify key species. Volunteers are required to pass an exam prior to participating in any mangrove surveys.

Study Sites

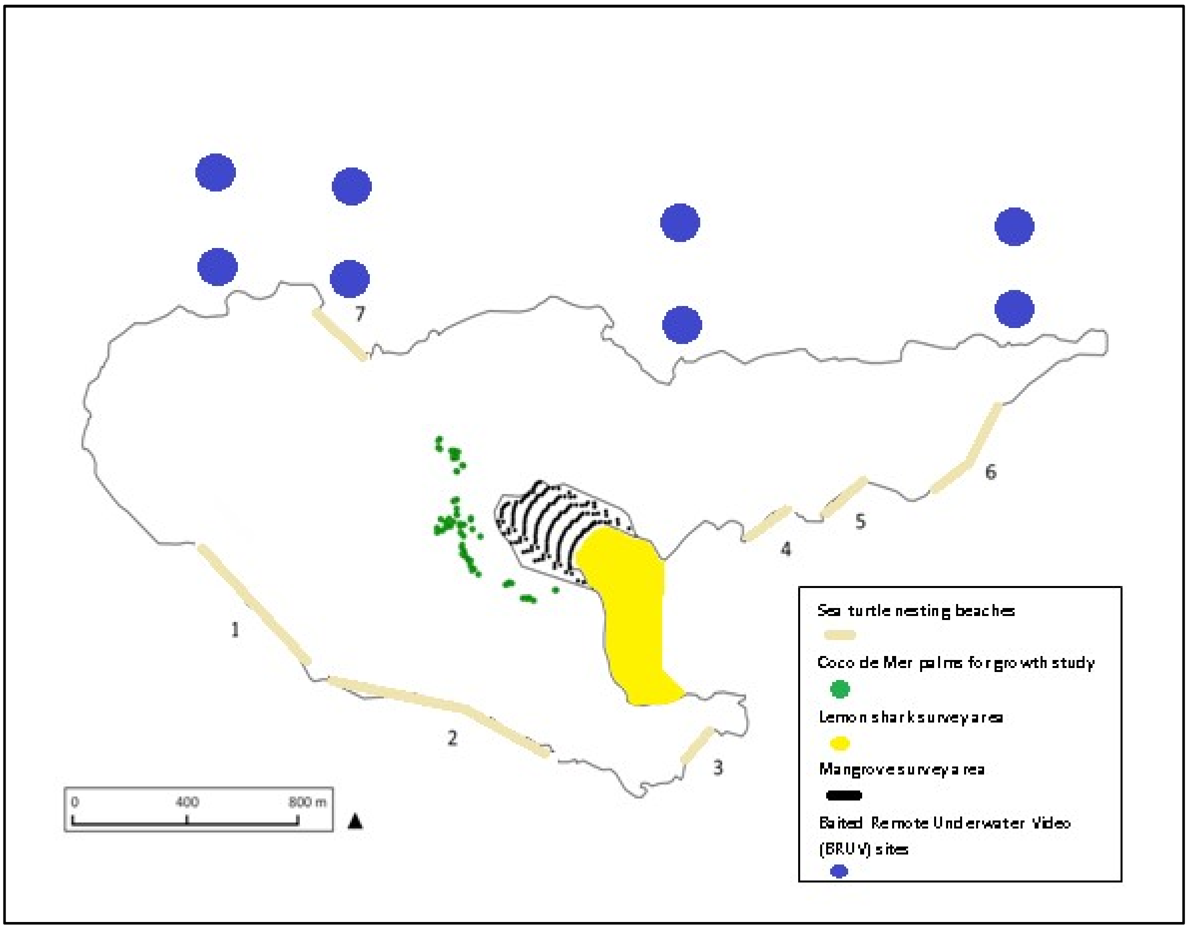

Figure 1. Curieuse Island, showing current survey sites. 1. Anse Caiman/Cimitiere, 2. Anse St. Jose, 3. Anse Mandarin, 4. Anse Laraie, 5. Anse Papaie, 6. Grand Anse, 7. Anse Badamier.

Aldabra Giant Tortoises

Introduction

Most islands in the Western Indian Ocean, including the Inner Granitic Islands of Seychelles, once hosted wild populations of giant tortoises (Stoddart et al. 1979). However, populations have declined due to exploitation and exportation since the 1700s. Currently, the only natural wild population of Aldabra giant tortoise (*Aldabrachelys gigantea*) is believed to be found on Aldabra Atoll (Gerlach *et al*, 2013). All remaining giant tortoises in the Inner Granitic Islands are thought to have been relocated from Aldabra.

The ‘Curieuse Experiment’ introduced around 250 tortoises over a period of four years, starting in 1978. Initially released on Curieuse near the Ranger Station at Baie Laraie, the majority of the tortoise population remained there, however some have migrated and individuals can now be found throughout the island (Sanchez et al. 2015; Samour et al. 1987).

Tortoises are clearly reproducing successfully on Curieuse, as hatchlings have been found by SNPA Rangers and GVI personnel each year. Heightened efforts to increase recruitment into the population and hatchling survival have been taken by SNPA. A nursery was established to protect hatchlings from any predators, poaching, and handling by tourists. At the age of approximately five years old, when they are large enough to be safe from rat predation, they are released into the free ranging population.

In 2013, the first annual GVI census found 125 tortoises, significantly fewer than the 250 originally released on the island over 30 years ago. The overall decrease in population size is alarming, and stresses the importance of conducting an annual census and consistent monitoring of the population.

Aims

The primary aims of the annual census were to reveal how many of the originally relocated tortoises from Aldabra remain on Curieuse and to locate any free ranging individuals added to the population since then, along with their basic movements across the island. Over time, this census is designed to determine tortoise growth rates, home range, age (when followed from hatchling size), and the size at which tortoises begin to display sexual characteristics. Aldabra giant tortoises have been researched on Aldabra Atoll; however, habitat differences between the atoll and the Inner Granitic

Islands likely have an impact on the habits and growth rates of the tortoises. The lack of an increase in the population size raises questions related to population recruitment and hatchling survival. The census also aims to increase the likelihood of discovering hatchlings that have successfully hatched in the wild. Another aim is to locate as many tortoise nests as possible, and subsequently conduct excavations in an attempt to shed light on clutch size and hatching success rates.

In addition to the yearly census of the free ranging tortoises, there is also a biannual census of hatchlings in the nursery, where similar growth measurements are taken to allow hatchling growth to be tracked.

Methodology

The giant tortoise program was conducted in two parts: an annual census of the free ranging population throughout Curieuse Island, and regular monitoring of juveniles residing in the nursery.

Giant Tortoise Census

In line with GVI census methodology from previous years, the island was searched using a map (Figure 2) marked with location codes based on a previous tortoise study (Lewis *et al*, 1991). GVI personnel spent time in each location, but especially areas known to be favoured by the tortoises, namely the Ranger Station, Anse Papaie, Grand Anse, the north and south mangroves and Anse Badamier. It was assumed if GVI personnel could not traverse certain terrain, then neither could the tortoises. If no signs of tortoise activity (e.g. droppings) were found, especially in difficult to access areas, further effort was expended in other locations.

Each time a tortoise was encountered, it was first identified to determine whether it had been encountered previously. There are a number of ways to identify previously encountered individuals, including a unique ID number which is applied to the carapace of the tortoise using a yellow paint stick (*Sharpie MeanStreak*) on the 4th or 5th dorsal scute (Figure 5). This allows for rapid identification of individual tortoises without the need to scan for internal tags. However, this mark is not permanent and lasts only weeks or months. Therefore, if a tortoise was unmarked, it was scanned for an existing Passive Integrated Transponder (PIT) tag (*Trovan ID 100*) using a scanner (*Trovan GR250*). If previously untagged, a PIT tag was inserted.

When tortoises were initially relocated to Curieuse, and again during a census on Curieuse in 1997, a metal disc was attached to the 4th dorsal (D4) carapace. A plastic disc was also attached to D4 to a majority of the tortoises in 2013. If any discs were still present and legible, the numbers were recorded. If it was obvious there once was a tag (evidence of glue remaining from a missing disc), ‘MD’ for ‘missing disc’ would be recorded. If neither the tag nor glue from the Aldabra and/or the Curieuse census discs was present, then an ‘N’ for ‘not present’ was recorded.

If it was determined that a tortoise had not yet been encountered during the present census, then the date and time was recorded. In order to aid future surveys, and to monitor the movement of PIT tags throughout the tortoise’s body, the PIT Tag Location (where the PIT tag was detected with the scanner, e.g. left rear hip or tail) was recorded. The location of each tortoise encounter was recorded using a GPS. Additionally, the location was matched to an Area Number (Figure 2) in order to allow for current data to be compared with historical data.

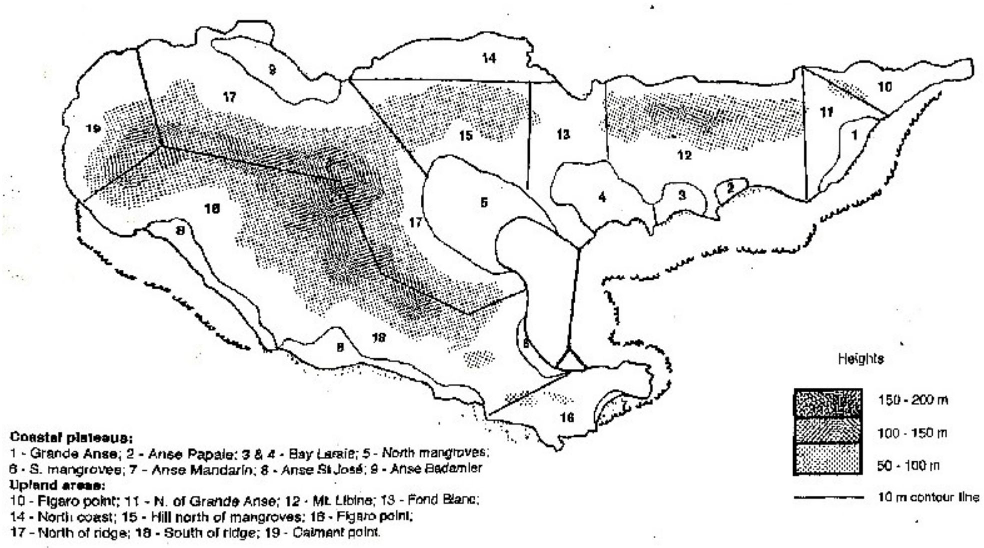

Figure 2. Map of Curieuse Island used in the annual census, originally used by the Oxford Expedition team in 1990 (Lewis et al. 1991).

Various measurements were taken for each individual tortoise to allow for analysis of growth. The carapace width and over-the-curve carapace length (OCCL), as well as the width of the 3rd dorsal scute were measured (Figures 3 and 4). Three characteristics (tail length, plastron shape, hind claw length) are thought be indicative of gender. A plastron can be defined as being ‘concave’, ‘slightly concave’ or ‘flat’. The length of the tail was recorded as being ‘long’ or ‘short’, with long tails being those that extended past the midline of the 11th marginal scute (Table 1) and short tails being those that didn’t. Length of the second claw from the rear of the hind right leg was measured.

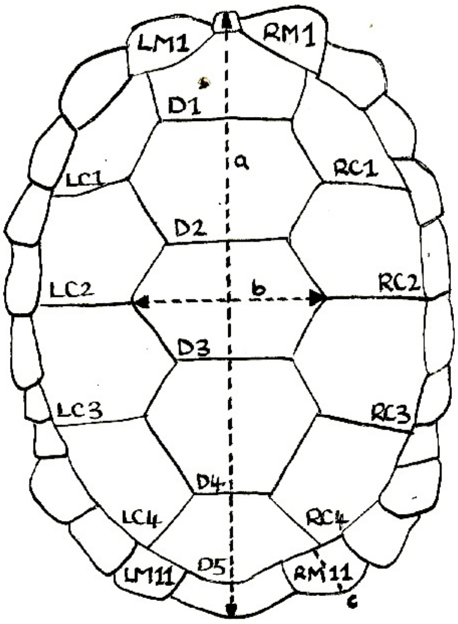

Figure 3. Tortoise carapace (upper shell). Dotted line a*:* over-the-curve carapace length (OCCL), b: width of the 3rd dorsal scute, c: point on marginal scute 11 on either side, past which the tail is considered to be Long.

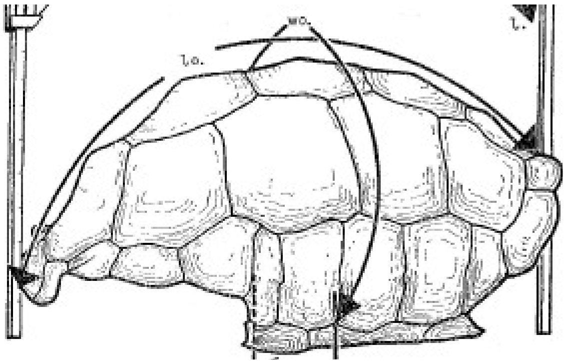

Figure 4. Width measurement as recorded over the carapace (from Gaymer 1968).

A scale to determine the thickness of the white lines between scutes was developed, with the theory being that a thick white line indicates that a tortoise is not yet fully grown. These lines, if present, were measured to the nearest millimetre. After all data had been collected and the yellow ID number repainted if needed, a photo of the ID number was taken to identify the tortoise along with a photo of the 3rd dorsal scute and any distinguishing marks and injuries.

Sexually mature males are believed to typically have long tails, concave plastrons and short hind claws while females have the opposite. The apparent sex of a tortoise was determined by the criteria defined in Table 1, with these criteria being introduced to the methodology in 2015. Only sexually mature males can be sexed using visual characteristics alone; a small tortoise with a short tail and flat plastron could either be an immature male or a female. Only a tortoise that has been seen digging a nest, laying eggs, or cooperatively engaging in copulation can be confidently sexed as female, though these events are not seen often. Juveniles were classed as having an OCCL of less than 70cm. Therefore, for the purpose of this analysis, tortoises were classed as: 'full male', 'potential male' (those starting to display male sexual characteristics, specifically a slightly concave plastron), 'reproductive female', 'juvenile', or 'unknown' (immature male or female).

Table 1. Visual characteristics used to assess the life stage or gender of a giant tortoise.

	Male	**Potential Male**	**Reproductive Female**	**Juvenile**	**Unknown (immature male or female)**
Tail	*L=long	n/a	S=short	S=short	S=short
Plastron	*C=concave	SC = slightly concave	F=flat	F=flat	F=flat
OCCL	≥70 cm	≥70 cm	≥70 cm	*<70 cm	≥70 cm
White Line	n/a	n/a	n/a	2-3	n/a
Growth since the 1997 census?	n/a	n/a	No	n/a	n/a
Female Reproductive Activity	n/a	n/a	*NB or CoCop	n/a	n/a

Grey background indicates a supporting characteristic
* indicates a characteristic that is definitive independent of other characteristics
NB - Nesting behaviour is defined as nest digging or egg laying
CoCop - Cooperative copulation defined by observed penetration

Monitoring of Captive Hatchlings

Upon discovery in the field, any new hatchlings were taken to the nursery, where they were implanted with a PIT tag (*Therachip ISO FDX-B* transponder scanned with *Petscan RT 100 V8*). Fortnightly checks were made to ensure tag retention and hatchling survival. Similar growth data was collected as for the free ranging individuals every six months. Hatchling age was estimated based on body size and condition when they were found. Hatchlings were measured (width, OCCL, width of 3rd dorsal), weighed, and marked (with *Sharpie MeanStreak*) on specific marginal scutes with a unique pattern for future identification. Photographs were taken of the carapace, plastron and any distinguishing marks such as extra/missing scutes (i.e. front, back, top of carapace, underneath of plastron, right side, and left side).

Results

Tortoise Location

In this year’s census, 122 tortoises have been encountered and identified. Eight individuals were not encountered in 2018; two of which (121 and 123) have not been found for the past five years, three (006, 038 and 115) have not been found since 2014, and one (050) has not been seen since 2015. An additional two tortoises were added to the Curieuse census population this year. Two new juvenile tortoises were found in January (135) and October (136). Both had measurements taken and PIT tags inserted. 135 was worked up and later released at the Ranger Station, while 136 (of larger size) was both worked up at, and remained at Grand Anse. As such, a population of 122 free ranging tortoises are known to be present on Curieuse.

Tortoises were encountered primarily at the Ranger Station (n=99, 81.15%). Grand Anse had the second largest concentration (n=12, 9.83%) followed by Anse Papaie (n=4, 3.27%) and the mangroves (n=3, 2.46%). Anse St. Jose, Anse Badamier, Figaro Point and Mt. Libine had the lowest concentrations (all with n=1, 0.82%).

Gender and Size of Tortoises

This year, the census population consisted of five juveniles, 68 full males, 30 potential males and 19 unknown. Table 2 displays the average width, OCCL, 3rd dorsal width and hind claw length for tortoises in each age/sex class in 2018. When hind claw length was considered as a percentage of the OCCL, unknown sex had the highest ratio (3.68%), followed by full males (3.67%), potential males (3.66%), and then juveniles (3.15%).

Table 2. Mean ±SD for measurements taken over each age/sex class in the 2018 census.

	Juvenile		**Potential Male**		**Full Male**		**Unknown**	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD
3rd Dorsal (cm)	19.22	2.12	33.36	3.61	33.24	7.29	32.70	4.19
Width (cm)	63.58	6.92	119.25	14.33	118.68	16.45	116.42	16.15
OCCL (cm)	60.34	8.45	121.57	17.82	121.03	18.12	118.52	20.24
Hind Claw Length (cm)	1.90	0.49	4.46	0.77	4.44	0.77	4.36	0.83

Average OCCL growth from 2013-2018 by age/sex class is shown in Table 3. Tortoises classified as unknown had the highest average growth (4.22cm), and potential males showed the lowest average growth (3.48cm).

Table 3. Mean OCCL growth for each of the age/sex classes between the 2013 and 2018 censuses.

	Potential Male (n=26)		**Full Male** (n=67)		**Unknown** (n=16)	
	Mean	SD	Mean	SD	Mean	SD
OCCL (cm)	3.48	3.91	3.50	3.86	4.22	4.76

Tortoise Hatchlings

With 25 hatchlings stolen by poachers in July 2016, growth measurements are reported for the 2016 cohort currently residing to present (24 hatchlings), for which average growth is provided in Table 4. All tortoises measured exhibited consistent positive growth.

Since the poaching incident, the number of hatchlings present in the nursery has been steadily increasing again (Figure 5). The nursery population decreased slightly between January and August 2017 and again from January - October 2018 due to predation by rats, death due to natural causes and the release of individuals 126, 128 and 135. 20 new hatchlings have been found since November 2019 bringing the current nursery population to 74.

Table 4. Average growth in OCCL and weight for 24 hatchlings after approximately two years of growth (3/10/2016 to 6/7/2018).

	Mean	**Standard Deviation**
OCCL (cm)	3.05	1.08
Weight (kg)	0.08	0.05

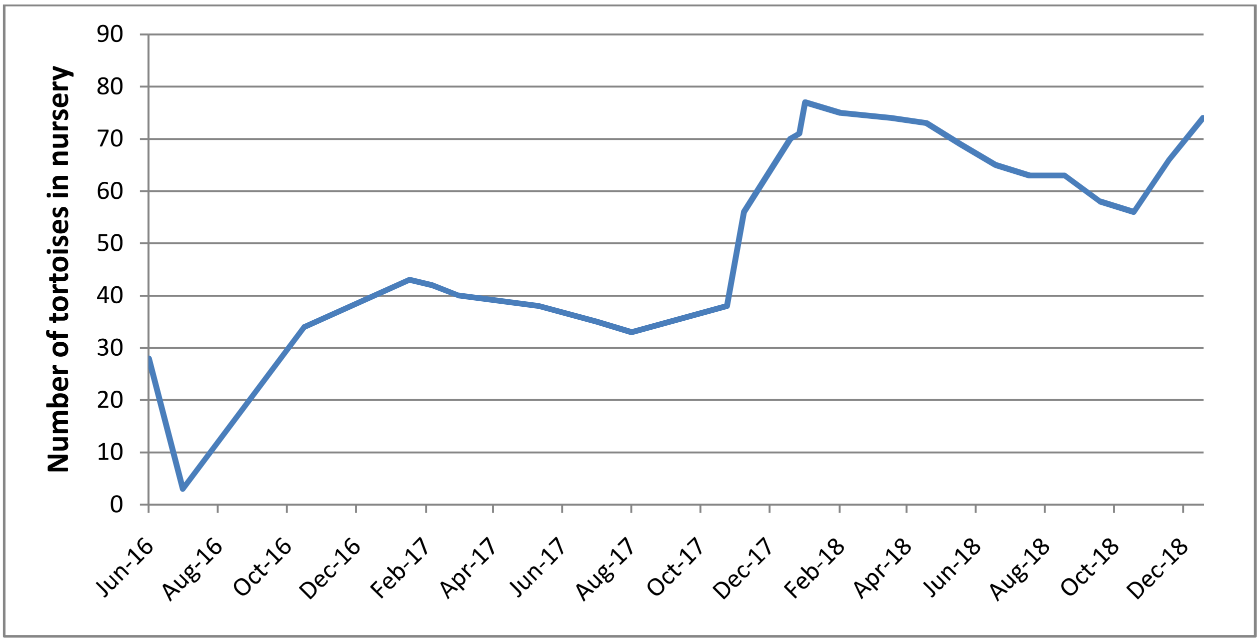

Figure 5. Number of tortoises in the nursery from June, 2016 to December, 2018. Juvenile tortoises 126, 128 and 135 have been included (until their release in November 2018), but are analysed in the sub-adult/adult census population.

Discussion

The 2018 annual giant tortoise census continues to show that the methodology used for encountering tortoises remains effective, with 122 out of 130 known individuals encountered (93.8% of the free ranging population), including two previously undiscovered individuals. However, as with previous censuses, this year has also demonstrated the difficulty of locating some individuals, and that tortoises are capable of navigating difficult terrain. With the discovery of two new sub-adult tortoises, it is evident that some undiscovered tortoises likely still remain.

In the previous censuses and over four years, at least five tortoises are known to have died, including one from the 2016 census. In the 2017 and 2018 censuses, no tortoises are known to have died, although in early 2018 tortoise 095 was in an unhealthy condition and not seen during the 2018 census. The mortality rate on Curieuse does not appear to be as high as other places such as Aldabra Atoll, however it is not possible to make direct comparisons due to the greater difficulty locating individuals on Curieuse on account of the drastically different habitat. Over future years of consistent data collection, a better picture of mortality rates may be established.

There has been little variation in encounter locations since 1986, with the majority of tortoises being consistently located in Areas 3 and 4, comprising the Ranger Station and surrounding area. This year 99 individuals were found there. There was initially an attempt to keep all of the animals in this area using a moat which over time was breached, allowing all tortoises to roam freely (Samour et al, 1987). The fact that tortoises are not distributed evenly across the island has been accounted for by difficult terrain and loss through poaching (Hambler 1994, Samour et al. 1987). Despite the steep and often rocky terrain, certain individuals have left the Ranger Station and have proven that giant tortoises are more than capable of navigating over difficult ground (Hambler 1994, Samour et al. 1987, Stoddart et al. 1982). Of particular note are tortoises 014 and 095, having previously made the journey over to the south side of the island. These two individuals have had to cross very difficult rocky terrain, with number 014 making the journey twice after being returned to the east side of the island by SNPA.

Grand Anse now has a known population of 12, as one new sub-adult was located there. Grand Anse is one of the largest beaches on the island, with a large stretch of swamp/forest behind it. It is possible that there is a feature of this habitat (e.g. food availability, shelter, etc.) that results in tortoises remaining there. Eight individuals were also found there in previous censuses, indicating they prefer this location. It will be interesting in future years to observe whether the number of individuals in this area continues to rise. Also, tortoise 124, which has been at Anse Badamier on the north coast of Curieuse for the past four years, appears to have established residency in the area.

Curieuse Island is a substantially different habitat to Aldabra Atoll, and different environmental pressures therefore affect the tortoise populations. It is not known at what age or size Curieuse tortoises begin to display sexual characteristics; therefore, we are not able to confidently state how many males and females are on the island. Reportedly on Aldabra, giant tortoises become sexually mature when they reach a size of 70cm OCCL and a 3rd dorsal scute of more than 21cm (Lewis et al. 1991). All Curieuse tortoises displaying slight indications of being male (i.e. slightly concave plastron) have an OCCL of greater than 74.3cm and a 3rd dorsal of more than 22.3cm. This implies that simply because a tortoise is larger than the threshold given by Lewis et al (1991)and has not yet displayed sexual characteristics does not mean that they will not. It could be that tortoises on Curieuse are either reaching sexual maturity at a later age, or growing at a faster rate. It has been previously hypothesised that tortoises on Curieuse may grow at a faster rate than on Aldabra, based on data from tortoises with a known age (Sanchez et al. 2015). This could be the case, though reports of the age of tortoises are generally unreliable unless scientifically monitored from soon after hatching. Only when a tortoise's exact age is known will it be possible to accurately compare growth rates on Curieuse and Aldabra, as well as determine the ages at which tortoises begin displaying sexual characteristics.

Data from the 2015 census appeared to support the theory that females have proportionally longer hind claws in relation to their size than males, but not in subsequent years. This theory still appears to be somewhat unproven and is currently not well supported by our data. For this theory to be examined thoroughly it will require more investigation in future years.

Since the start of the GVI annual census in 2013, OCCL was one of the measurements consistently taken, and is the best representation of overall growth. A subset of individuals with consistent data collected since 2013 showed that unknown tortoises had the highest average OCCL growth of 4.22cm between 2013 and 2018. This supports the theory that younger individuals display higher growth rates since this class contains a mixture of immature males and females, as they will still be growing to reach sexual maturity and maximum size. Potential males and full males displayed an average growth of 3.48cm and 3.50cm respectively between 2013 and 2018. This differs somewhat to previous years as full males have had a considerably slower growth rate than potential males. Continued yearly monitoring is crucial to determine if this trend continues. Some tortoises exhibited 0cm OCCL growth in this time, whilst two, an unknown who grew into being a male, and a full male, grew by 11.8cm and 14cm respectively. This shows that there is considerable variation in the growth rates of these tortoises.

Using the current classification system, no females have been reliably identified in the adult population. However, this cannot be the case since tortoises are clearly reproducing on the island and hatchlings are regularly relocated to the nursery. As such, it can be stated that the current classification system does not effectively identify females. It is suggested that another method should be used for sexing, and from there, measurements can be continued into growth rates specific to males and females. For example, the entire Curieuse tortoise population could be assessed for sex using an endoscope, which would provide an internal view of the reproductive organs, using a methodology similar to Kuchling et al. (2013) and Kuchling and Griffiths (2012). This would also streamline the tortoise census by removing the need to take hind claw and tail measurements, and the plastron classification, which is very much open to interpretation by individual observers, while providing more robust data on sex ratios and age/sex dependent growth rates.

The first reports of the Curieuse population naturally reproducing were in 1980, two years after they were relocated to the island (Stoddart et al. 1982), while Hambler (1992) found the number of emerging hatchlings to be slightly less than Aldabra, assumed to be due to differences in soil acidity. Mating is rare in the dry season on Aldabra, but Curieuse receives a greater amount of rainfall, which makes for better mating conditions year-round (Hambler 1992; Hambler 1994; Lewis et al. 1991; Swingland 1977).

Hambler (1994) estimated that 2,100-3,900 hatchlings had already been produced on Curieuse by 1993. Survival of these hatchlings is not supported by our census. Juvenile tortoises are difficult to locate as they tend to hide in the leaf litter (Grubb 1971; McFarland et al. 1974; Swingland and Coe 1979), and it is thought they may head for higher ground to avoid predation (Hambler 1994). By now these hatchlings would be sub-adults and would expectedly return to lower ground; this does not appear to be occurring. This suggests that when hatchlings remain free roaming on the island, mortality is high in the first few years (Gibson and Hamilton 1984). Curieuse has a large number of feral rats, which have been linked to the depletion of giant tortoise populations on other West Indian Ocean islands and in the Galapagos (Hambler 1992; McFarland et al. 1974; Swingland and Coe 1984). It is currently thought that Curieuse tortoises only lay one clutch a year (Lewis et al. 1991), and that rat predation could certainly be having a negative effect on hatchlings that are not found and taken to the nursery (Rainbolt 1996).

The tortoise nursery was established in order to bring hatchlings discovered on the island into a safe environment until they grow large enough to be safe from rat predation, and to protect them from the risk of poaching, especially after recent security improvements. Additionally, the nursery allows GVI to take consistent biannual growth data beginning shortly after hatching and ideally continuing throughout life. Considerable clearing of vegetation to the rear of the Ranger Station has allowed more tortoise nests to be found this year, largely explaining the increase of hatchlings in the nursery compared to previous years. For the future, the potential exists for SNPA and GVI personnel to establish surveys to search for tortoise nests, and cage the nests against predators. This, coupled with control of rats over time, would hopefully further increase the number of hatchlings and their survival rate on Curieuse.

Conclusion

This study aimed to continue the annual census of the population of Aldabra giant tortoises relocated to Curieuse. This year’s dataset included 122 tortoises in the free ranging population, the majority of which have remained close to their release site at the Ranger Station. Despite the fact some individuals were not found this year, looking back at previous censuses, it is not unusual for several tortoises not to be found in any particular year. Although it is known that at least five of the census tortoises have died, it may be that other missing individuals are alive in more inaccessible areas of the island. The addition of two new sub-adults indicates that more undiscovered individuals could remain.

Whilst the free ranging population does not appear to be increasing significantly, there doesn't appear to be a significant population decrease. However, it seems that many of the tortoises originally from Aldabra are no longer on Curieuse. With an increase in the number of hatchlings being found and taken to the nursery, it is promising for the future growth of the population.

GVI's annual census of the Aldabra giant tortoise on Curieuse Island over the past six years is beginning to provide the basis of a good quality long-term study. However, with only six years of data and considering the life history of this species, it could be many more years before significant trends in population structure, age at sexual maturity, and growth rates for the tortoises on Curieuse become clear.

Baited Remote Underwater Video Surveys

Introduction

Baited remote underwater video surveys (BRUVs) are a cost-effective, non-invasive tool that is becoming increasingly common for generating relative abundance and diversity indices for many marine species including sharks and other large reef fish (Malcolm et al. 2007; Brooks et al. 2011). BRUVs involve deploying a frame containing a video camera and a metal arm fitted with a canister containing bait (e.g. pulverised fish), which attracts fish and other marine species within the field of view of the camera. Traditional means of sampling fish assemblages can be extractive and often include the use of fishing gear such as longlines (used to monitor shark populations through direct catch) (Brooks et al. 2011). However, the use of longlines and subsequent shark handling can result in shark mortalities; potentially leading to a disturbance at the population level. Moreover, extractive fishing with complete or significant mortality as a method of sampling fish abundance is not in line with the conservation objectives of a no take marine protected area such as CMNP. Underwater visual surveys using SCUBA are a robust method of monitoring fish assemblages, though this technique is expensive due to equipment, logistical costs and specialised training (Bacheler and Shertzer 2015). Also, shy species such as sharks and other commercially important fish are typically less likely to be detected through underwater visual surveys using SCUBA (Willis et al. 2000).

BRUVs represent a viable, non-extractive, low-cost alternative to SCUBA, longlines and other potentially harmful methods for the assessment of fish assemblages within a marine protected area. BRUVs are also less size and species selective compared to baited fishing gear, and deeper waters can be sampled more easily than SCUBA based surveys (Bacheler and Shertzer 2015). There has been an overall lack of marine monitoring of fish assemblages within Curieuse Marine National Park (CMNP) since the marine monitoring program ceased at CMNP in 2011. In this context, BRUVs could serve as a cost effective, easily replicable, and robust tool for the assessment of fish assemblages over time.

Aims

The primary aim of this study is to establish baseline data on the diversity and relative abundance of target fish species within CMNP, which can be used to track changes in fish populations over time. This can aid in informing management actions related to the preservation of fish populations through adaptive management.

Methodology

Study Site and Sampling Structure

BRUVs were deployed at a total of four locations along the north coast of Curieuse Island (Figure 6); West Point (WP), Anse Badamier (AB), Fond Blanc (FB), and Point Rouge (PR). Each location contained a shallow (≤10m bottom depth) and deep (≥15m bottom depth) site. Coastal landmarks on Curieuse Island were used to locate the correct deployment location. Individual deployment sites were chosen based on depth requirements and seabed topography, with flat substrates containing sand being ideal for deployment.

Survey Method

The Seychelles Fishing Authority (SFA) provided two Horizontal BRUV (H-BRUV) Stereo Camera Frames (*SeaGIS*) for use during this study. These were modified to be mounted with a single wide angle video camera (*SJCam SJ4000*; 170° field of view, HD1080p) (Figure 7). A bait arm held an aluminium canister containing bait (bonito (*Euthynnus affinis*) or bigeye mackerel (*Selar crumenophthalmus*)). BRUV units were lowered from a boat to the seabed at each deployment site. Care was taken to avoid rocky and/or uneven terrain due to the risk of damaging coral or entangling the BRUV unit. The following additional information was recorded for each deployment: date and time, GPS location (using a *Garmin GPS 73*), weather conditions, turbidity (Secchi disc), and bottom depth (marked and weighted line).

Figure 6. Study area on the northern shore of Curieuse Island showing the four deployment locations. Squares indicate deployment zones of approximately 100x100m; grey squares denote shallow deployment zones (≤10m depth) and dark blue squares denote deep deployment zones (≥15m depth).

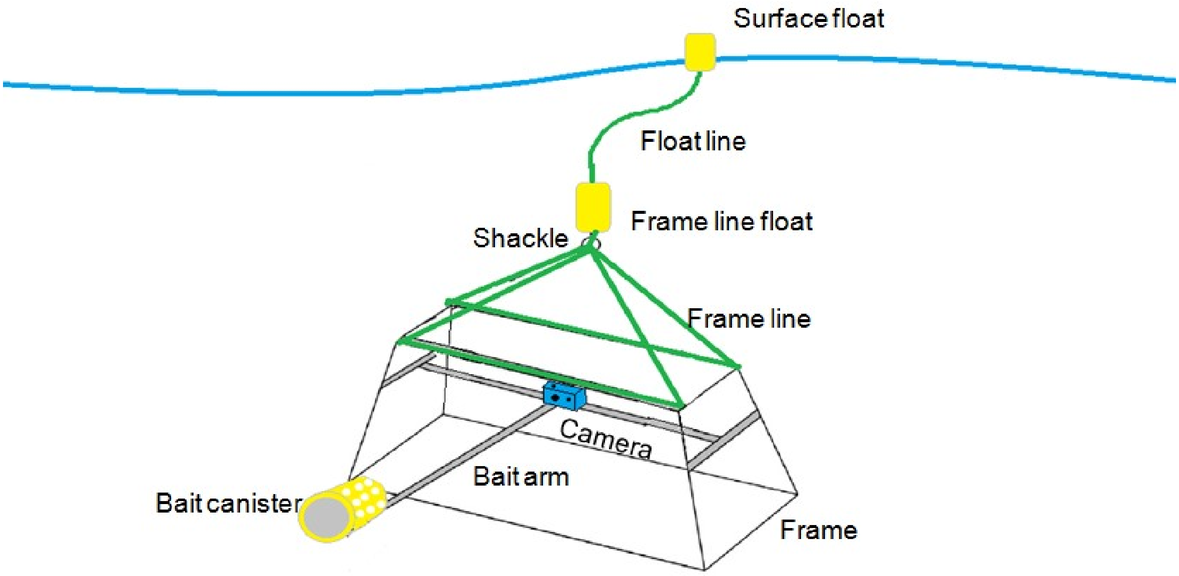

Figure 7. Depiction of a deployed BRUV unit.

Cameras were left to record for 90 minutes without disturbance. Upon retrieval of the units 60 minutes (sometimes less due to battery life) of recording time was analysed. Substrate was categorised as either: sand, rock, rubble, or sand/rock. Relative abundance estimates were made by recording, per deployment, the maximum count of target species entering the field of view at one time, (MaxN). This method was used to avoid repeated counts of the same species, providing a conservative estimate of relative abundance (Willis et al. 2000). Species diversity at each site was calculated using the Shannon-Weiner Diversity Index (Shannon 1949):

$$H = -1^* \sum [(pi) * \ln(pi)]$$

where∑= Summation, pi= MaxN of species x in sample a/sum of MaxN in sample a; MaxN used as a proxy for abundance.

Statistical analysis of diversity index by site was conducted using t-test. Target species (Table 5) selected for monitoring included: 1) predatory and scavenger fish that would be attracted to the bait canister, 2) species of economic significance to Seychelles, and 3) butterfly fish, included due to the simplicity of positive identification and their utility as indicators of changes in conditions on coral reefs. Species of shark (orders *Orectolobiformes* and *Carcharhiniforme*), eel (family *Muraenidae*), and stingray (family *Dasyatididae*) were added to the target species list as encountered. A number of target fishes were recorded only to genus (unicornfish, soldierfish and squirrelfish) or family (porcupinefish, barracuda, shark sucker and moorish idol). Fishes recorded to genus or family were treated as species in data analysis. When species identification (of target families) was not certain (due to distance from camera and/or low visibility), they were not recorded or used in analysis. June-December 2017 BRUV data is not included in analysis because of: 1) low replication number (only two per site), 2) parameters of shallow deployments were not strictly followed, 3) video analysis was based on all species, rather than selected target species, and 4) deployments were conducted June - December rather than April - March. However, 2017 BRUV results will be referenced in the discussion.

Results

General

Four replicates at both shallow and deep sites were completed at all four locations between the 27th of March 2018 and the 10th of May 2018; 32 BRUV deployments total. A total of 103 target species were selected for monitoring and 71 were positively identified (Table 5). Water visibility averaged 10.4m (6 – 16 m; ±3.2 SD) and video recordings averaged 58.7 minutes (45-60 min ±2.8SD). Shallow

BRUV deployments were executed between 7-10m while deep deployments were between 15-24m. Deployment substrates in shallow areas encompassed all substrate categories: rock, sand/rock, sand and rubble; substrates in deeper deployment sites consisted solely of sand and rubble.

Table 5. Target families and species monitored for in BRUVS, 2018. Species encountered and used in analysis are highlighted in yellow.

Butterflyfish species	***Chaetodontidae***	**Emperor Species**	***Lethrinidae***	**Rabbitfish species**	***Siganidae***	**Grouper Species**	***Serranidae***
Vagabond butterflyfish	*Chaetodon vagabundus*	Redfin emperor	*Monotaxis heterodon*	Blackeye rabbitfish	*Siganus puelloides*	Slender grouper	*Anyperodon leucogrammicus*
Threadfin butterflyfish	*Chaetodon auriga*	Large-eye emperor	*Monotaxis grandoculis*	Coral rabbitfish	*Siganus corallinus*	Peacock grouper	*Cephalopholis argus*
Chevroned butterflyfish	*Chaetodon trifascialis*	Blue lined large-eye emperor	*Gymnocranium grandoculis*	Honeycomb rabbitfish	*Siganus stellatus*	Blackfin grouper	*Cephalopholis nigripinnis*
Black-backed butterflyfish	*Chaetodon melannotus*	Longnosed emperor	*Lethrinus olivaceus*	Forktail rabbitfish	*Siganus argenteus*	Coral hind grouper	*Cephalopholis miniata*
Merten's butterflyfish	*Chaetodon mertensii*	Blue-scaled emperor	*Lethrinus nebulosus*	African whitespotted rabbitfish	*Siganus sutor*	Tomato grouper	*Cephalopholis sonnerati*
Indian redfin butterflyfish	*Chaetodon trifasciatus*	Redear emperor	*Lethrinus rubrioperculatus*	**Snapper species**	***Lutjanidae***	Leopard grouper	*Cephalopholis leopardus*
Indian ocean teardrop butterflyfish	*Chaetodon interruptus*	Yellowlip emperor	*Lethrinus xanthochilus*	Paddletail snapper	*Lutjanus gibbus*	Honeycomb grouper	*Epinephelus merra*
Bennett's butterflyfish	*Chaetodon bennetti*	Thumbprint emperor	*Lethrinus harak*	Red emperor snapper	*Lutjanus sebae*	Foursaddle grouper	*Epinephelus spilotoceps*
Raccoon butterflyfish	*Chaetodon lunula*	Pinkear emperor	*Lethrinus lentjan*	Longspot snapper	*Lutjanus fulviflamma*	Camouflage grouper	*Epinephelus polyphekadion*
Klein's butterflyfish	*Chaetodon kleinii*	Orange-striped emperor	*Lethrinus obsoletus*	Blue-lined snapper	*Lutjanus kasmira*	Whitespotted grouper	*Epinephelus caeruleopunctatus*
Speckled butterflyfish	*Chaetodon citrinellus*	Yellowfin emperor	*Lethrinus erythracanthus*	Bengal snapper	*Lutjanus bengalensis*	Brown-marbled grouper	*Epinephelus fuscoguttatus*
Spotted butterflyfish	*Chaetodon guttatissimus*	Small-tooth emperor	*Lethrinus microdon*	Onespot snapper	*Lutjanus monostigma*	Potato grouper	*Epinephelus tukula*
Lined butterflyfish	*Chaetodon lineolatus*	Gold spotted emperor	*Gnathodentex aurolineatus*	Brownstripe snapper	*Lutjanus vitta*	Blacktip grouper	*Epinephelus fasciatus*
Saddleback butterflyfish	*Chaetodon falcula*	Snubnose emperor	*Lethrinus borbonicus*	Flametail snapper	*Lutjanus fulvus*	Blue yellow grouper	*Epinephelus flavocaeruleus*
Meyer's butterflyfish	*Chaetodon meyeri*	Mahsena emperor	*Lethrinus mahsena*	Mangrove jack snapper	*Lutjanus argentimaculatus*	Speckled grouper	*Epinephelus ongus*
Yellow-headed butterflyfish	*Chaetodon xanthocephalus*	**Triggerfish species**	***Balistidae***	Red snapper	*Lutjanus bohar*	White blotched grouper	*Epinephelus multinotatus*
Zanzibar butterflyfish	*Chaetodon zanzibarensis*	Titan triggerfish	*Balistoides viridescens*	Russell's snapper	*Lutjanus russelli*	Redmouth grouper	*Aethaloperca rogaa*
Longnose butterflyfish	*Forcipiger flavissimus*	Flagtail triggerfish	*Sufflamen chrysopterum*	Black snapper	*Macolor niger*	Yellow-edged lyretail grouper	*Variola louti*
Angelfish species	***Pomacanthidae***	Bridled triggerfish	*Sufflamen fraenatum*	Green jobfish snapper	*Aprion virescens*	Long spined grouper	*Epinephelus longispinis*
Three-spot angelfish	*Apolemichthys trimaculatus*	Black Triggerfish	*Melichthys niger*	**Eel Species**	***Muraenidae***	Saddleback grouper	*Plectropomus laevis*
Emperor angelfish	*Pomacanthus imperator*	**Octopus Species**	***Octopodidae***	Giant Moray	*Gymnothorax javanicus*	African coral cod grouper	*Plectropomus punctatus*
Regal angelfish	*Pygoplites diacanthus*	Common Reef Octopus	*Octopus cyanea*	Yellowmargin Moray	*Gymnothorax flavimarginatus*	**Sweetlip species**	***Haemulidae***
Semicircle angelfish	*Pomacanthus semicirculatus*	**Puffer species**	***Tetraodontidae***	**Fish recorded to family**		Oriental sweetlips	*Plectorhinchus vittatus*
Stingray Species	***Dasyatididae***	Silver Puffer	*Lagocephalus sceleratus*	Porcupinefish	*Diodontidae*	Spotted sweetlips	*Plectorhinchus picus*
Marbled ray	*Taeniurops meyeni*	Map Puffer	*Arothron mappa*	Barracuda	*Sphyraenidae*	Silver Sweetlips	*Diagramma pictum*
Feathertail ray	*Pastinachus sephen*	Star Puffer	*Arothron stellatus*	Shark sucker	*Echeneidae*	Gibbus sweetlips	*Plectorhinchus gibbosus*
Wrasse species	***Labridae***	**Requiem Sharks**	***Carcharhinidae***	Moorish idol	*Zanclidae*	**Guitarfishes Species**	***Rhinobatidae***
Tripletail wrasse	*Cheilinus trilobatus*	White tip reef shark	*Triaenodon obesus*	**Fish recorded to genus**		White-spotted Guitarfish	*Rhynchobatus djiddensis*
Redbreasted wrasse	*Cheilinus fasciatus*	**Nurse Sharks**	***Ginglymostomatidae***	Unicornfish	*Acanthuridae Naso*		
Cheeklined splendour wrasse	*Oxycheilinus digramma*	Tawny Nurse Shark	*Nebrius ferrugineus*	Soldierfish	*Holocentridae Myripristis*		
				Squirrelfish	*Holocentridae Sargocentron*		

Relative abundance

MaxN values of species pooled over 22 categories across all sites indicate that emperor species were the most commonly observed (pooled MaxN=188); with the redfin emperor (*Monotaxis heterodon*) having the highest relative abundance (pooled MaxN=56, composing 30% of emperor sightings). Snapper species followed in relative abundance (pooled MaxN=94), with the red snapper (*Lutjanus bohar*) seen most commonly (pooled MaxN=47, 50% of snapper sightings). Shark sucker species were seen the third most commonly (pooled MaxN=86), rabbitfish (forktail rabbitfish (*Siganus argenteus)* seen most commonly*)*, butterflyfish (threadfin butterflyfish (*Chaetodon auriga)* seen most commonly*)*, grouper (peacock grouper (*Cephalopholis argus*) seen most commonly) and triggerfish (flagtail triggerfish (*Sufflamen chrysopterum*) seen most commonly) species were moderately common (pooled MaxN=70, 59, 56 and 32 respectively), while all other species categories were observed relatively infrequently (MaxN <25) (Figure 8). Two species of shark were recorded, *Triaenodon obesus* and *Nebrius ferrugineus,* and averaged 0.125 sharks/hr. Emperor species showed high relative abundance across all substrate types, with sand/rock having the highest, and sand the lowest. Snapper species showed the highest relative abundance over rock and the lowest over sand. Shark suckers showed significantly higher abundance over sand substrates, and the lowest over rock substrates. Rabbitfish, butterflyfish and grouper all showed the highest abundance over rock and sand/rock substrate, and the lowest abundance over sand substrate. Triggerfish showed the highest abundance over rock and rubble, and the lowest abundance over sand (Figure 9).

Species Diversity

A clear species diversity trend related to both deployment site and substrate type was evident. Shallow sites had significantly higher average species diversity than deep sites (p-value ≤ 0.05) (Figure 10). All four locations had higher average species diversity at their shallow sites; PR's shallow site having the highest average species diversity than any other site (H=2.8). Conversely, species diversity was lower at deep deployment sites at all four locations; AB's deep site having the lowest average species diversity (H=0.64). Overall, average species diversity did not show sizeable differences between locations: AB, FB, PR and WP with diversity indices of 1.28, 1.69, 1.75 and 1.47 respectively (Figure 11). Rock substrate had the highest species diversity (H=2.7), followed by sand/rock (H=2.2), rubble (H=1.5) and sand substrate had the lowest (H=0.82) (Figure 12).

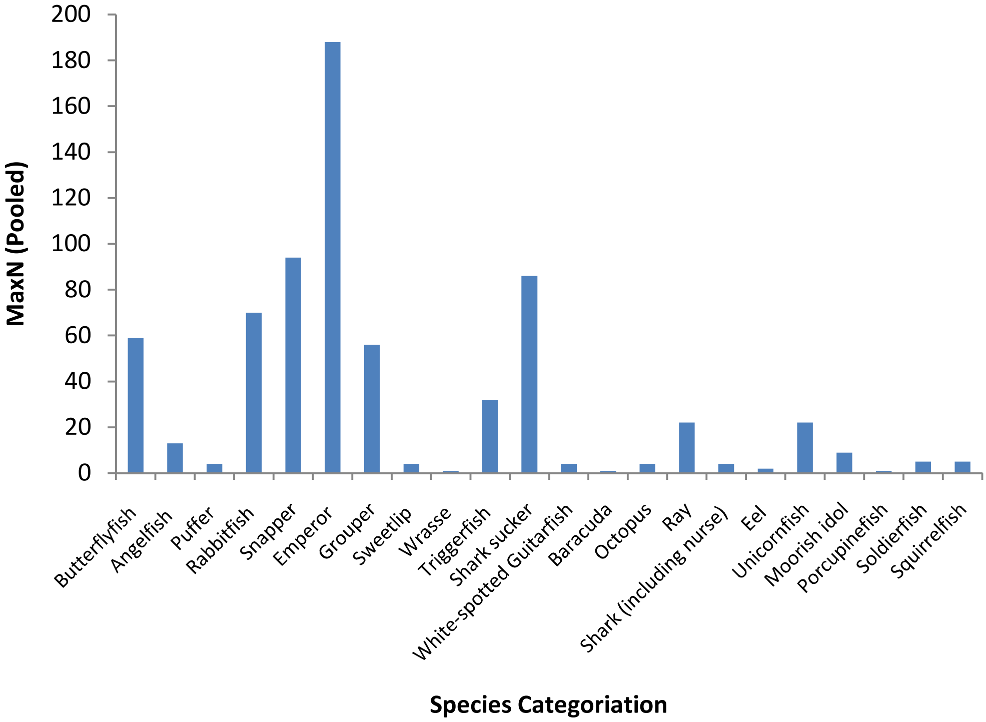

Figure 8. Relative abundance (MaxN) of species pooled over species category and BRUV deployments sites conducted along the northern shore of Curieuse Island.

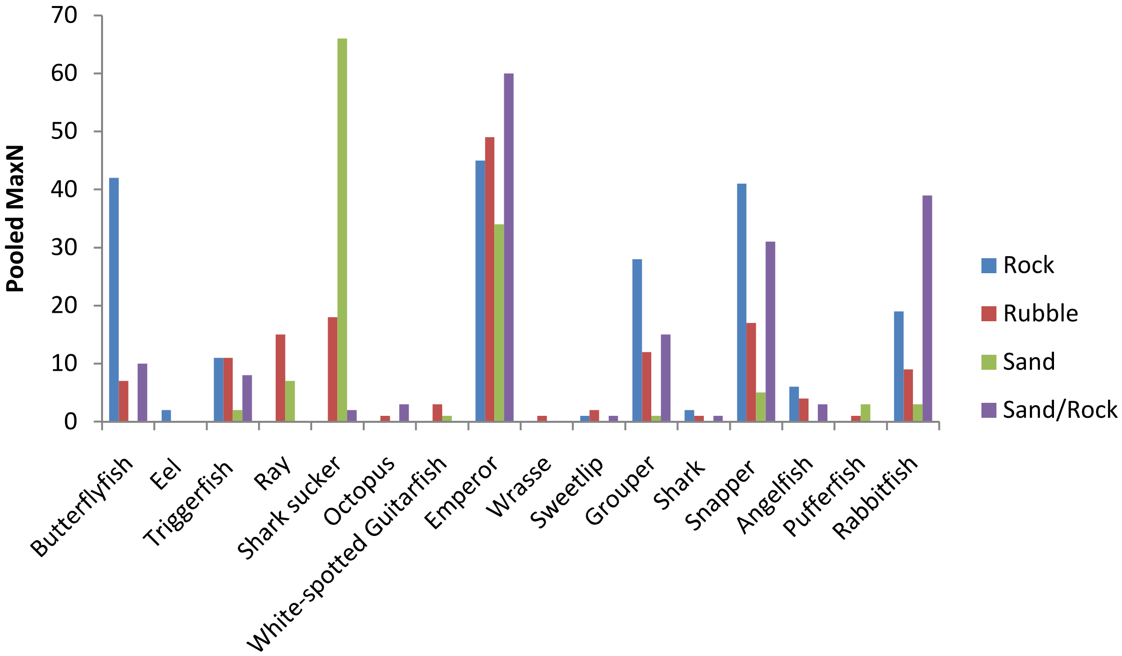

Figure 9. Relative abundance (MaxN) of species pooled over species category and BRUV substrate type.

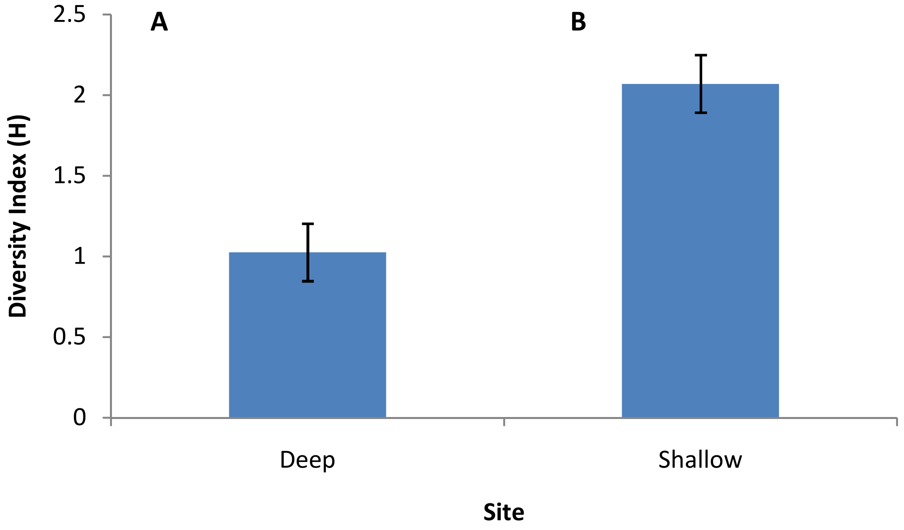

Figure 10. Shannon Weiner diversity index (H) by site (shallow and deep) (±SEM).

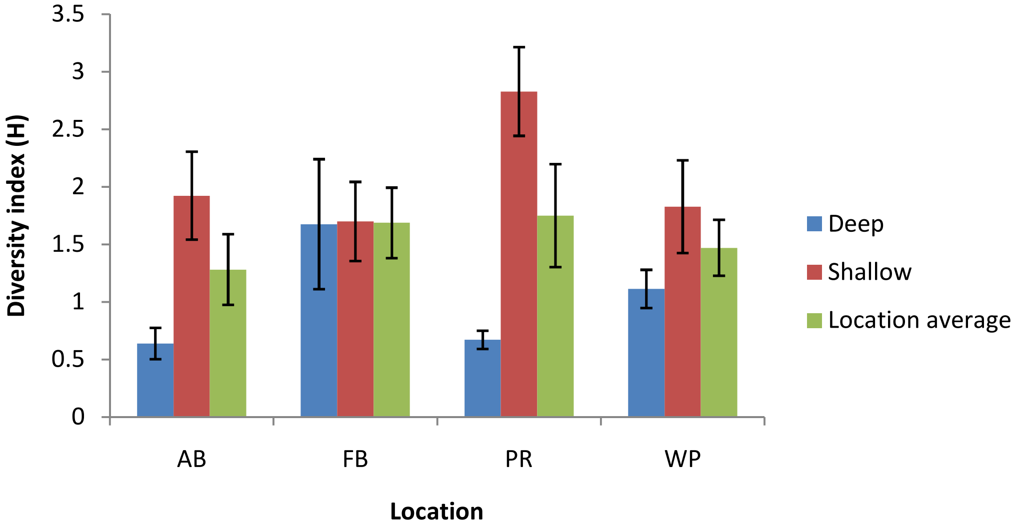

Figure 11. Average Shannon-Weiner Diversity Index (H) (±SEM) by location. Included are average shallow (red bars) and deep (blue bars) deployment sites as well as the overall average diversity index by location (green bars).

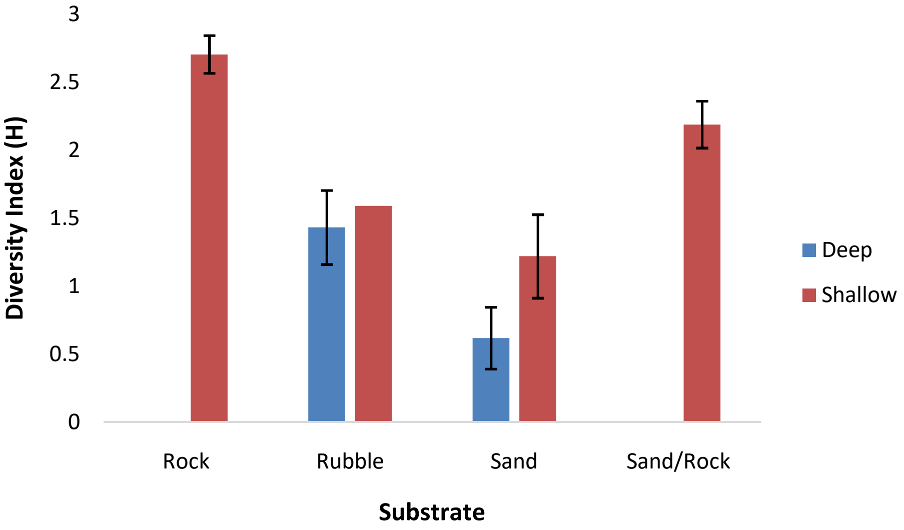

Figure 12. Average Shannon-Weiner Diversity Index (H) by substrate and depth (±SEM).

Discussion

The primary aim of this BRUV study was to establish baseline data on the diversity and relative abundance of target species within CMNP, which can be used to track changes in their populations over time. This aim was achieved through the collection and analysis of data on relative abundance and species diversity from four locations along the northern shoreline of CMNP. BRUV monitoring should aid in informing management actions related to the preservation of target populations through adaptive management. Results obtained from this BRUV season will serve as the baseline data for moving forward with monitoring in future seasons, with the overall aim of incorporating BRUVs to assess the effectiveness of management action in the preservation of target populations within CMNP.

Although the project experienced some initial difficulty related to bait acquisition and storage, and manoeuvring surveys around weather/sea conditions, a satisfactory number of replicates/site were successfully collected. A trivial number of deployment videos were under 60 minutes, which should have no significant effect on results; De Vos et al. 2014, Gladstone et al. 2012, Misa et al. 2016, reported no significant differences in precision for MaxN from analysing as little as 15-49 minutes in their BRUV studies.

Initial findings show that emperor species (family *Lethrinidae*) were by far the most abundant target species, which was also seen in the 2017 BRUV study. Emperor, snapper, grouper and rabbitfish species have all composed a consistent and significant proportion of landings by artisanal fishermen in Seychelles (Daw et al. 2011 and Seychelles fishing authority 2014); in combination with observing these species in elevated abundance in this study, it can be suggested that CMNP promotes the overall Seychelles population stock of emperor, snapper, grouper and rabbitfish species, and helps ensure species population health and sustainable fisheries outside of CMNP. However, the lack of elasmobranchs is cause for concern as only two species were recorded at 0.125 sharks/hr at CMNP. In comparison, five species were recorded at 0.5 sharks/hr at North Island (Green Island Foundation 2015) and six species were found at 6.73 sharks/hr at Aldabra (Clarke et al. 2012).This may indicate elevated fishing pressure within Seychelles' inner islands. Therefore, the continued monitoring of these species, and other target species, in the context of adaptive management is therefore critical in ensuring that the closure of fisheries within CMNP effectively serves the purpose of preserving source populations for both larval and adult fish spill-over outside of the MPA.

Shallower sites in this study had the highest level of target species diversity; likely because 69% of shallow deployments contained the two highest diversity substrates: rock and sand/rock, and 100% of deep sites were composed of the two lowest diversity substrates: rubble and sand. Although species diversity by depth was significant, the confounding factor of substrate type needs to be addressed. Two parameters are known to explain a large proportion of variability in species diversity and abundance: depth and habitat, with species abundance and diversity typically increasing with rugosity and decreasing with depth (Friedlander et. al 1998). It would therefore be inappropriate to discuss BRUV 2018 diversity results solely by depth, while disregarding substrate, or vice versa. With this in mind results from the 2018 BRUV surveys can effectively compare species diversity over sand substrate by depth; results showing decreased species diversity over sand substrate by depth. This could support the conclusion of a lack of complex habitat (rock substrate) in neighbouring areas at deep sites because if in presence, would provide shelter to species that could potentially travel into nearby sand substrate and increase species diversity. Comparisons of species diversity over rubble, rock and sand/rock by depth cannot be made due to limited, or no deployment replications completed. Nevertheless, results still strongly support the continued protection of waters surrounding Curieuse, especially protection of shallow areas with rock substrates.

All deployments were stratified random samples of the northern shore based off four locations (PR, WP, AB, FB), and in combination with low species diversity over sand substrates (as mentioned above) it can be suggested that there is a correlation between the overall lack of rock and sand/rock substrates at the study's four deep location sites and the low diversity. In conjunction with the four locations having similar average species diversity, elimination of deployment location restrictions is recommended, and instead randomly sampling the entirety of CMNP's northern shore separated into deep and shallow areas. This should increase chances of replicating substrate type at shallow and deep sites to effectively compare species diversity and abundance based on known indicators (depth and substrate). However, the resulting lack of rock substrates at deep sites may indicate that the entirety of deep areas off CNMP's northern shore may lack rock-based complex habitats. If this is the case it may prove difficult to find replications of desired substrate at depth due to requirements of specialised equipment to guarantee substrate type at time of deployment. Nevertheless, attaining an understanding of species diversity at deep sites over varying substrates is critical to understanding target species in CMNP and accurately informing management on species diversity status in CMNP.

Continued BRUV deployments are recommended starting from roughly the end of March until May to track annual changes in target species diversity and abundance. It is also recommended to aim for eight replicates of rock, sand/rock, rubble and sand at both shallow and deep sites without location restrictions; totalling 32 BRUV deployments.

Conclusion

Establishing BRUVs as a tool for the assessment of species assemblages at CMNP has continued to be successful thus far with the provision of baseline data on target species abundance and diversity at eight sampling sites along the northern shore of Curieuse Island. A total of 71 species were positively identified out of 103 target species. Species diversity and abundance were highest over shallow, rocky substrates. Future BRUV research should continue to be performed along CMNP's northern shore but instead aim for eight replicates of rock, sand/rock, rubble and sand at both shallow and deep sites without location restrictions. Watson et al. (2005) suggest that BRUVs are very useful in examining a specific impact (such as fishing) on fish assemblages. As such, this tool can be used to guide adaptive management by assessing the impact and effectiveness of management actions such as deterring illegal fishing or marine habitat restoration on fish populations within CMNP.

Beach Profiling

Introduction

Curieuse Island is located on the Seychelles Bank, where coastal plateaus are comprised of calcareous sand accumulated from fringing reefs surrounding the granitic islands (Nentwig et al.

2014). Throughout the year, Curieuse is subjected to changes in wind and wave direction. The Southeast Monsoon occurs between May and September, producing wave energy from that direction, switching to the Northwest Monsoon and resulting northwest wave energy between November and March (Payet & Agricole, 2006). Between the monsoon periods, there are several weeks where wind direction fluctuates and the sea tends to be calmer.

Since GVI became established on Curieuse in 2007, substantial seasonal morphological changes to the beaches have been observed. However, until 2015 there was no continuous quantifiable data collection on these changes. In the past, Seychelles has been impacted by significant events such as the 2004 tsunami (Ramalanjaona, 2011) and Tropical Cyclone Felleng in 2013 (Leister, 2013). The collection of baseline data is therefore vital in our ability to measure the impact any future storm events or changes in sea level may have on Curieuse's beaches.

The beaches of Curieuse Island also provide important nesting habitats for the critically endangered Hawksbill (*Eretmochelys imbricata*) and endangered Green sea turtle (*Chelonia mydas*) (Burt et al. 2014). Having an understanding of the changing morphology of these nesting beaches, particularly during peak Hawksbill nesting season from October to February (Mortimer, 1998), could guide SNPA in future decision making regarding sea level rise, coastal changes, and the management of sea turtle nesting.

Aims

Beach profiling monitors changes in erosion and accretion; this study aims to track these changes on six of Curieuse's beaches. It has been possible to track changes corresponding with the two monsoon seasons, and as this project is now entering its fourth year, long-term fluctuations can be monitored.

Methodology

A total of 18 transects were surveyed on six of the beaches on Curieuse: Anse Caiman (two), Anse Cimitiere (one), Anse St. Jose (six), Anse Laraie (four), Anse Papaie (two), and Grand Anse (three). The number of transects installed on each beach was dependent on the beach length, with longer beaches having more transects. The positions of transects were chosen by SNPA, and currently only beaches located along the eastern and southern coastlines are being surveyed due to time and resource constraints (Figure 13).

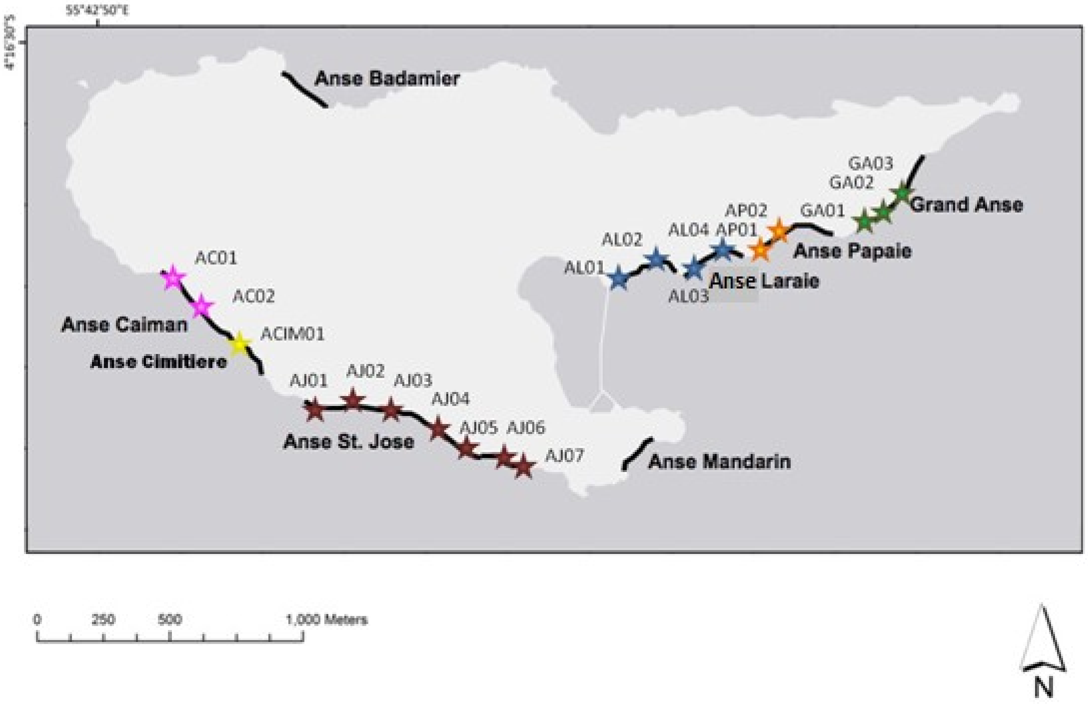

Figure 13. Approximate positions of the beach profiling transects along Anse Caiman, Anse Cimitiere, Anse St. Jose, Anse Laraie, Anse Papaie and Grand Anse in 2018 (note: AJ07 is no longer being monitored).

Individual transects were surveyed once every two months, two hours either side of the lowest tide of the month, as surveying at low tide usually permitted access to the offshore step. The beaches were separated into two groups with Anse Caiman, Anse Cimitiere, and Anse St. Jose all being surveyed in the same month, and Anse Laraie, Anse Papaie, and Grand Anse all surveyed the following month.

Each transect was surveyed by following a set methodology, consistent with previous years. Firstly, the height of the reference mark was measured from the ground to the top of the spray painted line of the mark (all measurements were recorded in metres and measured to the nearest centimetre). Using a compass and the fixed bearing given for that profile, the transect trajectory was established. Robust reference marks for three transects on Anse St. Jose were re-established in the month of July due to an extreme erosion event, which potentially compromised existing reference marks. These re-established reference marks were set further back into the vegetation but retained the same compass bearing so as to continue surveying the original beach profile. The transect was then surveyed in segments using an Abney level and two ranging poles from the reference mark down to the sea. One pole was initially placed by the reference mark with the second pole placed where the terrain changes in slope angle. Then, one person (the same person for an entire transect to ensure consistency) used the Abney level to measure the angle of the slope in degrees and minutes. The

Abney level was held at a comfortable height level at one of several pre-prepared marks on the ranging pole and read by using the corresponding mark on the other pole. The length of the segment was also measured, ensuring this was done once again between two corresponding points on the poles to ensure accuracy, and no segment measured was greater than 10m in length. After this the first pole was moved past the second to be placed at the next point of slope change for the next segment. Once the segment including offshore step had been recorded, one further segment was measured to complete the profile. Once each transect was complete, a photograph was taken of the entire profile (perpendicular to the beach). For each segment surveyed, the angle, horizontal distance, and any obstacles/substrates of interest were recorded (e.g. rocks, logs), and a sketch of the beach profile noting the approximate positions of the ranging poles was also drawn.

All data was entered into the Beach Profile Analysis (Version 3.2) software, which was used to produce profile graphs and provide beach width (m) and area (m^2) measurements. All metadata (e.g. dates, times, survey teams, and comments) was recorded in a separate Excel spreadsheet.

Results

2016 represented the first full year of beach monitoring on Curieuse Island, which continued into 2017 and 2018 with six months of data collected for each of the six studied beaches. Previous reports, e.g. Hodgkiss et al. (2016), analysed each beach transect by transect, which uncovered seasonal trends in erosion and accretion along the length of beaches that are consistent with patterns for the current year. For this report, as in our 2017 report, data is presented by beach, as it provides a more useful output for management. It should be noted that the software has extrapolated length and slope data for some profiles, which could lead to some inaccuracies.

Anse Caiman and Anse Cimitiere

Anse Caiman increased in area from March (23.8m^2) to July (36.1 m^2) (Figure 14A), then decreased into September (22.75 m^2). Area sharply increased (by 55.16% and with high variation) into November (35.3 m^2). Width fluctuated throughout the year (Figure 15A) but slightly peaked at 25.15m in May.

Anse Cimitiere increased in area from January (22.1 m^2) to March (27.6 m^2), then drastically decreased (by 79.71%) into May (5.6 m^2) after an extreme weather event on May 18th (Figure 14B). Area fluctuated for the remainder of the year after an increase from May to July (23.2 m^2, at over 300%). Width varied throughout the year; the highest occurred in July (23.5m) while the lowest occurred in September (16.7m). Anse Caiman and Anse Cimitiere were among the lowest for both beach width and area (along with Anse Papaie) (Figures 14 & 15).

Anse St. Jose

Anse St. Jose slightly decreased in area from January (40.18m²) to May (35.66m²) then increased (by 215%, with high variation) in July (113.24m²). Area then decreased and stabilised for the remainder of the year (Figure 14C). As with area, width showed little variation from January (29.05m) to May (31.25m), and peaked in July (41.60m). Width then stabilised for the remainder of the year (September at 27.75m and November at 32.87m) (Figure 15C). Individual transects on Anse St. Jose showed considerable variation between transects and months.

Anse Laraie

Anse Laraie displayed fluctuations in area with February (46.08 m²), August (41.9 m²) and December (48.23 m²) having the highest values, while April (34.43 m²) had the lowest. The largest decrease in area was from February to April (25%) (Figure 14D). Width increased from February to June (29.1m to 33.3m), stabilized in August (32.15m), and decreased in December (28.9m) (Figure 15D). The greatest decrease in width occurred between August and October (13.06%), and the greatest increase occurred between April and June (12.69%).

Anse Papaie

Anse Papaie showed high variation within each month of beach profiling surveys. Overall, area decreased 15.37% from February (23.1 m²) to April (19.55 m²), then increased 49.9% into June (29.3 m²). Beach area steadily decreased through October (26.2 m²), and then increased to the yearly high in December (29.9 m²) (Figure 14E). Width peaked in June (23.3m) and August (21.4m) and steadily decreased into December (20.5m) (Figure 15E).

Grand Anse

Grand Anse area values decreased 40.4% from February (53.77 m²) to April (32.03 m²) then increased 46.7% from April to June (47.0 m²). After a slight decrease in August (35.6 m²) area values steadily increased into December (46.9 m²) (Figure 14F). Width values remained relatively stable with a maximum in February (29.7m) and a minimum in April (22.13m) (Figure 15F).

Upon comparing mean beach width among all beaches across 2016, 2017 and 2018 (Figure 16), five out of the six survey beaches exhibited an average reduction in width every year; only Anse Caiman has increased (1.33m) since 2016. The greatest erosion since 2016 was observed on Anse Papaie and Grand Anse (losing 8.92m and 13.21m, respectively). Anse Papaie and Grand Anse were also among the top three in variation among transects and over time.

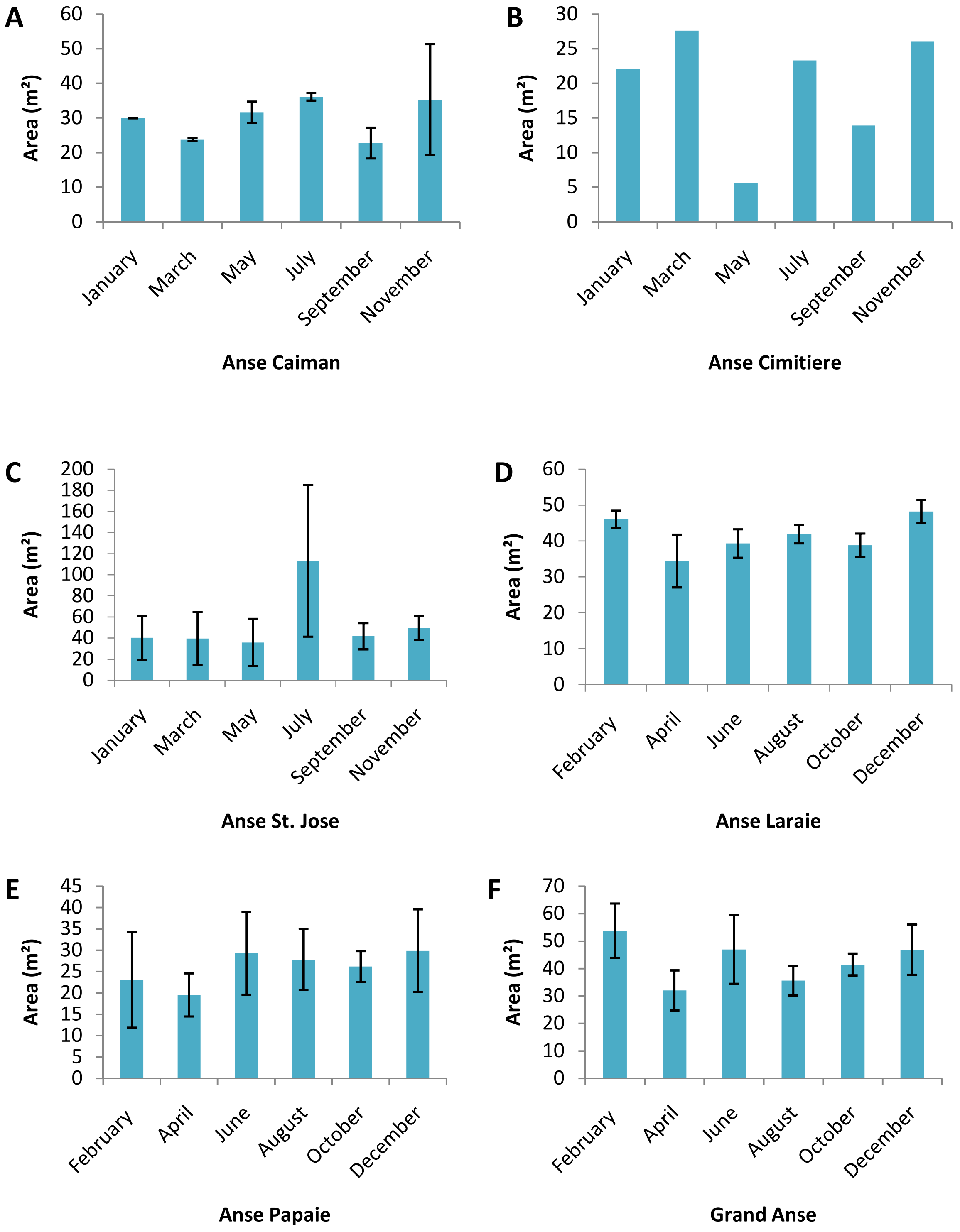

Figure 14. Mean area (m²) over 2018 for A) Anse Caiman, B) Anse Cimitiere, C) Anse St. Jose, D) Anse Laraie, E) Anse Papaie, F) Grand Anse.

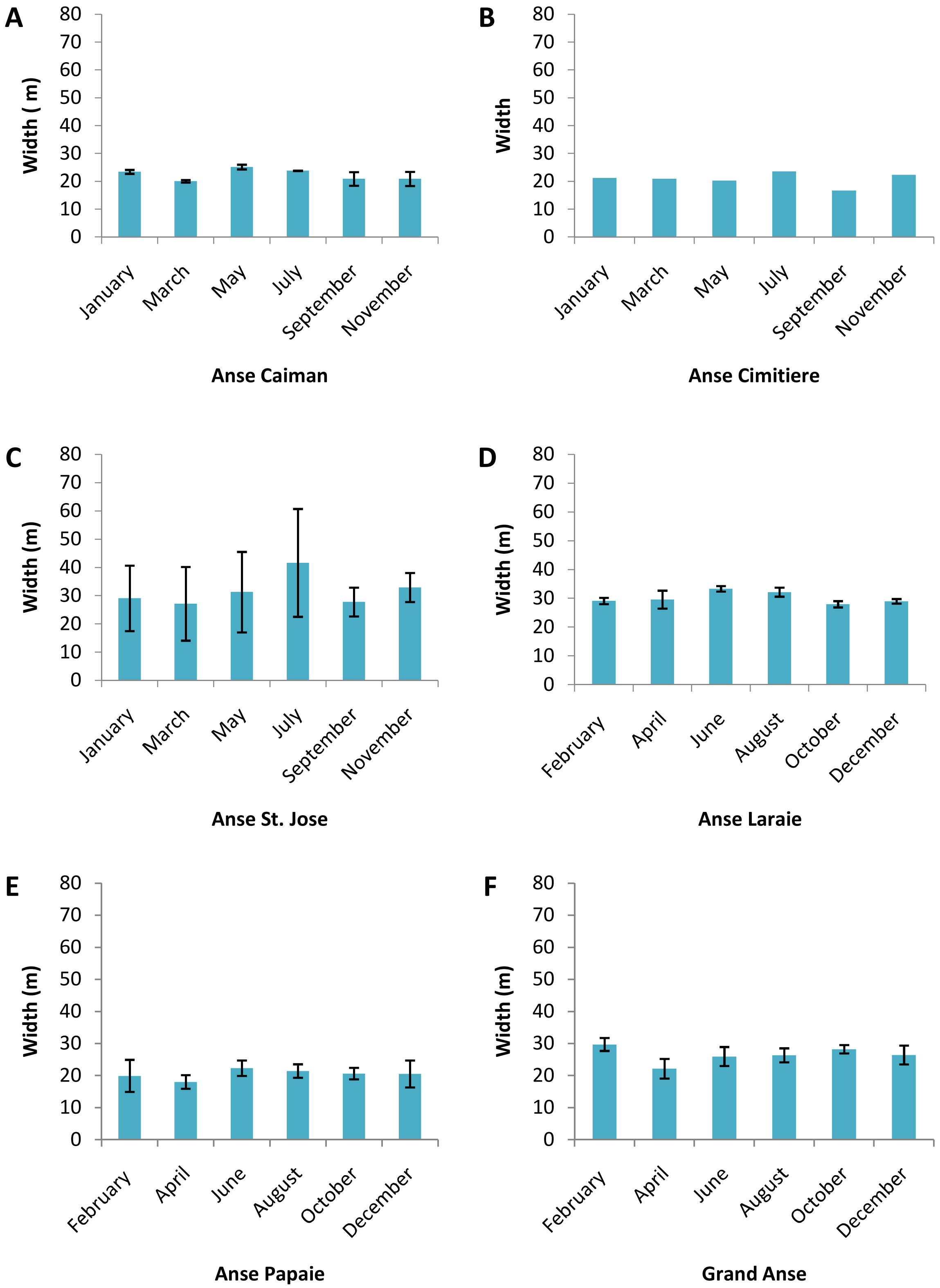

Figure 15. Mean width (m) over 2018 for A) Anse Caiman, B) Anse Cimitiere, C) Anse St. Jose, D) Anse Laraie, E) Anse Papaie, F) Grand Anse.

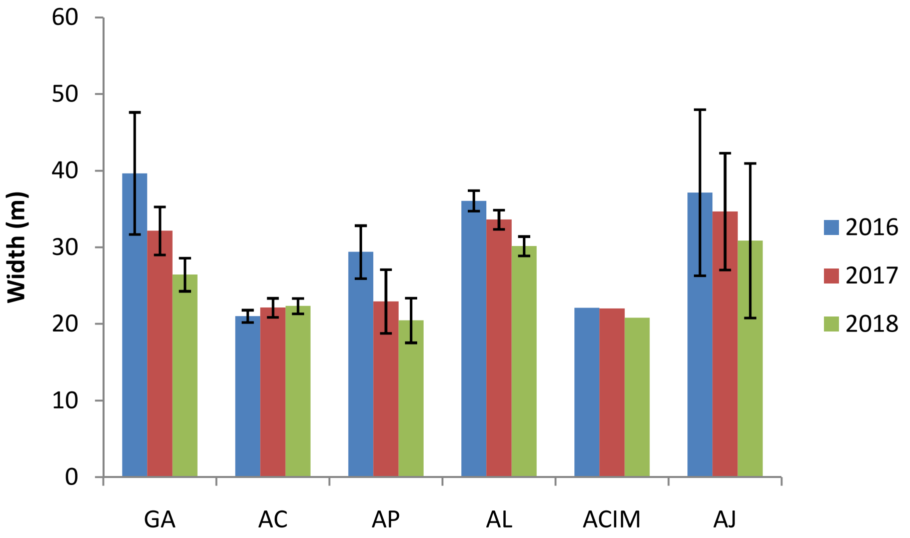

Figure 16. Mean beach width (±SEM) for all beaches over 2016 (blue bars), 2017 (red bars) and 2018 (grey bars).

Mean beach area (Figure 17) followed a similar trend, with Anse Caiman increasing by 4.69m^2 since 2016 while the remaining five beaches decreased in area on average. Anse St. Jose and Grand Anse decreased the most in area by 22.33m^2 and 19.64m^2 respectively since 2016. Anse Papaie and Grand Anse were also among the top three in variation among transects and over time.

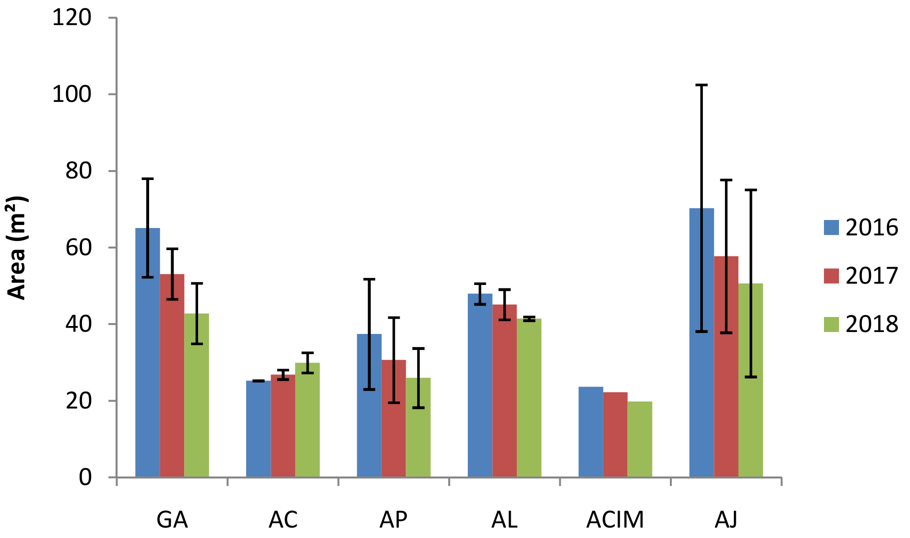

Figure 17. Mean beach area (±SEM) for all beaches over 2016 (blue bars), 2017 (red bars) and 2018 (grey bars).

Discussion

The six beaches studied fluctuated in width and area throughout 2018 due to seasonal variation in the individual transects based on the monsoon season. As highlighted in Kowalski et al. 2017, these seasonal trends appear to be strongest on the beaches along the southern coast of Curieuse (Anse St. Jose, Anse Cimitiere, and Anse Caiman). Here, the movement of sediment with respect to both beach area and width is observed to generally be shifting north-westerly during the Southeast monsoon season, and south-easterly when the wind direction changes in the Northwest monsoon. This was observed to a lesser extent with those beaches oriented in a more westerly/easterly direction (Anse Laraie, Anse Papaie, and Grand Anse). These sediment shifts coinciding with the monsoon seasons, first recorded in 2015, 2016, and 2017, were again observed in 2018. For this reason, beach transects were combined by beach for analysis in this report, as was done for the 2017 annual report. This way, a more concise overview of inter-annual changes to and variation in beach width and area can be provided, which should be more practical for informing the management of CMNP's beaches.

With the exception of Anse Caiman, there was a general downward trend in both area and width between 2016, 2017 and 2018. Although all six beaches have shown increases and decreases in width and area annually (Figures 2 and 3), trends observed from 2016- 2018 (Figures 4 and 5) give rise to concern. The largest decreases in width and area have occurred on the primary turtle nesting beaches on Curieuse, and if this is to become an annual trend with further reductions in width and area, it may lead to a reduction in available sea turtle nesting habitat. However, the large amount of variation present across transects in these beaches suggests that this data should be approached with caution. The extrapolation of beach width and area from a small number of transects provides only an estimate of trends for an entire beach, and these should be assessed over a greater period of time in order to assess actual changes.

Storm events can cause considerable beach erosion and therefore significantly influence beach profiles (Morton, 2002). This may have the potential to introduce bias in this study, as changes to beach profiles can change rapidly following storm events due to extreme waves and surge (Bird 1996). It would be prudent to capture these events to gain a greater understanding of the effects of erosion (such as on nesting sea turtles), what is causing erosion and when exactly it is occurring. At present, the beach profiling conducted by GVI is strictly scheduled and takes place every month, usually over the span of two days, and timed where possible to the lowest tides of the month. Storm events are often unpredictable and adding additional beach profiling surveys is generally not logistically possible at this time. However, dates of storm events were recorded in order to understand effects on beach profiles. On the 18th of May 2018 a significant storm event came through Seychelles and Curieuse suffered extensive erosion. As this analysis looks at overall beach averages for width and area, this storm event was only visible on Anse Cimitiere's area analysis, due to being the only beach with one transect. Nevertheless, Anse Cimitiere bounced back in area when the Southeast monsoon commenced. Still, it is crucial to monitor these erosion events as they can replace soil with sand, and although the beaches may be staying stable in area and width, it could be that overall beach positions are moving inward. As such, sediment condition at the reference marks should be described, and continued beach profile data should be collected long-term in order to assess trends in beach width and area over time.

Moreover, sediment budget, defined as the balance between the sediment gains and losses within a specified control area over a given time, requires the identification of sediment sources (sediment inputs, such as from land erosion/river outputs/coral reef erosion, etc.) and sinks (a point/area where beach sediment is irretrievably lost from the system such as estuaries, sand dunes or deep seabed channels) of a defined system (Rosati 2005). The rate of sediment movement must also be estimated, and unpredictable variables can make it difficult to obtain accurate sediment budget estimates (Rosati 2005). Ideally, we would be able to calculate the sediment budget for Curieuse Island in order to better understand the effect of the coastal processes on the island, and whether they are cycling annually as would be expected or whether each year more sediment is being lost than gained. If such an imbalance were to be discovered, then future management plans to preserve these valuable sea turtle nesting beaches would need to be explored and implemented to preserve the viability of the island as a sea turtle rookery. To properly investigate sediment budgets on Curieuse would require a much more detailed analysis of the island and its littoral sediment movements, which GVI is not in a position to conduct at this time.

Another major contributing factor in determining the shape of a beach profile is sediment characteristics, including grain size, sorting, and distribution (Hanson, 2016). It has been suggested that collecting data on these factors would provide greater insight into the reason behind some of the changes in beach profiles being seen, and increase the usefulness of the conclusions being drawn. Other major influencing factors affecting beach profiles are wave climate (wave length/period/height) and wave generated currents (e.g. longshore drift), tides, and the strength of swash and backwash (Hanson, 2016). However, collecting this type of data is currently beyond the capabilities of our project. This type of initiative would require further collaboration with SNPA and possibly other organisations with further training, equipment, and staff.

Several issues have been noted with the current computer program being used (Beach Profile Analysis (Version 3.2)), one of these being that the program extrapolates the data from the end of the profile inputted. However, depending on the data entered this is only done for some transects and not for others. This appears to be having some effect on results, and at this stage data should still be treated with caution. Some initial research has begun into possible alternative computer programs, and it is hoped in the next year of this project other programs may be trialled. The current program provides limited ways in which profiles can be viewed, a problem which could be addressed if a new program was available to GVI that is compatible with GIS software. The ability to view changes in our beaches with GIS imagery could provide decision makers with a much clearer view of the processes that have occurred in the past and those to come in the future.

Conclusion

Completion of three full years of beach profiling data has provided a baseline to assess trends in the width and area of beaches on Curieuse Island over the coming years. At this stage, it appears five of the six study beaches have decreased in width and area between 2016 and 2018; however, a high level of variation is present in the data. Nevertheless, these noticeable decreases in width and area highlight the necessity to assess long-term seasonal and inter-annual changes in order to determine if any intervention is necessary. Planned improvements to the methodology and analysis will also support stronger conclusions regarding the observed trends in beach profile island-wide. Trends observed thus far demonstrate that a project such as this is worth running at Curieuse Marine National Park in order to guide future management of its vitally important sea turtle nesting beaches.

Coco de Mer

Introduction

Coco de Mer (*Lodoicea maldivica*) is a species of palm endemic to Seychelles, which produces the largest seed pod of any plant on earth. This is a classic example of island gigantism, and the Coco de Mer (CDM) also holds records for leaf length, fruit size and weight, and the largest female flowers of any palm (Edwards et al. 2003, summarised in Fleischer-Dogley 2006). The CDM population on Curieuse represents one of three major global populations of this species, with the other two being found on neighbouring Praslin Island at Fond Ferdinand (FF) and the Vallée de Mai (VDM), a UNESCO World Heritage site. An iconic species of Seychelles, the CDM is also classified as endangered according to IUCN criteria (Fleischer-Dogley et al. 2011a), which highlights the importance of studying the species on Curieuse. As an iconic species for Seychelles, this palm provides significant revenue through tourism as well as through the harvest and sale of seed pods.

The CDM seed pods (commonly referred to as ‘nuts’) are popular tourist souvenirs, as their suggestive shape provides considerable appeal. Nuts are harvested legally on Praslin and Curieuse in numbers thought to be sustainable. Processed, non-viable nuts are sold by licensed vendors for £150-£400, and are sold with an individual identification stamp and card to verify their origin and legality. The Seychelles government exercises strict control over the nuts in order to protect the genetic heritage of the islands, and it is illegal to collect or sell unlicensed and/or viable nuts. However, CDM nuts are in high demand on the black market, and poaching represents a significant concern for all extant populations. These trees have particular life history traits including late age at reproduction (20-40 years to reach sexual maturity) and long development period of nuts (six to seven years) (Edwards et al. 2003), which would result in difficulty for population recovery following reduction through intensive nut collection, fire, or disease.

The CDM has been the subject of numerous studies; however, few researchers have investigated the Curieuse population long-term. The first major study of the Curieuse population was conducted by GVI Seychelles and SNPA, consisting of a census taken of the island’s CDM population (2009-2014) over five years. The census produced a population count and basic life stage information (Sanchez et al 2015; Dunn et al 2014). However, relatively little was known regarding the growth rate of this species and how long it takes for them to transition between life stages on Curieuse.

Koch and Kaiser-Bunbury (2010) conducted a CDM growth rate study on Praslin, though CDM trees on Curieuse exhibit distinct differences compared to those on Praslin (Fleischer-Dogley et al. 2011b), and results from studies of Praslin’s populations are not necessarily applicable to the Curieuse population. In order to harmonise data collection between the three populations, GVI Seychelles initiated a long-term growth rate study following the same methodology as Koch and Kaiser-Bunbury (2010) in April 2014. The initial monitoring phase highlighted variation between Curieuse and Praslin (Sanchez et al. 2015), albeit with only limited growth data as the project was in its early stages. Monitoring has continued throughout 2018, with the 57 month study period having produced more informative data.

Aims

The primary goals of the growth survey were to assess survivorship, by documenting the time spent by CDM at each life stage, and to determine leaf and trunk growth rates. The difference between life stages and populations can then be compared. Additionally, assessing inflorescence production among male and female trees allows for the assessment of variation in reproductive output between populations. Understanding the life stages, growth, and reproductive output of trees in this study improves our understanding of this population, informing adaptive management and ultimately resulting in improved protection for this unique population of endangered plant.

Methodology

A total of 75 trees (15 of each life stage - male, female, immature, juvenile, and seedling) were selected in the hillside area west of Baie Laraie (Figure 18). Seedlings represent young plants possessing three or fewer leaves, while juveniles possess more than three leaves but no trunk, and immature trees possess more than three leaves and a visible trunk or defined crown base. Adult trees possess trunks and produce sexual characteristics (female inflorescences or male catkins). The methodology and life stage classification used in this study mirrors that used on CDM trees studied at VDM (see Koch and Kaiser-Bunbury 2010).

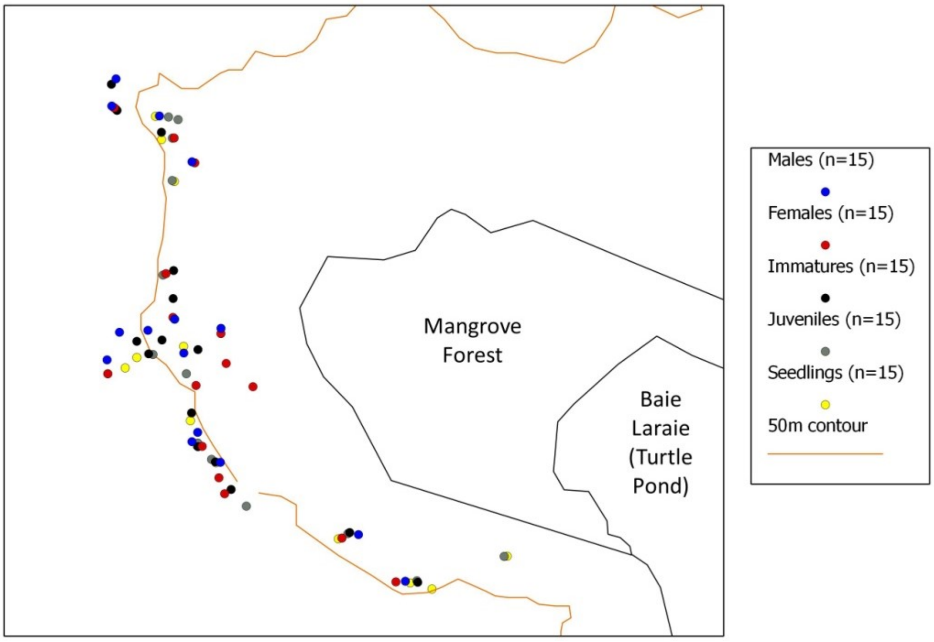

Figure 18. Tree distribution for the Coco de Mer growth survey by life stage and sex.

Trees were selected for this study based on tree clusters; each cluster had a mixture of life stages. Another influence on tree selection was its accessibility to researchers. Each tree was given a unique code, and their exact position recorded using a GPS.

Between the onset of the study in April 2014, and June 2016, each tree was measured every monitoring period (approximately every three months), as per the methodology used at VDM. However, beginning in July 2016, the monitoring period was extended to approximately every six months due to the slower growth rates exhibited by the Curieuse population when compared to the population at VDM. Since trees were not measured exactly every six months to the day, growth was calculated as a daily growth rate. Therefore, extending the duration of the monitoring period will not affect the usability of and inferences made from the data collected. In addition, following analysis of growth rate estimates from the first two years of the study, it was determined that a six month study period was sufficient for measuring growth in the individual components of the plant, with leaf growth being the critical measurement in that regard.

During the initial setup, the three youngest (most central) leaves of each tree were identified and labelled L1 – L3 according to age (oldest-youngest respectively). The total length of each leaf was recorded, and a mark was painted at 40cm above the base of the leaf. Referred to as ‘Mark A’, this mark was re-measured on each visit to determine leaf growth. The total length of bayonets (early stage leaves growing from the centre of the crown that have not yet opened into full leaves) was also recorded at each visit. Once open, a bayonet was then considered a leaf and was set up using the aforementioned technique and given a leaf code (sequential from the previous youngest leaf (e.g. L4, L5, etc). Measurements for a specific leaf were discontinued when it stopped growing, i.e. when no change was seen in leaf length after three visits (Koch and Kaiser-Bunbury 2010).

Additional data was collected for each tree, including the number of green leaves, trunk height and Girth at Breast Height (GBH) of the trunk. During setup, or once they reached an appropriate size, immature and adult trees were marked with paint 10cm below the swell at the most accessible point (paint to swell), and the GBH of the trunk was marked and measured at a height of 150cm above the ground level. With the exception of trunk height, all these additional measurements were re-taken on subsequent visits to determine growth.

Reproductive information was also collected for adult trees. The number of flowering catkins (male trees), and the number and classification (primary, maturing, or ripe) of nuts per inflorescence and number of inflorescences (female trees) were also recorded.

Results

To date, 13 monitoring periods have been completed (most recently, period 13 was completed in December 2018). From the study's inception, periods were completed on a quarterly basis (mean 103.4days ±35SD), with variation in period duration being largely due to fluctuations in volunteer numbers and available manpower.

Some issues have been identified with the methodology, and work has been done to remedy issues with the existing data set. For example, a review of the existing data has highlighted a number of measurements that indicate minor negative growth in leaves, trunk height, and GBH. For the purpose of this report, these values have largely been excluded from analysis, and the data set is currently being standardised over the longer study periods (six month intervals) in order to avoid inherent human error in the measurement of small changes in size over short time periods. To assess annual changes in leaf growth and reproductive output, trees in this analysis were analysed for changes during periods 11 (11/07/2017 to 27/12/2017) and 13 (25/07/2018 to 22/12/2018).

In period 13, female CDM exhibited taller trunks (321.4±99SD) but with a smaller GBH (86.5±11.6SD) than males (trunk height: 189.6±83.8SD; girth: 88.5±5.8SD) (Table 6). Throughout the study, immature plants exhibited the smallest mean trunk height (159.9±64.8SD) and GBH (83.1±11.28SD). For P13 only one immature tree had its GBH successfully measured. Although all measured classes exhibited an annual increase in height, only a slight increase in GBH was detected in male and immature trees, while females displayed an overall reduction in GBH.

Females exhibited a larger average annual increase in trunk height (7cm) than males (2 cm) and immature trees.

Immature trees showed the longest average leaf length (466.2±145.0SD) followed by juveniles (452.7±173.2SD), males (404.4±139.8SD), females (381.5±115.7SD), then seedlings (213.3±97.0SD) (Figure 19). With regard to leaf growth rate, leaves of female trees grew fastest (0.208cm/day ±0.33SD), followed by males at nearly the same rate (0.207cm/day ±0.34 SD), juveniles (0.19cm/day ±0.29 SD), immatures (0.15 cm/day ±0.28 SD) and seedlings (0.06 cm/day ±0.08 SD).

Table 6. Means ±SD of growth parameters for male, female, immature, juvenile, and seedling CDM trees between period 11 (P11) and period 13 (P13) (encompassing 2018). Total leaf length includes bayonets.

	P11	**P13**	**Annual Change**
Males			
Trunk height (cm)	187.4±83.3	189.6±83.8	+2.0
GBH (cm)	87.6±7.0	88.5±5.8	+0.9
Total leaf length (cm)	380.0±108.3	404.4±139.8	+24.4
Green leaves per tree	13.9±1.9	13.53±2.13	-0.3
Females			
Trunk height (cm)	314.2±98.2	321.4±99	+7.0
GBH (cm)	87.2±11.28	86.5±11.6	-0.8
Total leaf length (cm)	374.7±102.2	381.5±115.7	+6.8
Green leaves per tree	15.1±3.2	16.1±2.94	+1
Immatures			
Trunk height (cm)	157.1±68.4	159.9±64.8	+2.0
GBH (cm)	82.5±11.3	83.1	+0.6
Total leaf length (cm)	440.6±147.5	466.2±145.0	+25.6
Green leaves per tree	10.6±2.6	11.07±2.87	+0.4
Juveniles			
Total leaf length (cm)	438.2±186.0	452.7±173.2	+14.5
Green leaves per tree	5.6±2.1	5.8±2.04	+0.2
Seedlings			
Total leaf length (cm)	214.9±97.4	213.3±97.0	-1.59
Green leaves per tree	2.5±0.9	2.13±0.74	-0.3

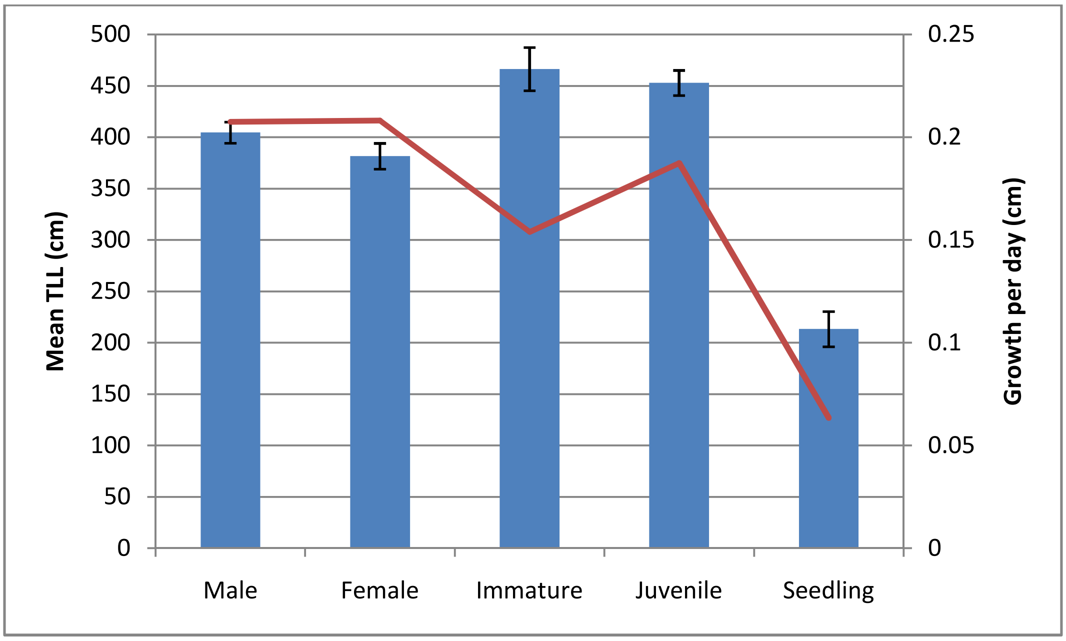

Figure 19. Mean total leaf length (mean ±SD; blue bars) and mean growth rate (red line) for the various life stages and genders of CDM trees for periods 11 to 13 (encompassing 2018 and including bayonets).

Overall, relatively little variation existed in the number of green leaves per tree over 2018, while certain individuals appeared more productive than others. On average, females produced the most leaves (16.1±2.94SD), followed by males (13.53 ±2.13 SD), immatures (11.07 ±2.87 SD), juveniles (5.8 ±2.04SD) and seedlings (2.13±0.74SD) (Table 6). Male inflorescence production (number of flowering catkins) did not exhibit any clear trends. The maximum number of flowering catkins recorded on a single tree was two and the minimum was zero (Table 7). Period 11 represents the lowest recorded period of catkin production in this study.

Table7. Mean ±SD and maximum number of flowering catkins recorded per tree from initial setup (April 2014) until the end of P13 (December 2018).

	Setup	**P1**	**P2**	**P3**	**P4**	**P5**	**P6**	**P7**	**P8**	**P9**	**P10**	**P11**	**P12**	**P13**
Number of catkins	0.87± 0.64	0.60± 0.63	0.67± 0.72	0.87± 0.64	0.87± 0.74	0.43± 0.51	0.33± 0.49	0.7± 0.46	0.76± 0.73	0.67± 0.49	0.53± 0.64	0.20± 0.41	0.6±0 .63	0.63± 0.63
Maximum	2.00	2.00	2.00	2.00	2.00	1.00	1.00	1.00	2.00	1.00	2.00	1.00	2.00	2.00

Female nut production appears to be increasing over time; however, an unforeseen nut harvesting event of the study trees during P12 is clearly visible in Figure 20. In P13 there already seems to be some recovery from the harvesting event (1.47±1.807SD over 2018).

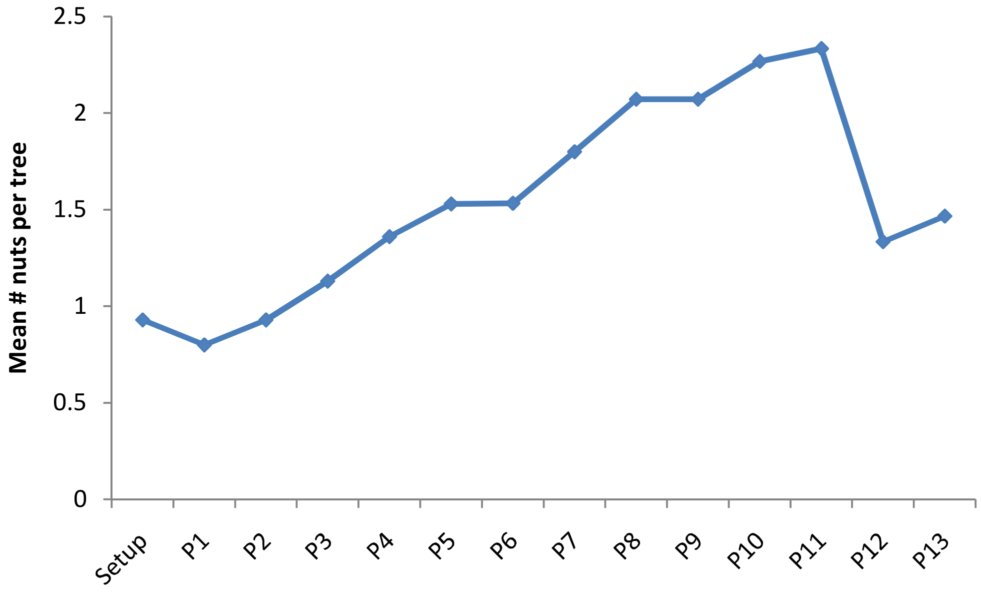

Figure 20. Mean number of nuts per CDM tree from initial setup (April 2014) until the end of P13 (December 2018).

Discussion

A comparison of Curieuse data against the two main Praslin populations, Fond Ferdinand and Vallée de Mai, suggests distinct differences (summarised in Sanchez et al. 2015), which can be attributed to varying environmental conditions among sites and the high phenotypic plasticity of *L. maldivica* (Fleischer-Dogley et al. 2011).

Growth trends over 2018 appear to be consistent with 2017 data, but different from previous years, in that immature trees did not exhibit the fastest mean leaf growth rates among the life stages in the study area. Instead, the two adult classes of tree experienced similarly high mean leaf growth rates of 0.21 cm/day. Overall, mean growth in all tree classes decreased in 2018, both in comparison to 2017 and previous years.

A number of factors could result in the observed decrease in leaf growth rates by adult trees, particularly females. Firstly, the average number of nuts per tree had been steadily increasing since the onset of this study until the unexpected harvesting in P12; in fact, the mean number of nuts per tree had more than doubled since early 2014. Moreover, Dunn et al. (2014) observed an even lower mean number of nuts per tree (0.797, N=546) during the island-wide CDM census, which preceded the current growth study. Silverton (1987) suggested that the energy expended in seed production could explain the reduced size of female trees on Praslin, in comparison to males. As such, this observed increase in reproductive output on Curieuse is potentially reducing available energy for leaf growth. Although mean nut production on Curieuse is far lower than that of females in FF & VDM (8.86 and 6.38 nuts per tree, respectively (Fleisher-Dogley 2006)), leaf growth could also be limited by a combination of increased nut production coupled with environmental factors such as the relatively poor soil conditions on Curieuse. With the unexpected harvesting of two of the study trees, it could be of interest to determine if harvested trees begin to display an increase in leaf growth rates, giving us a better understanding of how nut production may influence leaf growth.

Soil and other environmental variables may also be negatively influencing the growth of CDM at all life stages at present. Rainfall patterns can have a particularly strong impact on plant growth, and while Curieuse experienced a higher amount of precipitation (20%) this year compared to 2017, precipitation levels have still decreased by 19% compared to 2015. Leaf growth could thereby be water-limited by such trends. Moreover, a number of tropical tree species are known to decrease in trunk diameter during drought conditions, which could explain the reduction in GBH across all studied classes in 2017, and why females still showed a reduction in 2018 (Baker et al. 2002; Nath et al. 2006).

Hodgkiss et al. (2016) highlighted a correlation between trunk height and nut production, suggesting that as trees grow and age, greater numbers of nuts per tree should be expected. At FF and VDM on Praslin, female trunk height was 7.79m and 9.26m respectively (Fleisher-Dogley 2006) compared to approximately 3.5m on Curieuse, and if tree growth continued over time, females on Curieuse could potentially increase nut production to higher levels than at present. However, it is suggested that nut production be assessed at a greater scale with a sample size larger than 15 female trees in a variety of CDM habitats throughout Curieuse Island. A more robust study of nut production would also have the added benefit of being able to inform the sustainable harvest of CDM nuts within CMNP.

As noted by Hodgkiss et al. (2016), there is no strong correlation to monthly rainfall in the number or frequency of flowering catkins produced by male trees. Moreover, no monthly or seasonal variation is evident either. While flowering catkins may be recorded on some trees, the number of male trees surveyed in one month, for example, can often be less than five. In order to truly assess the influence of rainfall patterns on reproduction, the sample size or sampling frequency would need to be increased to detect any patterns in male or female reproduction with any level of certainty.

To date, only one tree has been observed to change life stage, developing from immature to male with the production of its first recorded catkin. However, this tree had a trunk height of approximately 250cm, which is taller than the male mean, and it produced its first catkin following the initial monitoring phase. Therefore, it is possible that this individual either did not have a catkin present or was incorrectly identified during setup.

It is also suggested that the trees in this study are not completely representative of the range of individuals and environmental conditions on the island. For example, Hodgkiss et al. (2016) noted that there are much larger and presumably more productive trees on Curieuse than those sampled in this study, which has the potential to result in bias. This is particularly evident in the fact that the trees studied were selected in part due to their accessibility for researchers lacking advanced climbing equipment and specialised training in tree climbing. Therefore, while this study provides an important comparison against other CDM populations as well as insight regarding the baseline biology of this species on Curieuse, care should be taken when making generalisations regarding this population based on only 75 smaller trees in a relatively small area out of a population of over 6000 individuals over the entire island.

Conclusion

Following nearly five years of study, a strong set of baseline data regarding growth, leaf size, trunk size, and reproduction has now been established by this project, providing a basis for comparison well into the future. A clear and promising trend is becoming evident in female nut production, and this information, alongside growth patterns, is critical to the effective management of the species on Curieuse. However, this information should be taken with caution due to only 15 female trees being assessed. Harvesting of CDM nuts was conducted on Curieuse this year, and a more robust study regarding island-wide nut production with a larger sample size should aid Seychelles National Parks Authority in determining a sustainable level of harvest. As such, it may be extremely useful to CMNP to recruit interns and/or rangers to intensively collect further information regarding nut and catkin production. Overall, it is important to continue studying this population of Coco de Mer to ensure that this rare and endangered plant receives appropriate management considerations based on the best available science.

Mangroves

Introduction

Seven species of mangrove are present in Seychelles, six of which were once present on Curieuse Island (SNPA 2012). Currently there are five species, along with a mangrove associate species, found within the national park. Mangrove ecosystems play an important role in ensuring a high level of water quality and clarity, and are essential for adjacent corals to thrive by trapping sedimentation and land run-off. Mangroves also represent vital nurseries for fish and crustaceans, and provide an important habitat for birds, algae, and bryozoans. The mangrove ecosystem also supplies essential nutrients for marine creatures such as fish and shrimp. Additionally, it represents a crucial buffer zone for protecting inland areas from high wave action and events such as tsunamis (Lewis 2005; Yoshihiro et al. 2002).

The mangrove forest on Curieuse is of particular interest. In 1910, a causeway wall was built at Baie Laraie in a failed endeavour to rear sea turtles. The wall had a lasting impact on the bay, as it reduced wave intensity, providing a suitable environment for mangrove seedlings to settle and grow. In December 2004, a tsunami damaged the wall, allowing larger waves to enter the bay more frequently, causing an influx of sediment. This is currently altering the mangrove population structure by decreasing abundance and species richness (SNPA 2012).

Previous studies were completed in an effort to determine mangrove distribution patterns in relation to temperature, hydrology and salinity, and tree growth rates within the mangrove forest (refer to previous GVI Annual Reports).

Aims

The primary aim of the survey is to provide baseline data to facilitate decision-making regarding the health of the mangrove forest and placement of mangrove nurseries in the near future. Current surveys assess mangrove diversity and abundance, as well as mortality and recruitment rates.

Methodology

Five 10m x 10m permanent quadrats were set up in June 2015 in various locations throughout the mangroves. The locations of these quadrats were chosen by SNPA and all lie within the seaward half of the mangrove forest (Figure 21). In January 2017, three additional 10m x 10m permanent quadrats were established close to the seaward edge of the forest. The abundance and growth rate of mangrove species within these quadrats were measured biannually. Within each 10m x 10m quadrat are four 1m x 1m quadrats positioned at each corner.

Figure 21. Location of mangrove quadrats 1-8

The total number of mangrove trees (>1m high; >4cm Girth at Breast Height (GBH)) and their species were recorded within each 10m x 10m quadrat. Within each 1m x 1m quadrat, all species of mangrove seedlings (<1m high), saplings (>1m high; <4cm GBH), and trees were counted. All mangrove trees within each 1m x 1m quadrats also had their GBH measured, which was set at 130cm from trunk base during the initial survey or beneath the first stem if the trunk was less than 130cm. When no seedlings, saplings or trees were present inside of the 1m x 1m quadrat, the species of roots present were recorded, or a lack of mangroves was noted.

Results

Results from surveys completed in February and August 2017 and 2018 have been included in this report.

10x10 Quadrats

Results from the 10m x 10m quadrats indicate that *Rhizophora mucronata* was the dominant species in all quadrats, except Quadrat 3 where *Avicennia marina* was most abundant (Figure 22). Between 2017 and 2018, *R. mucronata* exhibited changes in average annual abundance; Quadrats 1, 2, 5, and

6 showed an increase, while Quadrats 4 and 7 showed a decrease. There was no change in average annual abundance of *R. mucronata* in Quadrats 3 and 8. However, there was an increase in the *R. mucronata* average annual tree count within Quadrat 2 between 2017 and 2018, rising from 21 to 57 trees. *B. gymnorrhiza* exhibited a decrease in average annual abundance between 2017 and 2018 in Quadrats 1 and 7, and an increase within Quadrat 4. Within Quadrats 3, 5, and 6 there was no change in average annual abundance of *B. gymnorrhiza*. Between 2017 and 2018 *A. marina* exhibited no change in average annual abundance in Quadrats 1 and 3, but decreased within Quadrat 2.

None of the quadrats contained *Xylocarpus* trees, and only Quadrat 1 contained *Lumnitzera racemosa*, exhibiting a decrease in annual average abundance, from 5 to 4, between 2017 and 2018. The second most abundant tree species throughout the quadrats in the most recent survey (August 2018) after *R. mucronata* (n=418) was *B. gymnorrhiza* (n=57), followed by *A. marina* (total n=33). This follows trends found in previous surveys.

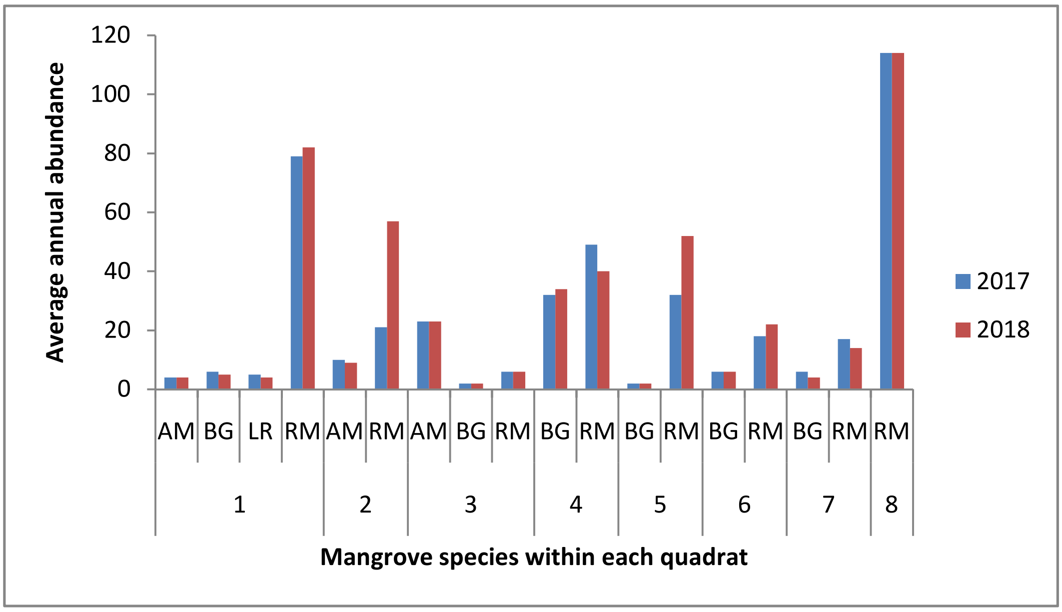

Figure 22. Distribution and average annual abundance for four mangrove species at eight 10x10m quadrat sites within the Baie Laraie mangrove forest during 2017 and 2018; AM= *Avicennia marina*, BG = *Bruguiera gymnorrhiza*, LR= *Lumnitzera racemosa*, RM= *Rhizophora mucronata*.

1x1 Quadrats

R. mucronata, *B. gymnorrhiza* and *A. marina* were the only species to have seedlings and/or saplings present in the 1m x 1m quadrats (Figure 23). Quadrat 2 had the most seedlings averaging 18 and 30 (2017 and 2018 respectively), which is considerably more than the other quadrats. Quadrat 2 also contained the most saplings averaging 22 in both 2017 and 2018. Overall, Quadrat 2 accounted for 61.9% and 63.8% (2017 and 2018 respectively) of all seedlings and saplings. There were no seedlings or saplings present in Quadrat 3.

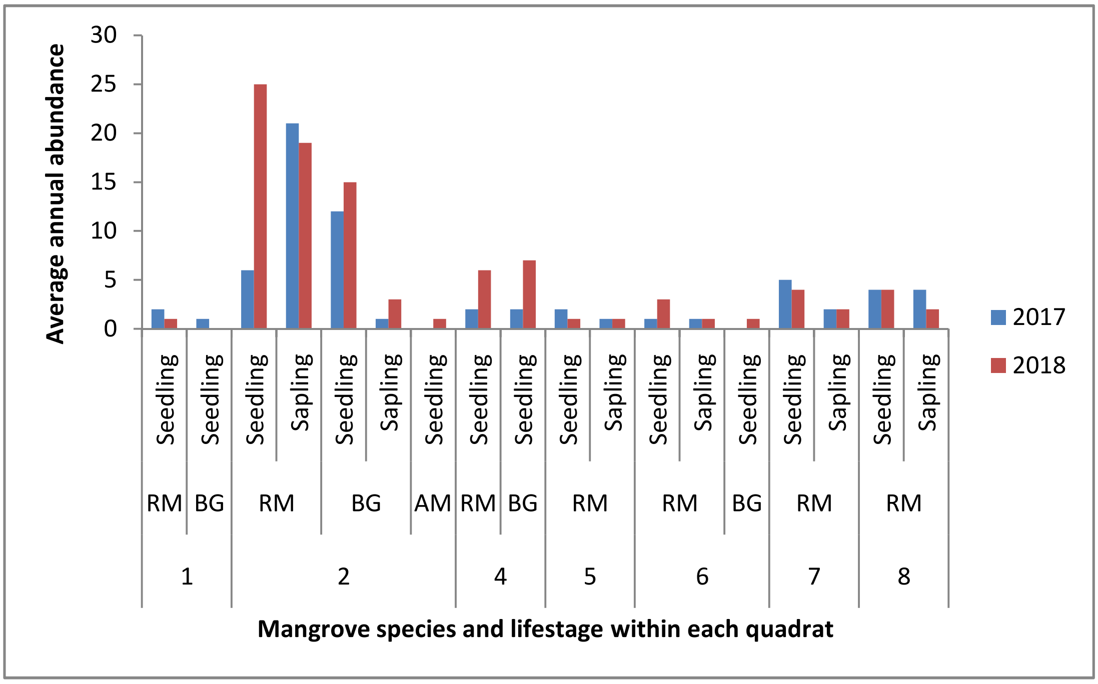

Figure 23. Seedling and sapling distribution and average annual abundance for three mangrove species at eight quadrat sites, containing four 1x1m quadrats, within the Baie Laraie mangrove forest during 2017 and 2018; AM= *Avicennia marina*, BG = *Bruguiera gymnorrhiza*, RM = *Rhizophora mucronata*.

R. mucronata had the most seedlings and saplings, with combined annual averages of 48 and 54 per quadrat site (2017 and 2018 respectively), compared to *B. gymnorrhiza*, with combined annual averages of 15 and 25 (2017 and 2018 respectively) (Table 8). Both *R. mucronata* and *B. gymnorrhiza* exhibited an increase in annual average seedling presence (21 to 31, and 14 to 23 respectively). Annual average sapling presence also increased in *B. gymnorrhiza* from one to three, but decreased in *R. mucronata* from 28 to 24.

None of the 1m x 1m quadrats contained any seedlings, saplings, trees or roots of *A. marina*, *R. mucronata,* or *B. gymnorrhiza*.

Table 8. Average annual abundance of seedlings and saplings of three mangrove species (*R. mucronata, B. gymnorrhiza,* and *A. marina*) at eight quadrat sites, containing four 1x1m quadrats, within the Baie Laraie mangrove forest.

	R. mucronata	***B. gymnorrhiza***	***A. marina***
Seedlings			
2017	20.5	14	0
2018	30.5	22.5	1
Saplings			
2017	27.5	1	0
2018	23.5	2.5	0
Combined Average			
2017	48	15	0
2018	54	25	1

Discussion

Species abundances from the 10m x 10m quadrats were as expected based on the previous 1m x 1m and 3m x 3m quadrat surveys carried out in 2013 and 2014. *R. Mucronata* was the most dominant species in all quadrats, apart from Quadrat 3, where *A. marina* was the most dominant. Quadrat 3 was the most landward quadrat, which is likely to be the reason for this difference in species distribution. The most noticeable increase in the number of *R. mucronata* trees between 2017 and 2018 was recorded in Quadrat 2, which had the highest number of seedlings and saplings. Quadrat 5, which was close in proximity to the seaward edge of the forest, also exhibited a noticeable increase in *R. mucronata* trees. These results may indicate a positive rebound of this species since the increased amount of wave action due to the partial destruction of the seawall.

R. mucronata and *B. gymnorrhiza* being the most abundant species may account for why they also had the highest abundance of seedlings and saplings present throughout the 1m x 1m quadrats. Overall, there has been an increase in *R. mucronata* seedlings and *B. gymnorrhiza* seedlings and saplings, and a decrease in *R. mucronata* saplings between 2017 and 2018. Quadrat 2 had the highest number of seedlings and saplings, for both the 2017 and 2018 surveys, and these were all located in the eastern 1m x 1m plot. It is unclear why the eastern 1m x 1m plot within Quadrat 2 was so productive. However, it is located on a slightly raised sandy bank surrounded by a channel that is often inundated; the elevation of this area may offer a substrate for the seedlings to establish themselves on with less tidal disturbance, and the channel may act as a funnel to direct propagules towards this location. Furthermore, this concentration of seedlings may be self-propagating, as more seeds may become trapped within the stems of the existing bunched seedlings and saplings. A high concentration of juvenile mangroves in one particular area increases the risk of high juvenile mortality rates from threats such as giant tortoise grazing and tree fall. We have thus far been unable to reveal whether there have been any seasonal changes in seedling and sapling mortality rates, though this should become more evident with time. There is insufficient data at this early stage to be able to compare the growth rate of the mangrove trees located within the 1m x 1m quadrats.

Moving forward, while the current positioning of the quadrats allows us to collect consistent data on the mangroves in the seaward half of the forest, they exclude the middle and rear sections of the forest. As a result of this, species such as *X. moluccensis, X. granatum, L. racemosa* and *A. marina* are under-represented. The majority of the seaward edge of the forest is also excluded, which is an area of highest concern as it is where the highest mortality rates have been observed. To undertake future assessments of mortality rates, and understand whether or not this phenomenon has penetrated further into the forest, it is strongly suggested that more permanent quadrats are set up along the seaward edge of the forest, and further landward.

Since the partial destruction of the seawall there have been concerns that the increased wave action and influx of sediment may be resulting in the degradation of the forest. Therefore, establishing a mangrove nursery with the aim of rehabilitating the forest has been a priority for SNPA.

Prior to restorative planting of mangrove habitats it has been recommended that the removal of stress should be considered prior to attempting restoration (Lewis 2005). There have been ongoing discussions about whether or not to rebuild the seawall. When considering the options, it is important to think of the implications that this may have on not only the mangroves, but also on the multitude of species that inhabit this area, including the neonate Sicklefin lemon sharks that use the area as a nursery ground. One of the options would be to not rebuild the seawall and allow the mangrove forest to return to the state it was likely in before the wall was built in 1910. The concern surrounding this option is that it may lead to a decrease in its currently high level of biodiversity.

Another alternative to rebuilding the seawall would be to create natural buffer zones using *R. mucronata*, enhanced sea grass beds, and coral reef restoration. Planting hypocotyls from *R. mucronata* using the Riley's Encasement Method (REM) as outlined by SNPA (2012) could create a natural seawall. REM was developed to facilitate planting where shorelines have high-energy waves and in an effort to overcome the limitations of other mangrove planting schemes (Johnson and Herren 2008). Restoring the buffer zone near the wall with *R. mucronata*, if successful, would restore the hydrology of the mangrove system, which may allow the forest to naturally rebound. Increasing the sea grass cover within the Turtle Pond may also assist in reducing the impacts of wave action and sediment influxes on the mangroves. Studies in Florida have used combinations of Eastern oysters (*Crassostrea virginica*) and Smooth cordgrass (*Spartina altinaflora*) to diffuse and absorb wave energy, thus creating less erosion and sediment intake into coastal habitats (Manis 2013), and perhaps a local seagrass species could be used at Baie Laraie. Conducting coral reef restoration directly beyond the seawall's eastern side could also assist in alleviating wave action on the mangrove forest. With a coral nursery project currently underway on Curieuse Island, it might be prudent to include restoring the reef beyond the seawall in the future.

Conclusion

The current mangrove monitoring activities are part of a long-term project aimed at maintaining the ecological function of the mangrove habitat. The GBH surveys were completed last year while 2015 saw the completion of the salinity, temperature and inundation surveys. These projects have provided four years of data on which to base sound decisions for future rehabilitation plans. With the establishment of the eight permanent 10m x 10m quadrats within the seaward half of the mangrove forest, together with the future plans of establishing more permanent quadrats to cover a greater and more varied area of the forest, these quadrats should provide greater insight into species distribution, abundance, mortality, and recruitment. The data collected thus far has indicated that it is important to also establish permanent quadrats throughout the forest in order to represent all mangrove species present and provide sufficient insight into the changes occurring throughout the entire forest. Continued monitoring is required in order to assess whether the seaward edge of the forest will continue to degrade or whether a natural state of equilibrium has been reached.

The mangrove forest at Baie Laraie is an integral landscape for multiple faunal communities as well as neighbouring ecosystems such as the adjacent sea grass beds and coral reefs. Additionally, the area is heavily visited by tourists and school groups, with many island visitors walking through the mangroves where there are educational signs along the boardwalk. Many tour guides also stop their groups in this area to point out flora and fauna of interest. Moving forward, this high biodiversity area may benefit from the development of natural buffer zones, such as the planting of *R. mucronata* to act as a natural seawall, increasing seagrass cover and carrying out coral reef restoration to help mitigate the impact of increased wave action and sediment movement since the partial destruction of the seawall in 2004. Considering the value that mangrove forests provide in terms of ecosystem services and the potential to improve its state, it is vital that mangrove monitoring continues in order to better understand, protect, and rehabilitate the area.

Sea Turtles

Introduction

Globally important populations of sea turtles can be found within Seychelles, including one of the five largest nesting populations of the critically endangered Hawksbill sea turtle (*Eretmochelys imbricata*) (Mortimer and Donnelly 2008). Green turtles (*Chelonia mydas*) also nest in Seychelles, mainly on Aldabra Atoll, however relatively few utilise the inner granitic islands. The leatherback (*Dermochelys coriacea*), Loggerhead (*Caretta caretta*), and Olive ridley (*Lepidochelys olivacea*) also reside in Seychelles.

The largest Hawksbill populations remaining in the Western Indian Ocean occur in Seychelles, where an estimated average of 1,500 females nested annually in the early 1980s (Mortimer 1984). Since then, populations have declined due to the nearly complete harvest of nesting females from the 1960s to the 1990s (Mortimer 1998), following which a total ban on turtle harvesting was implemented in 1994. An exception to this downward trend was noted at Cousin Island, which has been well protected since 1970. The Cousin population has seen an eight-fold increase in annual nesting numbers in the 20 years to 2010 (Allen et al. 2010). The exploitation of Hawksbill turtles in Seychelles became particularly intense after the mid-1960s with the advent of the mask and snorkel, spear guns, underwater lights, outboard engines, and the high prices paid for raw shell (Mortimer 1984). Mortimer (1984) estimated that 47–71% of the total estimated annual nesting population in the granitic Seychelles islands was killed during the 1980–82 nesting seasons. Although it is now illegal to harvest any species of turtle in Seychelles, a small degree of poaching does still occur. In addition, the destruction of breeding and foraging habitat, especially in the granitic islands, is an increasingly serious problem (Mortimer 1998).

Small numbers of the endangered Green turtle nest on Curieuse (Seminoff 2004, Burt et al. 2015). Green turtles have been heavily exploited for their meat since the 17th century and are now very rare in the inner islands (Mortimer 1984), although there is some evidence to suggest they may have started to recover following the protection of all turtles in Seychelles in 1994 (see discussion).

The waters surrounding Curieuse are home to both Green and Hawksbill turtles, and the reefs and sea grass beds surrounding Curieuse provide ample food resources for both Green and Hawksbill turtles. Beaches also provide a nesting habitat for both species, with Curieuse hosting one of the most important nesting Hawksbill populations in the inner granitic islands (Burt et al. 2015). This alone is enough to highlight the importance of CMNP for sea turtles. Evidence suggests that the number of Hawksbills nesting on Curieuse has increased by as much as 100% since 1984. It should be noted however, that this increase is substantially lower than on several other islands that have benefitted from a much higher level of protection than Curieuse, such as the special reserves Aride and Cousin (Burt et al. 2015).

Hawksbill turtles in Seychelles, and along the east African coast, nest primarily during daylight hours in contrast to populations elsewhere, which tend to nest either strictly or primarily at night (Mortimer and Bresson 1999). Green turtles on the other hand nest primarily at night (Mortimer 1984). Historical data gathered in Seychelles indicates that both Hawksbill and Green turtles can nest during any month of the year. However, Hawksbill turtles show a distinct peak in nesting from October to February (Mortimer 1998).

Aims

Curieuse Island is an important sea turtle nesting rookery in the inner granitic islands of Seychelles. Sea turtle patrols were conducted in an effort to identify the annual nesting female population. Prior to these beach patrols, there were few estimates for the annual number of nesting sea turtles on Curieuse. Another objective of the sea turtle surveys was to measure hatching success rate on each of the nesting beaches through nest excavations. GVI Seychelles aims to continue to monitor nesting beaches and expand on current methodology.

Methodology

Patrols of the main nesting beaches were conducted four to five days a week from October to February (peak Hawksbill nesting season), with a minimum of weekly checks on all other nesting beaches. Outside of Hawksbill season, all beaches were checked at least once a week so that Green turtle nesting (nesting all year round) was sufficiently monitored.

Patrols involved walking along the high tide line and recording any sea turtle activities. For all nesting activity the date, time, beach, and turtle species were recorded. Track width was measured perpendicular to the direction of the track at its widest point. Estimated time of emergence was recorded as 0, 1 or 2, where 0 identified the activity as having been made within the past 12 hours, 1 being 12-24 hours old, and 2 more than 24 hours old. The time of an emergence can be estimated by a) knowing when the last patrol occurred, b) assessing the clarity of the track in the sand, and c) how much of the track has been washed away by the tide. Each track was further classified as one of nine emergence types (Table 9). If multiple attempts at nesting had occurred, the number of attempts was recorded. For every emergence, a GPS waypoint was taken using the code TUN for a nest, and TUA for all other types of emergences. For nests in which eggs were located, the location was triangulated and marked with flagging tape, with the distance from each mark (ΔL, ΔC, and ΔR) recorded. This facilitated nest excavations approximately ten days after the end of the estimated incubation period.

Table 9. Nine categories of sea turtle emergence types.

Half Moon	A. Wandering (but no digging) below high tide line
	B. Wandering (but no digging) above high tide line
	ESBO. Emergence stopped by obstacle
Did Not Lay	C. Considerable disturbance, evidence of digging (body pit & egg chamber) but no covering
	D. Evidence of body pitting, but no digging of egg chamber or covering
Laid	E. Considerable disturbance, evidence of digging and covering
Variations	F. Prob DNL. Probably Did Not Lay
	G. Prob Laid. Probably Laid
	?. Cannot tell if laid or not

When a nesting turtle was encountered on a beach patrol, expedition members followed appropriate behaviour to not disturb the turtle. Observation of the turtle occurred until the egg laying process commenced, at which point the turtle goes into a trance-like state and can be slowly approached from behind. Using a manual click counter, the number of eggs the female laid was tallied and recorded. Once the turtle had finished laying and started covering the eggs, measurements were taken including two over-curve carapace lengths: mid to tip (M-T) and tip to tip (T-T), and the width of the carapace at the widest point, usually across the third vertebral scute (Figure 24).

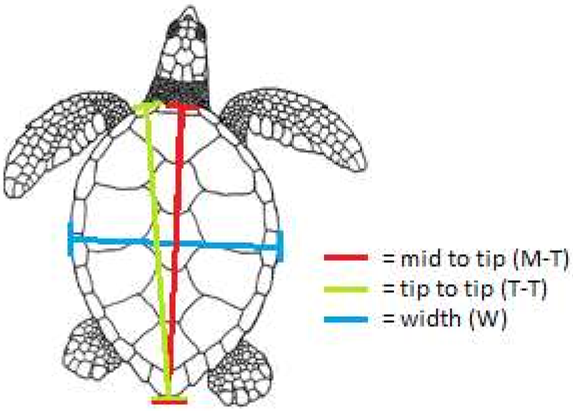

Figure 24. Measurements taken for each turtle encountered.

Each measurement was taken three times to ensure accuracy. Photographs of each side of the head were taken (without a flash) as well as photos of any distinguishing features for future identification in the case of tag loss. Tag numbers (if present), tag scars, evidence of disease/injuries or other distinguishing features were also recorded. If the turtle was untagged, two tags were fitted, one on the posterior edge of each of the front flippers, in the fleshy part just before the first scute. Tagging occurred when the turtle had completed laying and the covering process had begun. Tags administered during both the 2014-2015 and 2015-2016 seasons were 'SCA' series, while mostly 'E' series tags were administered during the 2016-2017 season. Tags administered during the 2017-2018 season were mostly 'SXX' series. The location of the eggs was triangulated as above. Once an activity had been recorded, marks were scored through all tracks so they could be identified as old tracks in subsequent surveys.

Population estimate was calculated by dividing total number of nests by a bracketed mean of 3-4 clutches per female per season for Hawksbills, and 3-5 clutches per female per season for Greens, in line with Burt et al. (2015).

Hatching Success

Success rates were determined by excavating recently hatched nests. Known nests were excavated a minimum of 70 days after triangulation. This allowed ample time for the nest to hatch and hatchlings to emerge.

When nests were excavated, the number of hatched eggs, any pipped (half in, half out of the egg) hatchlings, live or dead hatchlings in the nest, as well as the number of unhatched eggs were recorded. Unhatched eggs were opened and recorded as either undeveloped, stage one, stage two or stage three. Definitions of each excavation category can be found in Table 10. Nest depth was measured before the contents were replaced and reburied. Hatching success rate was calculated by dividing total number of hatched eggs by total number of eggs laid, indicating how many turtles successfully hatched from their eggs. Additionally, emergence success was calculated by subtracting the number of fully developed hatchlings found in the nest, either dead or alive, from the number of hatched eggs and dividing this by the total number of eggs, indicating how many hatchlings successfully emerged from the nest. Often, a small number of live hatchlings were found in the nest; these were released onto the beach to enter the sea otherwise unaided.

Table 10. Nest excavation categories and definitions.

Hatched	Empty eggshells
Live pipped	Hatchling has broken through eggshell but not entirely emerged
Dead pipped	As above, though hatchling is no longer living
Undeveloped	No discernible embryo
Stage one	Discernible embryo; eyes, spine, blood development but mostly yolk
Stage two	Partially developed embryo. Yolk sac is larger than the turtle foetus
Stage three	Mostly developed embryo. Turtle foetus is larger than yolk sac
Predated	Egg obviously consumed by crabs
Predated beyond recognition	Maggot and/or bacterial predation beyond stage recognition *When a small amount of maggots, bacteria or fungus was within an egg and the stage was still recognisable, the numbers of eggs with evidence of predation were accounted for in [ ] Example: Stage one: 5 [2] *5 was the total number of eggs within the stage one category *2 of those eggs contained maggots, fungus and/or bacteria

Results

This report contains a summary of the complete 2017 – 2018 nesting season. The 2019 annual report will contain a summary of the 2018 – 2019 season, once the season has been completed.

Nesting Adults – Hawksbills

The peak of the nesting season occurred in December, with 36% of all activities recorded (Figure 25). The total number of activities for this season was 445, of which 148 were nests, resulting in a population estimate of 37-49 individuals (Table 11). Grand Anse had the highest percentage of laid nests of the 2017 - 2018 season with 70% (Figure 26). The lowest percentage of laid nests was on Anse Laraie with only 1%. These findings follow previous season trends.

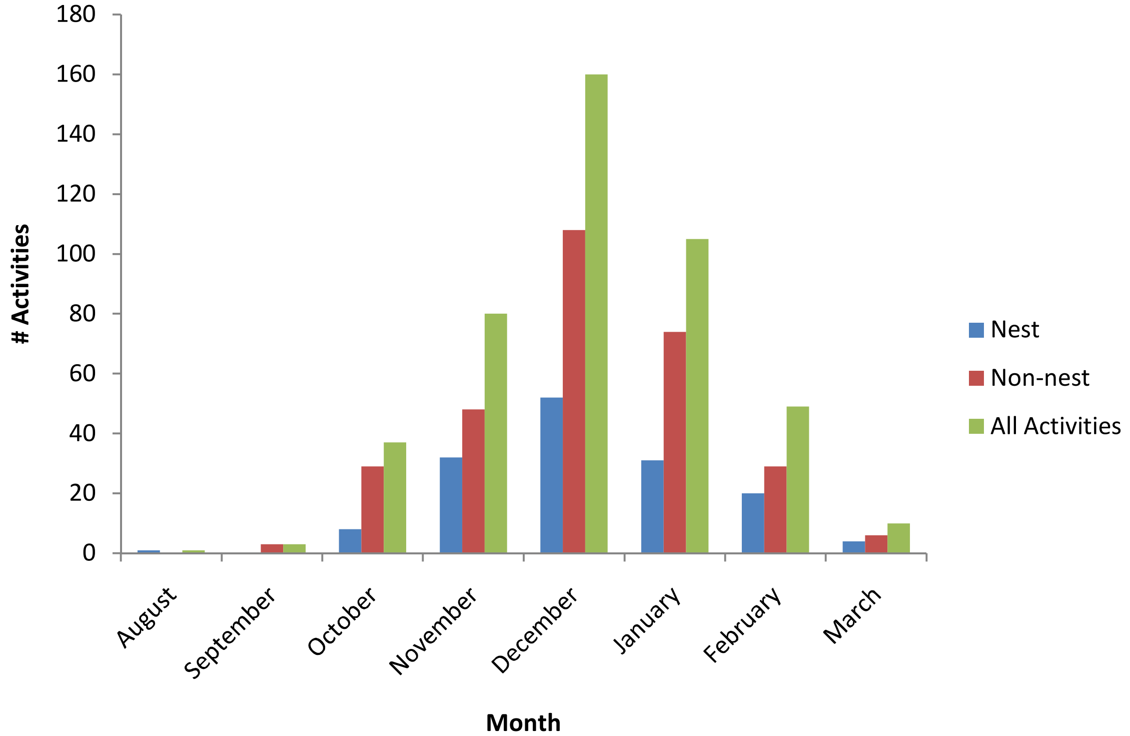

Figure 25. Number of activities by month (nest, non-nest and all activities) for the 2017-2018 Hawksbill season.

Table 11. Total number of activities and nests for Hawksbill and Green turtles recorded for the past eight nesting seasons, 2010-2018, and population size estimates. Numbers in red indicate years where monitoring was not consistent.

	Nesting Season	**2010-2011**	**2011-2012**	**2012-2013**	**2013-2014**	**2014-2015**	**2015-2016**	**2016-2017**	**2017-2018**
Hawksbill	Activities	312	367	522	323	428	596	479	445
	Total Nests	151	186	282	128	225	368	182	148
	Population Estimation	38-50	47-62	71-94	32-43	56-75	92-123	46-61	37-49
Green	Activities	8	14	9	6	53	47	21	56
	Total Nests	0	8	2	4	22	27	10	21
	Population Estimation	1-2	1-2	1-2	1-2	5-7	5-9	2-3	4-7

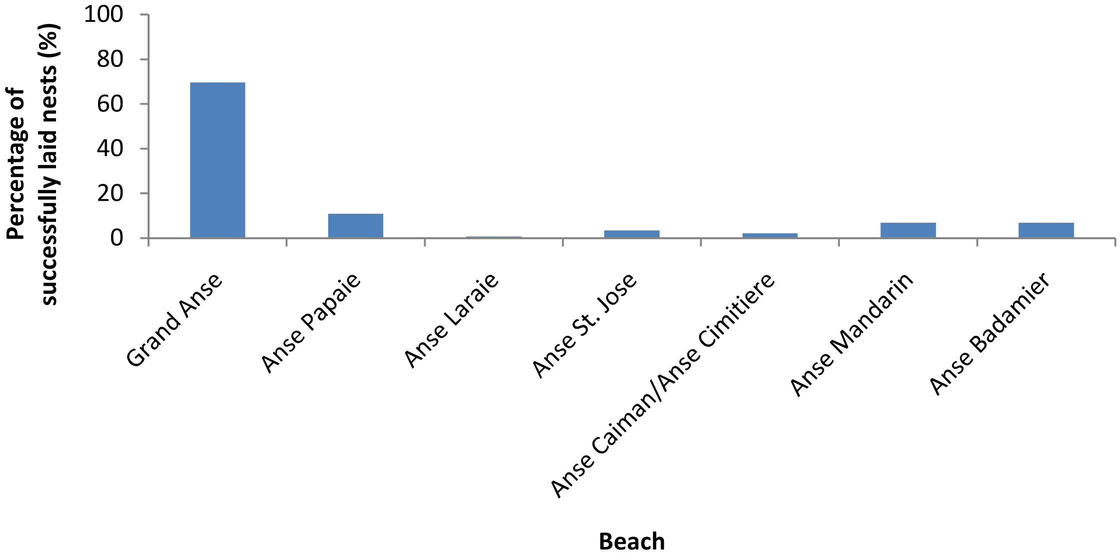

Figure 26. Percentage of total nests laid on each of the seven nesting beaches on Curieuse during the 2017-2018 Hawksbill season.

Beach suitability was compared by assessing nesting success on each beach. The higher the proportion of successful nesting activities compared to aborted nesting activities (i.e. non-nests), the higher the beach suitability. The most suitable beach for nesting was Anse St. Jose, with a nesting success rate of 63% (Figure 27). The least suitable beach was Anse Caiman/Cimitiere, with a nesting success of 7%. It should be noted, however, that small sample numbers from Anse St. Jose and Anse Laraie (n=8 and n=5, respectively) likely skewed the results for these two beaches.

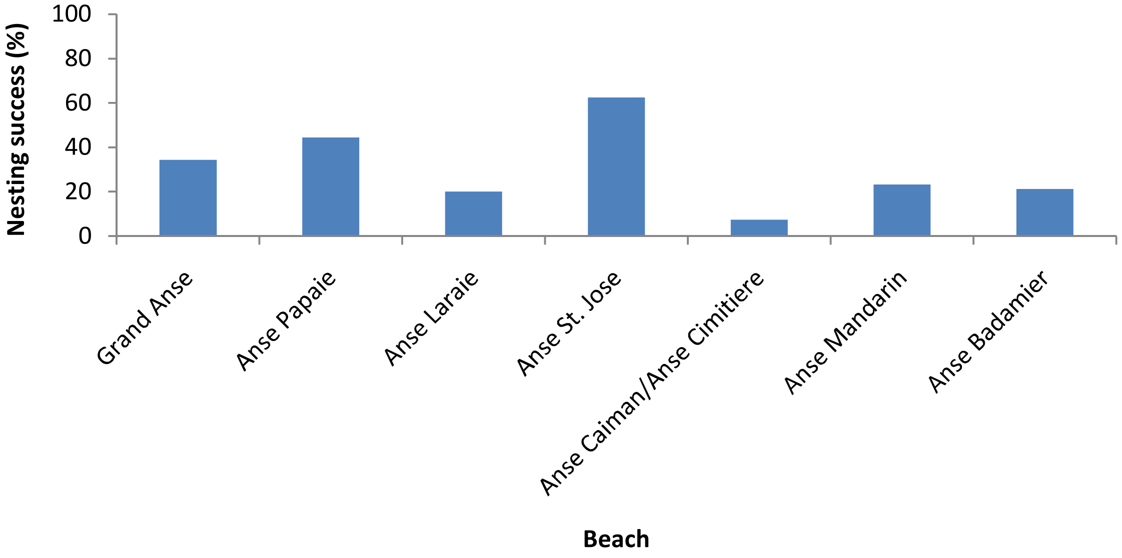

Figure 27. Beach suitability for the 2017-2018 Hawksbill season expressed as nesting success (number of nests divided by total number of activities) for each beach.

Nesting Adults – Green Turtles

For the 2017 – 2018 season, there were a total of 56 activities, of which 21 were nests, all laid on Grand Anse. The peak of nesting season occurred in July with 41% of all activities recorded. The population estimate for Green turtles was 4-7 individuals (Table 11).

Hatching Success

A total of 48 Hawksbill nests and seven Green nests were successfully excavated in the 2017 – 2018 season (Table 12). Hatching success was higher for Hawksbills (97.2%) than for Greens (89.7%). Grand Anse (94.6%, n=34) had the highest Hawksbill hatching success, followed by Anse Papaie (71.9%, n=9) (where enough excavations were done to obtain a reliable result). Three excavations were done on Anse Mandarin (89.8% hatching success), and one on each of Anse Laraie (62.6%) and Anse St. Jose (69.2%). One Green nest excavation was omitted from analysis due to nest exposure from erosion. 11 triangulated Hawksbill nests were lost to erosion over their incubation period.

Table 12. Hawksbill and Green turtle excavation parameters collected for the 2017 – 2018 season.

	Hawksbill	**Green**
Number of Excavations	48	6
Hatching Success (%)	97.2	89.7
Emerging Success (%)	88.6	89.7
Average Clutch Size	158	91
Average Nest Depth (cm)	47.4	65.8
Total Hatched Eggs	6837	475

Nesting Hawksbill Identification

There were a total of 34 encounters during the 2017 – 2018 season. Of these, 12 turtles had tags fitted by GVI and were recorded as newly tagged turtles. 17 already tagged turtles were encountered. Four turtles had a tag missing from one flipper; new tags were applied. One turtle had two new tags applied in addition to the old tags as they seemed likely to become detached. The majority of turtles were only encountered once, however four were encountered twice and one was encountered three times (two encounters occurred on the same day). No turtles encountered in the 2017-2018 season were encountered in the previous season.

Discussion

The 2017-2018 nesting season experienced a slow start, and fewer Hawksbill activities and nests were recorded compared to the previous season. However, this can be expected due to the biannual nesting nature of Hawksbill turtles. The encounter rate of Hawksbills was lower during the 2017-2018 season than the 2016-2017 season (n=34 and n=73, respectively). Higher encounter rates enable a closer look at inter-nesting behaviour of Hawksbills (discussion follows).

Table 11 highlights the importance of considering the amount of effort spent on recording this data by means of turtle patrols. Over the last few seasons, more time has been spent on turtle patrols and therefore more data collection has been possible, which could suggest why our estimations of number of individuals are higher, with the exception of last season. The understanding of Hawksbill population trends will be aided with the completion of the 2018-2019 season.

This was the sixth consecutive season of metal tag application by GVI staff. Tagged Hawksbills encountered this season have once again included tags placed on turtles on other islands in Seychelles, indicating movement of females between islands. The population size estimates were based on the assumption that Hawksbills lay an average of 3-4 clutches per season (Burt et al. 2015), and that all of these clutches were laid on Curieuse. If there is movement of nesting females between islands within a season, then this may be an underestimation of the number of females nesting on Curieuse annually. The continuation of metal flipper tagging and recording of tag numbers will hopefully allow for a greater understanding of the degree of inter-nesting intervals and changes in nesting site selection, and the calculation of population size using mark-recapture techniques. This may lead to a more accurate estimate of the nesting female population, though collaboration and sharing of data between islands may be required for a high resolution population estimate. The photo ID system will continue in order to compare newly tagged females with previously identified individuals. Once all photo ID individuals are given metal tags, photo ID will supplement flipper tag numbers as a backup system. It may also allow for the identification of turtles that are encountered but not tagged (such as those already leaving the nesting site).

During the 2017-2018 season one Hawksbill turtle was encountered on three separate occasions - two days in a row and twice in one day - and four individuals were encountered on two separate occasions. During the 2016-2017 season, the highest number of repeat encounters was three. With a higher proportion of turtles encountered more frequently, a better understanding may be gained of their inter-nesting behaviour. The 2017-2018 season had insufficient data to interpret inter-nesting behaviour; however, turtles encountered three times during the 2016-2017 season, had 43 day inter-nesting periods (interval between first and second encounter) followed by 13 days between nesting (second and third encounter), which correlates with the known inter-nesting behaviour of the species. Nevertheless, we must consider that these turtles may have nested, or attempted to nest, between these days without being encountered

Green sea turtles lay nests throughout the year in the inner granitic islands, though the low number of encountered tracks gives a poor indication of the nesting population. Green sea turtles lay at night and very infrequently throughout the entire year, making tagging and photo identification on Curieuse impractical. Green turtle activities for the 2014-2015 and 2015-2016 seasons were remarkably higher than previously seen on Curieuse. There was a considerable decrease in Green turtle activities during the 2016-2017 season. The 2017-2018 season saw an increase in activities, similar to the 2014-2015 and 2015-2016 seasons. Annual fluctuations of over 70 turtles have been recorded on various islands (Mortimer 2004). However, few Green turtles are estimated to nest in the inner granitic islands. A study from the 2001-2002 and 2002-2003 nesting seasons on Curieuse estimated that one to two Greens nested on Curieuse annually (Mortimer 2004). Data from 2012-2014 indicates a similar number annually (Burt et al. 2015). However, data from the 2014-2015 and 2015-2016 seasons suggests a significantly higher number (five to seven and five to nine respectively) of nesting Green turtles on Curieuse. The 2017-2018 season recorded 56 activities, of which 21 were nests, producing a population estimate of four to seven nesting Green turtles. With only a few years of year round, regular beach surveying, and unknown re-migration intervals (time between nesting periods) for Greens in Seychelles, it is impossible to draw any conclusions from this with regards to changes in population size. However, regardless of the status of recovery of the Green turtle population in the inner granitic islands, it is imperative that nesting females are protected and nesting is monitored consistently.

A study of Hawksbill hatching success was conducted on Curieuse for the 2001-2002 and 2002-2003 nesting seasons for a selection of nests (n=65). Overall, the hatching success (number of hatched eggs) was approximately 60% (Mortimer 2004). This differs somewhat from the current approximation, although excavation categories also differ slightly. The overall Hawksbill hatching success rate of 97.2% for the 2017-2018 nesting season (n=48) seems high when compared to previous data and other islands. Hatching success for Green turtle nests was lower (89.7%). However, the small sample size (n=6) should be taken in to consideration.

While in the past turtle nests were more evenly distributed across Curieuse's beaches, they are now mostly concentrated on 240m of beach at Grand Anse and Anse Papaie, resulting in an annual nesting density of 34 clutches per 100m (Burt et al. 2015). In the 2017-2018 season, Grand Anse continued to be the most utilised nesting beach for Hawksbills, followed by Anse Papaie, with 80% of all nests laid on these two beaches. The other beaches are less suitable for a variety of reasons including erosion (Anse Mandarin, Anse Badamier, Anse Cimitiere), high levels of disturbance from tourists/residents (Anse Laraie, Anse St. Jose, Anse Caiman), and a limited area of plateau behind the beach (Anse Badamier).

In light of recent discussions regarding increased development on Curieuse for tourism, it is imperative that Grand Anse and Anse Papaie remain safe, undisturbed areas for nesting Hawksbill and Green turtles. It is recommended that SNPA continues to actively prevent tourist access to Anse Papaie and Grand Anse, and install more educational boards to inform tourists of park zonation, restricted areas, and the appropriate code of conduct if encountering turtles on beaches populated by tourists during the day. Also, as recommended, GVI Seychelles staff and volunteers will continue to assist SNPA rangers to clear areas of Grand Anse that are currently inaccessible to turtles due to fallen trees in the hope that this reduces the number of unsuccessful nesting attempts due to obstacles. This may also potentially reduce the number of nests laid below the high tide line and increase hatching success for Greens in particular, since distance from the high tide line can be correlated with hatching success if the beach is prone to inundation by storm swells (Mortimer 1990).

Conclusion

Protection of the nesting beaches may be the most critical component of any sea turtle conservation program (Mortimer 2004). The knowledge that Curieuse Island may be used by up to 123 nesting Hawksbills, and up to nine Green turtles annually shows that it is essential to monitor these nesting populations and maintain high levels of conservation. It is possible that large scale annual fluctuations occur in the number of females arriving at nest sites (Limpus and Nicholls 1988) and therefore long-term monitoring is essential to document true population change (Meylan and Donnelly 1999). Therefore, the existing monitoring schedule of four times a week during peak Hawksbill nesting season, and at least once a week outside of Hawksbill season, will be continued to ensure reliable monitoring of turtle nesting. The fact that the Curieuse population of nesting Hawksbills has not experienced the degree of recovery witnessed at other more protected islands stresses how imperative it is that Curieuse Island turtle nesting beaches are not subjected to further development. Instead, a higher level of protection should be implemented to ensure the future of Curieuse as a vital Hawksbill rookery.

Sicklefin Lemon Sharks

Introduction

The Sicklefin lemon shark (*Negaprion acutidens*; Ruppell 1835) is one of two extant species of lemon shark, and one of 58 shark species known to inhabit the territorial waters of Seychelles (Seret 2002). This species ranges throughout coastal waters of the Indian and southwest Pacific Oceans, including many islands in Seychelles (Bester 2014; Figure 28). The Sicklefin lemon shark is a large shark of the family Carcharhinidae (requiem sharks), typically growing to a length of approximately 3.0m (Carpenter and Niem 1998). It is distinguished by the almost equal size of its two dorsal fins, and by the typically pale yellow colouration which gives rise to its name.

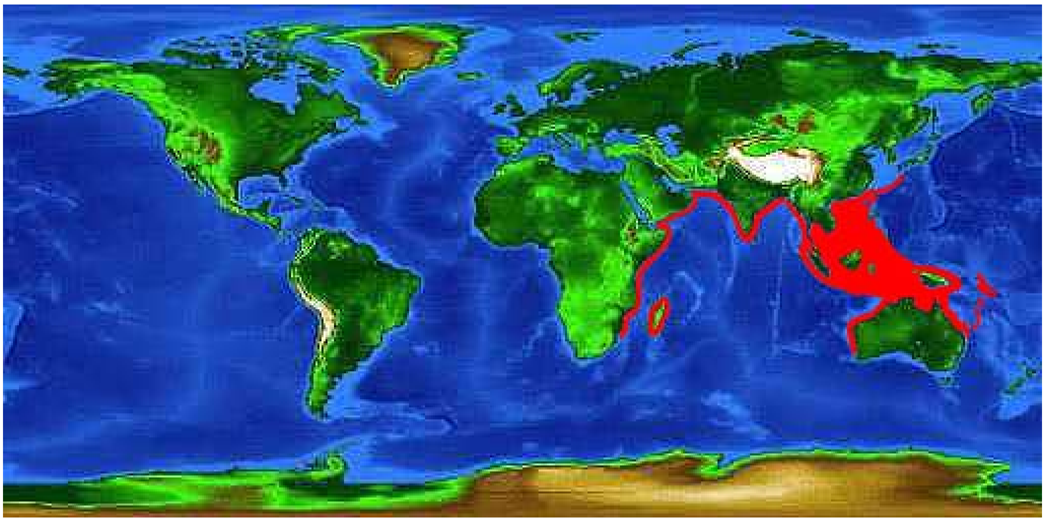

Figure 28. Distribution of the Sicklefin lemon shark, *N. acutidens* (Image from IUCN).

Categorised as vulnerable (IUCN 2014), in part due to its coastal preference and consequent proximity to human activity, it faces many threats to its continued survival. The species is fished throughout its range (Compagno 1990), and its small habitat range and limited movement patterns make it susceptible to local depletion (Stevens 1984, Stevens et al. 2000; Shultz et al. 2008).

There is an overall lack of information regarding Sicklefin lemon shark life history, with approximately only 35 publications available on the species from a limited number of populations and geographical locations. Although a greater amount of research has been conducted on the Atlantic congener *Negaprion brevirostris*, information regarding this closely related species should not necessarily be applied to the management of *N. acutidens*. It is essential to conduct species-specific studies concerning their life history patterns and population trends over time in order to effectively conserve and manage them, and potentially increase the population size.

In 2007, the government of Seychelles produced a National Plan of Action for the Conservation and Management of Sharks (NPOA)(Seychelles Fishing Authority 2007). The plan was updated in 2016 and recognises the nation's commitment to, and sets out national strategies for, the conservation of all shark species in Seychelles waters. The key aim is "that shark stocks in the Seychelles EEZ are effectively conserved and managed so as to enable their optimal long-term sustainable use," and one of the main mechanisms to achieve that aim is to collect more information on these shark species. The assessment of the NPOA confirmed that shark stocks in Seychelles have followed a pattern of decline over the past few decades as seen in the majority of shark populations worldwide. This, coupled with the paucity of information regarding Sicklefin lemon sharks overall, highlights the need for long-term studies of Sicklefin lemon shark populations, particularly within prominent marine protected areas such as CMNP.

Through observations by staff and volunteers from SNPA and GVI Seychelles in CMNP, it has been known for many years that juvenile lemon sharks are present in the mangrove and seagrass habitats at Baie Laraie. There appears to be a clear annual cycle of parturition beginning in September and lasting for three to four months (similar to observed parturition times on other Indian Ocean islands; Stevens 1984), with an influx of many newborn lemon sharks. Population numbers appear to decline throughout the year, with relatively few individuals observed between January and August each year.

The ongoing mark-recapture study of the Curieuse *N. acutidens* population began in October 2014 and is currently in its fifth research season. Also, an active acoustic tracking study of juvenile *N. acutidens* began in February 2017, but was not continued into the 2017-2018 season due to adverse weather conditions and lack of significant tag numbers. Throughout the five year study on Curieuse, Sicklefin lemon sharks have been the sole shark species captured in the lagoon, until this season. For the first time the IUCN Near Threatened juvenile Blacktip reef shark, *Carcharhinus melanopterus,* was also caught in the lagoon, using identical methodologies to the lemon shark study, and will be discussed in the results.

Aims

The primary aim of shark monitoring at CMNP is to collect data on the juvenile Sicklefin lemon shark population, which allows for the long-term monitoring of trends in population size and dynamics, body size, and body growth parameters. Current shark monitoring contributes to the key aim of the Seychelles NPOA by increasing knowledge on local shark populations, and by facilitating the adaptive management of this population using the best available science. This knowledge will also be used to educate people on the importance of sharks and the value of CMNP as a shark nursery.

Methodology

Study Site

Monitoring was conducted within the Turtle Pond and fringing mangrove forest at Baie Laraie (Figure 29). The Turtle Pond represents a 40 acre shallow lagoon partially enclosed by a sea wall across Baie Laraie, constructed in 1910, which was originally intended for the farming of Hawksbill turtles (*Eretmochelys imbricata*); however, this was unsuccessful and quickly abandoned. The wall, now partially destroyed by the 2004 Indian Ocean Tsunami, has created a unique environment that has allowed the lagoon's fringing mangrove forest to flourish into one of the largest and most diverse remaining in the Seychelles Inner Islands.

Figure 29. *N. acutidens* study area within the Turtle Pond at Baie Laraie, CMNP, Seychelles.

The Turtle Pond was chosen based on previous studies of nursery areas and site fidelity in *N. acutidens* and *N. brevirostris*, as it was believed that the shallow waters and mangroves would provide a suitable nursery area for neonates. It is also easily accessible for the transportation of research equipment. The seaward edge of the mangrove forest is predominantly comprised of *Rhizophora mucronata,* which is inundated up to 1.24m during spring tides (unpublished data: Global Vision International). Sand flats comprise the landward edge of the lagoon, which are exposed at low tides, while seagrass beds located in the central area of the lagoon are only partially exposed at the lowest tides. There are several deeper sections abutting the wall, with sandy substrate and sporadic patches of coral. At tides below 0.7m, the southernmost section of the lagoon is isolated, forming a pool approximately 25x50m, which is referred to as "Pat's Pool" (Figure 2).

Capture Methods

Surveys were conducted around dawn (approximately 05:00-08:00) and dusk (approximately 17:00-19:30). Due to sampling limitations resulting from the heterogeneous nature of the study area, several methods of capture were used:

1. Seine nets – 90x0.75m and 10x1.5m (the latter being decommissioned in July 2016), with a stretched mesh of 10mm. The 90m seine was designed to be deployed in the open waters of the Turtle Pond and used either as a purse or beach seine, or placed at the mouth of a drainage channel for the mangroves at low tide (coined the "Lemon Shark Highway"). The 10m seine was designed for blocking narrow channels and openings.
2. Gill nets – 25x1.5m, 18x1.5m, and 10x1.5m, with a stretched mesh of 60mm (the 10m net was decommissioned in August 2016 and replaced with the 25m net). Gill nets were used under constant observation, either static or dragged slowly in the shallows.
3. Hook and line – size 14 barbless circle hooks, with fish used as bait. These were used in the first research season but discontinued in April 2015 due to concerns over the welfare of hooked individuals.
4. Cast net – 3m in diameter, similar mesh to the gill nets. This method proves most useful in very shallow water in the Turtle Pond or in restricted areas within the mangroves.
5. Dip nets – 60cm diameter, similar mesh to the seine nets. Used either independently or to safely remove and/or transport sharks from the aforementioned nets to the workup station.

Tagging and Data Collection for Mark-Recapture Study

Upon capture, each individual was transported to the workup station by hand or dip net, then placed in a large water filled holding crate. During the workup process, sharks were transferred to a water filled PVC trough (150mm diameter) with an integrated measuring tape. This method reduces stress by allowing the sharks to respire in the water during tagging and measurement. New captures were tagged with an internal Passive Integrated Transponder (PIT) tag (2.12x12mm *AEG ID162 FDX-B*), which was injected into the musculature beneath the first dorsal fin on the left side of the shark.

After tagging, the following measurements were taken to the nearest millimetre: pre-caudal length (PCL, from the tip of the snout to the pre-caudal pit), fork length (FL, from the tip of the snout to where the tail begins to fork) and total length (TL, standard length from the tip of the snout to the end of the caudal fin held in a natural position). A tissue sample was taken for genetic analysis using a fin snip from the upper trailing edge of the anal fin, which additionally offers a permanent indication of prior sampling should PIT tag shedding occur. All samples were immediately fixed in 100% ethanol. Weight was measured using a sling and hanging scale (accurate to 50g) before returning the shark to the holding crate. The shark was then overturned to expose the ventral region to ascertain gender and state of umbilical scar closure (recorded as either: open, ¾ open, ½ open, ¼ open, closed (fresh) or closed), and the genital region and umbilical scar were photographed. Care was taken to ensure the mouth and gills were submerged whenever possible. The shark was then released, and each individual followed for as long as required to monitor recovery. For recaptured individuals, length, weight, gender, umbilical scar closure, and injury data was collected using the same protocol as new captures. Additionally, capture method was recorded and GPS position was taken for each individual capture location.

Population Estimates

Estimation of population size was calculated using the POPAN (Schwarz & Arnason 1996) module of the MARK 8.0 mark-recapture software. This model calculates a super-population N using a Jolly-Seber calculation (Jolly 1965, Seber 1965) from an input matrix consisting of capture histories of all individuals marked during the sampling period. This is an open population method of calculating abundance whereby individuals may enter or leave the study area from the super-population by emigration, immigration, births, or mortality. A number of conditions common to all Jolly-Seber models must be met: 1) Every animal present in the population has the same probability of capture, 2) Every marked animal has the same probability of survival until the following sampling time, 3) The method of marking is permanent and cannot be overlooked, 4) All samples are instantaneous and each release is made immediately after the sample (Pollock et al. 1990).

The input matrix consists of capture histories for each individual in binary form, e.g. 0011010 denoting the individual was first captured on sampling occasion three, and then again on occasions four and six, after which it was not encountered again. It will generally not be known whether the individual was not encountered due to absence at the time of sampling, permanent emigration from the study area, mortality, or evading capture. In most Jolly-Seber formulations sampling mortality can be a major confounding factor, however in contrast, the POPAN formulation accounts for sampling losses which may otherwise violate assumptions of the model, denoted by "-1" following the individual's capture history.

Condition Factor

Condition factor (CF) was calculated for all captures using the following equation (from Hussey et al. 2009): CF = (weight [kg]/PCL^3)×10^5). This is an insightful method of converting length and weight into a single value that can be used to track trends in body condition. Fish exhibiting a relatively high condition factor value may be indicative of favourable environmental conditions (e.g. population density, prey availability, habitat quality, etc.), and changes in condition may be related to changing environmental conditions over time (Blackwell 2000). This formula was used to analyse the condition of year-0 sharks over time.

Results

The following section is divided into the results for the lemon shark study, and the results for the blacktip study, undertaken over the 2017-2018 research season.

2017–2018 Lemon Shark Study

Research Effort

The first *N. acutidens* neonate capture of the 2017-2018 cohort was on the 9th of October 2017. This marked the starting point for captures of the 2017-2018 research season. Research activities for this season comprised a total of 55 survey sessions over 366 days. Sampling effort varied throughout the season (Figure 30), with an average of four (±2.5SD) surveys per month. The highest sampling effort by number of surveys was in October and November 2017 (n=9 surveys/month). Survey effort was relatively low for the month of December due to lack of available manpower, and from March until the end of the season due to low capture rates.

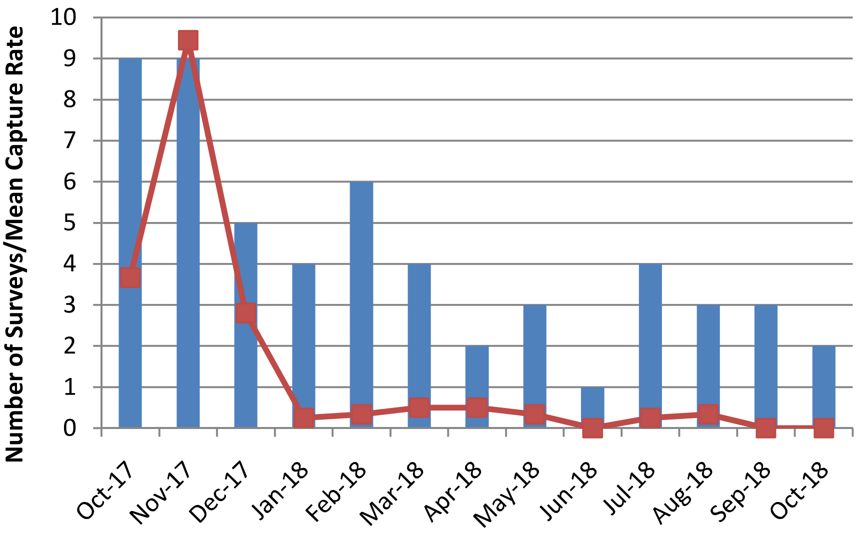

Figure 30. Total monthly sampling effort in number of surveys (blue bars) and mean *N. acutidens* capture rate per survey (red line) over the 2017-2018 season.

Capture Overview

A total of 141 captures were made over the 2017-2018 season, comprised of 113 new captures and 28 recaptures (of 23 individuals). No individuals initially captured during the 2016-2017 season were recaptured during the 2017-2018 season.

Compared to the 2016-2017 season, the total number of captures in the 2017-2018 season was 11% lower, with 5% less new captures and 30% fewer recaptures. The total number of captures varied among sampling sites (Figure 31), with the highest number of captures occurring in the North Turtle Pond (n=95), and lowest in the South Turtle Pond (n=2). Over the entire season, the mean capture rate was 2.6 (±4.5SD) captures per survey, however capture rates varied by month (Figure 30). Initial neonate capture began in October, with a mean capture rate of 3.7 individuals per survey ±4.18SD. The mean capture rate peaked in November, with a capture rate of 9.4 individuals per survey ±6.50SD. Mean capture rate then declined in December (2.8 ±1.64SD captures per survey) until the end of the season.

The most successful capture method involved the use of gill nets, which accounted for 78.0% of captures, followed by the seine net, which accounted for 18.4% of captures. The dip net and cast net methods accounted for 3.5% and 0% of captures, respectively.

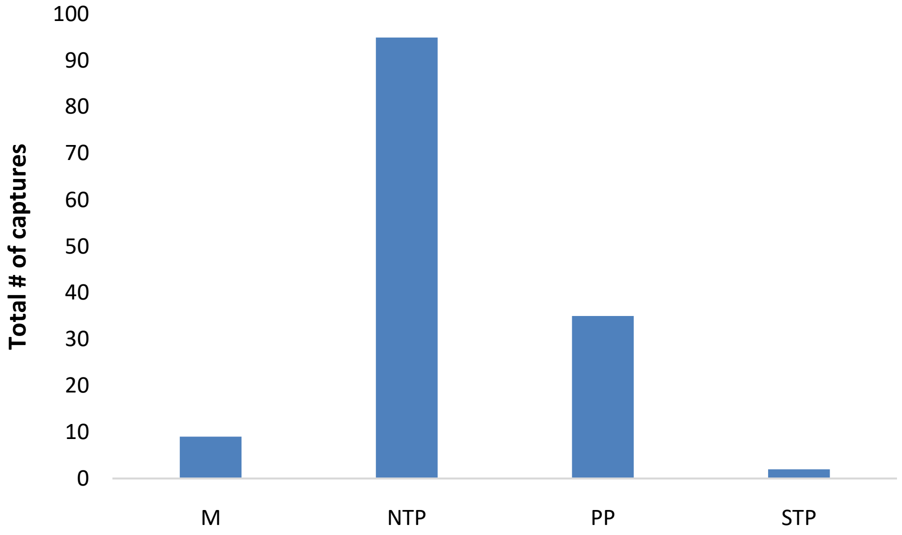

Figure 31. Total number of *N. acutidens* captured by sampling location, over the 2017-2018 season. M= Mangroves, NTP = North Turtle Pond, PP= Pat's Pool, STP = South Turtle Pond.

Population Estimate

The size of the juvenile *N. acutidens* population for this season was estimated at 661 (±278.7SE, range: 327–1515, 95%CI). This population estimate is 212.5%, 259.2%, and 181.6% higher than the 2014-2015, 2015-2016 and 2017-2018 seasons respectively (Figure 32).

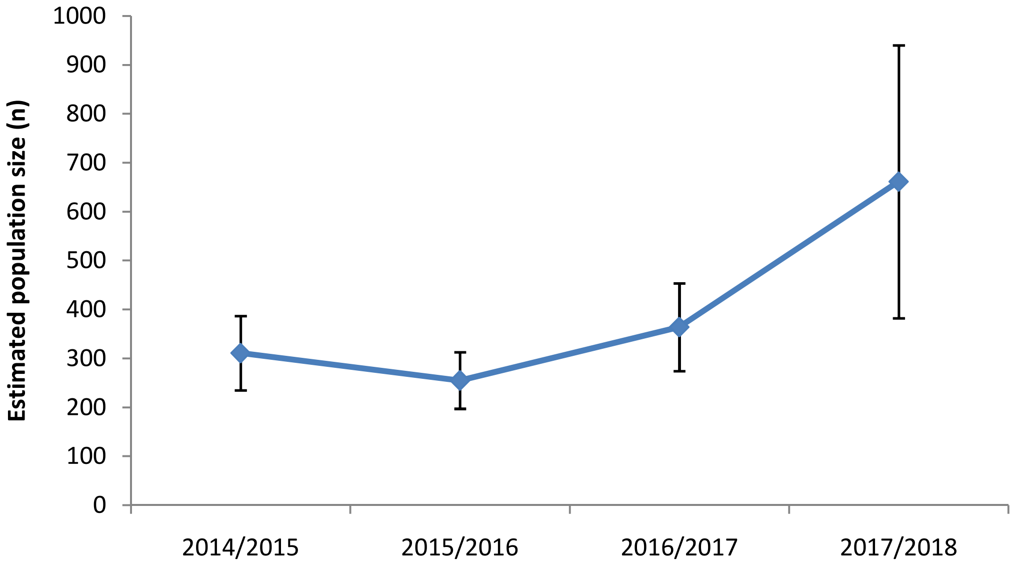

Figure 32. Population estimates (n±SE) for *N. acutidens* in the study area over four seasons.

Sex Ratio

A total of 54 females and 59 males were captured, resulting in a ratio of 109 males: 100 females. This compares to the 2014-2015 season which was also male-biased (126 males: 100 females). Both 2016-2017 and 2015-2016 cohorts were female-biased (98 males: 100 females and 83 males: 100 females, respectively).

Pupping Season

The first observed neonate of the 2017-2018 cohort was captured on the 9th of October 2017, marking the beginning of the observed pupping season. This was determined by the presence of an open umbilical scar on the captured individual. The final individual to be observed with an open umbilical scar was incidentally captured on the 4th of December 2017. Therefore, the 2017-2018 pupping season was an estimated 57 days long, 17 days shorter than 2016-2017 season and 18 days shorter than the 2015-2016 season.

Size at First Capture

All newly captured individuals were young of the year. Size at first capture data was used to produce the following summary statistics for the 2017-2018 cohort over the season. With regards to size, mean PCL was 50.7cm (±2.8SD), range: 45.1–66.7cm. Mean FL was 56.1cm (±3.0SD), range: 46.8–72.9 cm. Mean TL was 65.3cm (±3.7SD), range: 54.7–85.8 cm. No significant differences (α=0.05; t-test) in size (PCL, FL, and TL) were detected between sexes or seasons. The mean weight at first capture over this season was 1.5kg (±0.3SD), range: 0.85–3.75kg. No significant differences (α= 0.05; t-test) in weight were detected between sexes or seasons.

Size Trends

Upon comparing mean PCL by month (Figure 33), the length of captures slightly increased between September 2017 and January 2018, with a relatively low variation among individuals. Large fluctuations in the average length of captures were evident from February onward, primarily due to low capture numbers (0-2 captures/month).

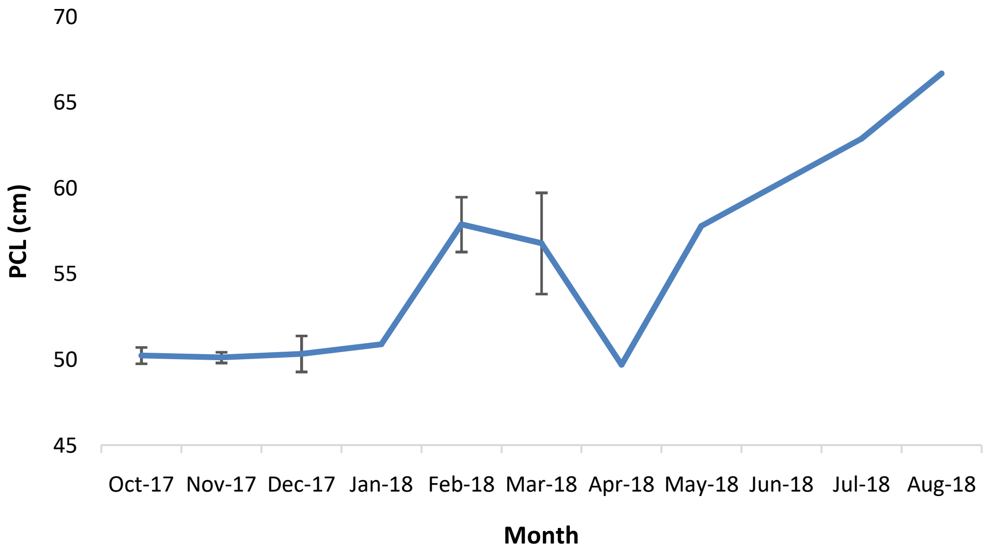

Figure 33. Mean PCL (±SEM) by month for *N. acutidens* captures over the 2017-2018 season.

The mean weight of captures by month (Figure 34) slightly decreased from September to December 2017. From December 2017 to January 2018 mean weight decreased, however only one neonate was captured in January. From February 2018 onward there was a steady increase in weight, except for an outlier in April.

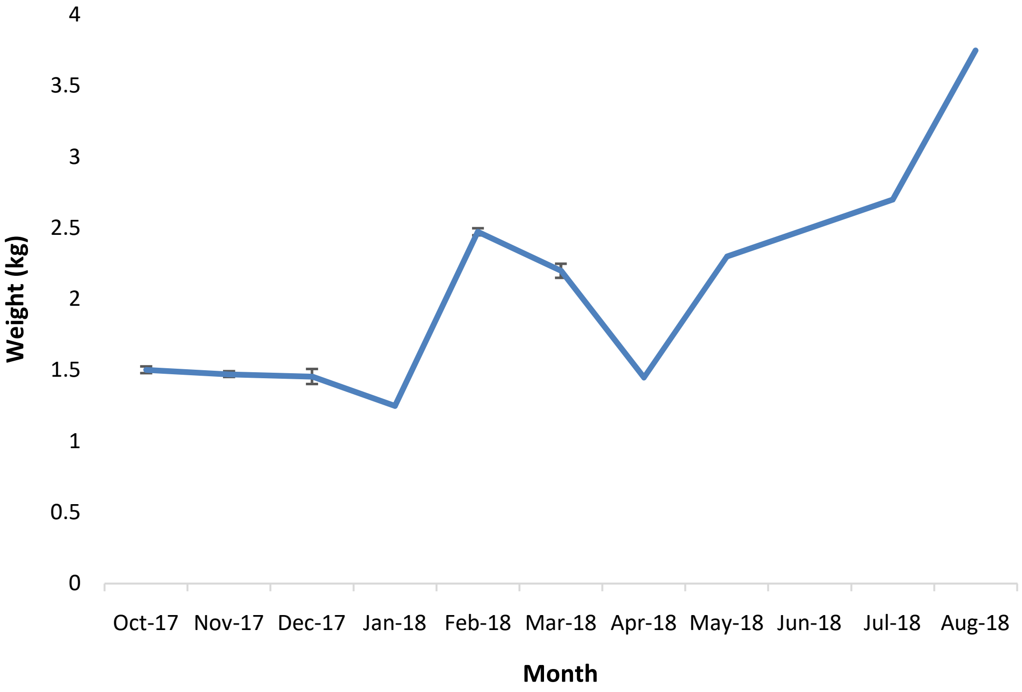

Figure 34. Mean weight (±SEM) by month for *N. acutidens* captures over the 2017-2018 season.

Condition Factor

The mean condition factor of the population over the 2017-2018 season was 1.17 (±0.14SD), range: 0.6–2.1. Upon comparing mean values by month (Figure 35), condition factor decreased between October 2017 and January 2018. Large fluctuations in the condition factor of captures are evident from January onward, most likely due to low capture numbers (0-3 captures/month). No significant difference (α= 0.05; t-test) in condition factor was detected between the sexes or seasons.

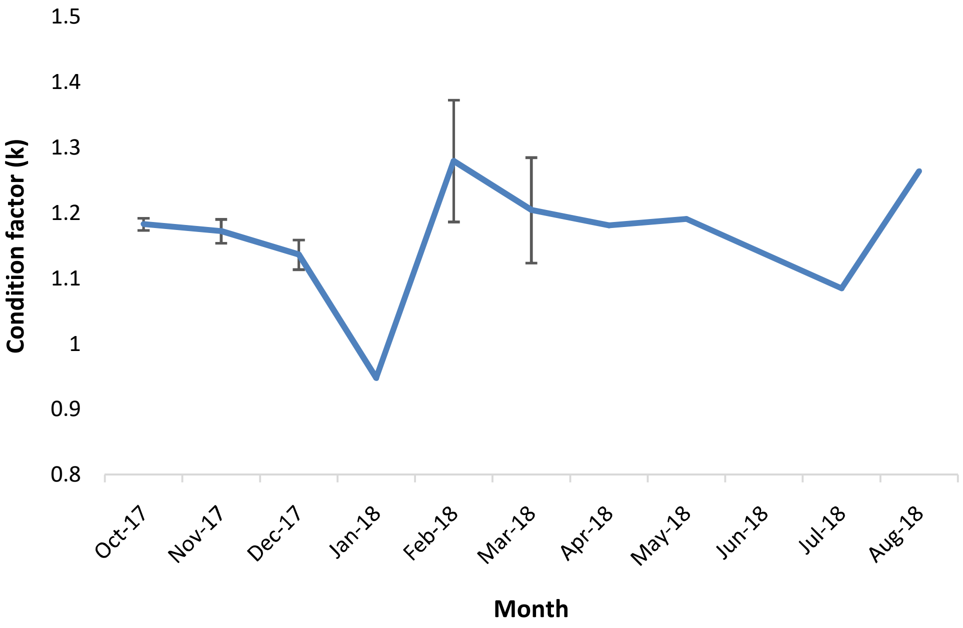

Figure 35. Mean condition factor (k) by month (±SEM) for *N. acutidens* captures over the 2017-2018 season.

Recaptures and Growth

2017-2018 cohort

A total of 28 recaptures were made of 23 individuals, which accounted for 24.8% of total captures in the 2017-2018 season. Of these recaptures, no individuals were initially captured during previous seasons.

Individuals of the 2017-2018 cohort were recaptured between one and three times. Recaptured individuals were at large for an average of 12.6 days (±10.7SD), range: 1-47. Mean growth in TL per day was calculated at 0.06cm/day (±0.13SD), range: -0.37–+0.4. Mean annual growth in TL was calculated at 21.17cm/year (±18.0SD), range: -133.83–+146.00. Mean growth in weight per day was calculated at -0.001kg/day (±0.013SD), range: -0.01–+0.019. Mean annual growth in weight was calculated at -0.037kg/year (±4.8SD), range: -3.3–+7.0.

Sampling Mortality and Natural Injuries

Sampling mortality for the 2017-2018 season was 0%. Natural injuries were observed in 1.8% (n=2) of captured sharks.

Natural Mortality

One juvenile shark was found dead in the Baie Laraie area on the 18th of December 2017. The individual was PIT tagged and had been originally captured 19 days prior and exhibited a healthy condition. Upon examination after death the individual did not display any signs of abnormal physical condition.

Juvenile Blacktip Reef Sharks

11 juvenile Blacktip reef shark captures were made between the 1st of July and 15th of October 2018. Of those, six were PIT tagged and confirmed as new captures, and one was recaptured. The 11 captures were made over seven consecutive surveys (targeting lemon sharks), averaging 1.57 (0.79±SD) blacktips per survey. Gill nets accounted for 100% of blacktip captures; 10 were captured in the North Turtle Pond, while one was captured in the South Turtle Pond. Of those tagged, three were male and three were female, resulting in a sex ratio of 100 males: 100 females. The final individual to be observed with an open umbilical scar was captured on the 28th of August 2018. Therefore, the observed 2018 blacktip pupping season was an estimated 59 days long. With regards to size at first capture (for the six tagged individuals), mean PCL was 42.0cm (±1.4SD), range: 39.8–43.5cm, mean FL was 46.9cm (±1.6SD), range: 44.6–48.5 cm, and mean TL was 57.5cm (±1.6SD), range: 55.3–59.2 cm. No significant differences (α= 0.05; t-test) in size (PCL, FL, and TL) were detected between sexes. The mean weight at first capture over this season was 1.1kg (±0.2SD), range: 0.9–1.4kg. No significant differences (α= 0.05; t-test) in weight were detected between sexes. The mean condition factor of the population over this season was 1.53 (±0.14SD), range: 1.4-1.8. No significant difference (α= 0.05; t-test) in condition factor was detected between the sexes.

Discussion

Currently in its fifth season, the Sicklefin lemon shark monitoring program at Curieuse has provided a wealth of information pertaining to the life history of this species, including baseline data on pupping season, body size, growth and condition, population size, and sex ratio. The continuation of this monitoring program will be critical in detecting any changes that may threaten this population of top predators that remain vital to the health and function of marine ecosystems within CMNP. As such, this data is valuable in the context of adaptive management, as the effectiveness of management actions targeted at preserving the species and its habitats can be informed by the data provided in this study.

Fluctuations in capture rates have been observed between and within seasons. The highest capture rates have tended to occur during October and November, and become reduced to low levels from late December/early January onward. This trend is consistent with juvenile populations of other shark species, which experience a population boom followed by a marked decrease; it is suggested that the large influx of neonates into an ecosystem at the start of a pupping season can be supported for a period of time, though natural selection through predation, competition for resources, and starvation results in large reductions in population size (Gruber et al. 2001; Lowe 2002; Duncan and Holland 2006; Heupel et al. 2007). Moreover, as year-0 sharks grow and mature they may also be utilising areas outside of the study site. Stevens (1984) noted that *N. acutidens* at Aldabra Atoll moved an average distance of 1.3km from their initial tagging site, with a maximum of 5km. This may indicate that the lower capture rates could also be the result of ontogenetic changes in habitat use.

Estimated population size appears to fluctuate from year to year, with the highest estimated population being observed in the 2017-2018 cohort (n=661), and the lowest during the 2015-2016 season (n=255). These fluctuations could result from a combination of the reproductive periodicity of *N. acutidens*, inter-annual changes in prey availability or habitat quality, and the number of reproductive females giving birth in the area. Similar inter-annual fluctuations in juvenile population size commonly occur in other shark populations (Bush 2003).

Between pupping seasons, sex ratio has remained relatively stable at approximately 1:1, ranging between 83 and 126 males per 100 females across all seasons. This is consistent with other populations of *N. acutidens* such as the population at Aldabra Atoll, Seychelles, in which 59% of captured individuals were female (Stevens 1984).

The actual duration of a pupping season is difficult to determine with a high degree of accuracy, as it is estimated based on the time between the first and final observations of open umbilical scars on captured individuals over a given season. Although a higher level of research effort is exerted in the weeks leading up to the previously estimated start dates, it is possible that parturition in this population can begin days or even weeks prior the first individual bearing an open umbilical scar being captured. Moreover, duration of the open umbilical scar state may vary greatly between individuals. Therefore, the durations of the pupping seasons provided here serve as a rough estimate (beginning and end of pupping seasons may be somewhat earlier than recorded). However, long-term research over future seasons should aid in determining the extent of the pupping season for *N. acutidens* in this population with increased precision. To date, the earliest recorded start of the pupping season is the 24th of September (2015), and the latest recorded end date is the 17th of December (2017), equalling a maximum estimated range of approximately 84 days.

Based on available data from the previous and current seasons, the consistent mean length, weight, and condition factor values among years suggests stability in these variables. However, long-term research must continue to be conducted in order to support this notion with any degree of certainty. Completion of the 2017-2018 season has resulted in data collection over three complete pupping seasons (the 2014-2015 survey began shortly after the estimated start of the pupping season).

On average, the length of individuals increased steadily throughout all four seasons, with relatively little variation among individuals between September and January. From February onward, variation in length among individuals was relatively high. In contrast, the mean weight of the population generally decreased or remained stable between September and January. The population exhibited positive growth throughout the remainder of season (bearing in mind sample size ranged from zero to two individuals per month). However, the individual caught in April 2018 displayed a lighter weight, but relatively normal condition factor, than those caught during the months before and after. This increase in weight and length dissimilarity over the later portion of the season was likely due to increased variation in the length of individuals as they mature, which has also been observed in *N. brevirostris* (Barker et al. 2005).

Mean condition factor between all seasons remained stable. However, it is evident that the mean condition factor of neonate sharks decreased significantly over the first three to five months following parturition (September to January). This reduction in condition is likely due to intense competition and/or difficulties learning to hunt over this period. Mean body condition fluctuated between February and July, and increased in August.

The trend of decreasing weight and body condition over the first three to five months of the season is consistent with the hypothesis of intense competition among congeners within a nursery habitat causing starvation, e.g. as suggested by Lowe (2002) for neonate scalloped hammerhead sharks (*Sphyrna lewini*). Moreover, a *N. brevirostris* population studied in Bimini, Bahamas was reported to have an estimated 35 to 62% neonate mortality in the first year, though largely due to predation (Gruber et al. 2001). It has been suggested that aside from predation, such mortality rates could also imply that nursery areas may not always provide sufficient resources (Heupel et al. 2007). However, difficulties learning to hunt and predation should not be ruled out as a cause for reduced capture rates later in the season; indeed in previous seasons 9.4–23.2% of captured individuals showed injuries that may have indicated predation attempts.

The TL growth rate for year-0 sharks reported here (21.17cm/year; ±18.0SD) is comparable to that of other studies of *N. acutidens* populations (12.5-15.5cm/year) (Stevens 1984) and Curieuse individuals from the 2015-2016 cohort (22.4cm/year; ±5.7SD). The growth rates provided here are strictly representative of young of the year individuals, usually with only weeks or a few months separating when measurements were taken. Growth rates from Stevens (1984) were based on data from a combination of year-0 and older sharks; unfortunately, intermediate life stages have rarely been caught in this study, limiting the scope of the growth data primarily to year-0 sharks.

Several individuals exhibited negative growth rates. Such negative growth rates in this population could be due to the overall trend of reduced body condition observed between September and January of each pupping season leading to small reductions in length. A number of studies of sharks and other fish have presented evidence of negative growth rates. For example, reductions in length have previously been reported in some shark studies (Pratt and Casey 1983; Meyer et al. 2014), and salmonids in environments with low food availability have been reported to experience reductions in length of up to 10% (Huusko et al 2011).

With improved capture and handling techniques, sampling mortality has been reduced to levels much lower than other studies of lemon sharks. For example, Gruber (2001) experienced 0 – 11.1% mortality in a study of *N. brevirostris* in Bimini, Bahamas. In this study mortality rates were 0%, 0.69% and 0% over the 2015-2016, 2016-2017 and 2017-2018 seasons, respectively. This low level of sampling mortality is likely the product of the continual review and optimisation of handling and research procedures, which are conducted in order to keep sampling mortality as low as possible.

Without further specific research into potential factors leading up to the observed natural mortality that resulted in one juvenile *N. acutidens* being found dead on the 18th of December 2017, it is difficult to determine the exact cause. As discussed in previous reports, natural mortality could be due to a number of reasons including: failing to learn to hunt, intensive competition among congeners, and/or disease.

Understanding the habitat use of a marine species is critical for managing its protection. To date, neonate Sicklefin lemon sharks have been captured in the Turtle Pond and sighted from the shoreline between Point Rouge and Anse Caiman. However, whether or not juveniles use habitat outside of the Turtle Pond on the east side of Curieuse has yet to be determined. In 2017 the active acoustic tracking technique was determined to be ineffective at locating Sicklefin lemon sharks for the purpose of this study. However, it is suggested that passive acoustic tracking via the use of stationary ultrasonic receivers placed strategically in an array along the eastern coastline of Curieuse Island would be more effective in determining critical habitat for Sicklefin lemon sharks in the future. Presently, an application (to SEYCCAT) for funding is under review for the purchase of 20 acoustic tags and 12 acoustic receivers. These 12 stationary receivers would detect the individual ultrasonic frequencies emitted from acoustically tagged sharks. This strategy would not only provide consistent habitat use data for both diurnal and nocturnal movements, but would also help determine the effectiveness of CMNP in protecting juvenile lemon sharks at this critical life stage. Moreover, this array of ultrasonic receivers could be used with a variety of other marine species of research and conservation interest.

The capture of juvenile Blacktip reef sharks emphasised the continued importance of Curieuse Island as a nursery to multiple juvenile shark species. It is difficult to determine any significant results from such a low capture rate, however based on the variation in time of neonates with open umbilical scars it is estimated that there were at least two birthing blacktip mothers. This could rule out a fluke event and instead support the theory that juvenile blacktips may soon become established within the CMNP in future years. The lemon shark study will continue to work up any blacktips captured to gain an understanding of this threatened species within CMNP, and also assess whether their presence influences the resident lemon sharks.

Conclusion

The Sicklefin lemon shark monitoring program on Curieuse, now in its fifth season, has provided a robust set of standardised and comparable baseline data regarding population parameters such as pupping season, body size and condition, growth rates, neonate population size, and sex ratio. The neonate *N. acutidens* population monitored within the Turtle Pond area of CMNP appears to be stable in population size, body size, condition factor, and average growth rate year on year. Data from our ongoing research will be used to compare against future trends in order to continually inform park management actions regarding this important species. As such, the following recommendations are made based on the available data:

1. Increase day and night patrols aimed at preventing illegal fishing within the marine protected area in CMNP during and following the Sicklefin lemon shark pupping season (September to December), as sharks at all life stages appear to congregate within the MPA during this time. The ability to enact this recommendation would result in direct protection for breeding adult Sicklefin lemon sharks and their offspring, with additional benefits to other species targeted by illegal fishing within the protected area.
2. Begin acoustic tracking of sharks to elucidate shark movements and critical habitat throughout the protected area. Acoustic tracking could enlighten researchers and park management as to if, when and where year-0 lemon sharks are travelling, and potentially help to explain the significant decrease in shark captures following the pupping season. This additional data should help park management target protection for critical habitats in need of special attention and monitoring.
3. Ensure that the proposed reconstruction of a stone causeway at the Turtle Pond, Baie Laraie controls for and/or mitigates any environmental damage to shark habitat (coral, mangrove, and seagrass) adjacent to the heavy construction site. It would also be prudent to perform the proposed construction outside of the known range of the *N. acutidens* pupping season (September to December) in order to minimise any potential negative impact on the year's cohort.

The inaugural capture of multiple juvenile Blacktip reef sharks in the Turtle Pond highlights the continued importance of CMNP in protecting multiple shark species in their vulnerable life stages. The presence of blacktips may also indicate increased health of both the CMNP and Seychelles waters in general.

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