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GVI Seychelles – Curieuse

Island Conservation Expedition

Annual Report

2019

(January 2019 – December 2019)

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Seychelles National Parks Authority (SNPA)

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To all volunteers and staff from January 2019 – December 2019 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 2019 and December 2019.

The total rainfall for this time period was 2451.5mm (compared to 2018 and 2017 rainfalls of 2222.3 and 1867.5mm, respectively).

The seventh year of the annual Aldabra giant tortoise census was completed in December 2019, with a total of 134 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 in the past year the population appears to be stable. 15 additional individuals were added to the known population of free ranging individuals through donations of unwanted pets from Praslin. The tortoise nursery on Curieuse now houses 43 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 third time, in order to assess the relative abundance and diversity of predatory and scavenging species. 32 deployments were conducted at randomised sites off the north shore of Curieuse, with equal numbers of shallow (≤ 10m) and deep (≥ 15m) deployments. A total of 109 target species were determined for monitoring and 79 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 have added to the 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 until July 2019, from when annual surveys will be conducted. Each year, substantial changes to beaches have been observed between the Northwest and Southeast monsoon seasons. Seasonal trends are clear, 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). The long-term trend of decreasing width has reversed on Grand Anse, Anse Papaie and Anse St. Jose. All six beaches increased in area between 2018 and 2019. However, large amounts of erosion have been evident during December 2019 on many beaches, especially Anse St. Jose, and it remains to be seen in 2020 whether there has been an overall loss of sediment.

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 were *Rhizophora mucronata* and *Bruguiera gymnorrhiza*. *R. mucronata* decreased in abundance since 2018 in all quadrats except Quadrat 5 which showed an increase. *B. gymnorrhiza* increased in Quadrat 1 and decreased in Quadrat 4. *A. marina* decreased in Quadrat 3. *R. mucronata*, *B. gymnorrhiza* and *A. marina* were the only species to have seedlings and/or saplings present, and Quadrat 2 contained the most. 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.

A rat eradication study began in a two hectare area of coastal forest and wetlands behind Anse St. Jose in January 2019, with the aim of testing a deployment of novel humane automatic A24 rat traps produced by Goodnature Traps. The technology has proved extremely effective in removing all rats from the study area and maintaining it essentially rat free. Phase 1 resulted in the development of skink excluder devices and other methods of avoiding bycatch. Phases 2, 3 and 4 resulted in the removal of an estimated 675 Black rats from the ecosystem between April and December 2019. The estimated density of rats in the study prior to the start of the project was 73 individuals per hectare, with a projection of a total island population of approximately 21,000 individuals. Scavenging rates have been high, resulting in many carcasses not being located. A preliminary analysis of rat stomach contents indicated the presence of soil dwelling nematodes, ants, invasive plant seeds and fruit, and arthropods in the rat diet. A study using Goodnature rodent detector cards was carried out to provide the data required for interpretation of bite marks for Seychelles species. There may be early indications of ecosystem recovery following rat removal.

The 2018 – 2019 Hawksbill turtle nesting season began in August 2018 and lasted until the end of March 2019. Nesting activities peaked in December, and Grand Anse remained the most heavily utilised nesting beach. A total of 506 Hawksbill activities were recorded, of which 234 were nests, an increase on 2018, and the estimate of the total population of reproductive females was 59 – 78 individuals. A total of 24 Green turtle activities were recorded, of which 18 were nests, a decrease on 2018. The peak of Green turtle nesting appeared to be in January, and the population of reproductive females was estimated to be four to six individuals. In line with previous seasons, excavations showed high reproductive success for both Hawksbill and Green sea turtles, with a hatching success rate of 90.1% for Hawksbills and 98.7% for Greens. Photo identification and metal flipper tagging are being continued in the 2019 – 2020 season. The variable numbers of Green turtle activities and nests between seasons has continued.

October 2019 marked the end of season five and the beginning of season six of the juvenile Sicklefin lemon shark PIT tagging project. The project has been very successful in providing detailed population statistics to inform the management of this important marine predator in the Curieuse Marine National Park. Since the onset of the project until December 2019, a total of 697 individuals have been captured and tagged, and a total of 205 recaptures have been made, which has provided a wealth of information on changes in size, weight, and body condition. Over the 2018 – 2019 season 132 new individuals were captured and PIT tagged, with 37 recaptures of 29 individuals. The population estimate for the 2018 – 2019 neonate cohort was 337 individuals and population size appears to be stable. Growth rates and capture rates have closely followed patterns seen in previous seasons. Eight juvenile Blacktip sharks, two juvenile Blacktip reef sharks, and one critically endangered juvenile Scalloped hammerhead shark were also captured and PIT tagged.

A passive acoustic tracking study of neonate Sicklefin lemon sharks was initiated in August 2019, with the installation of shallow and deep *Vemco VR12W* acoustic receivers and *V13* transmitters for range testing. Following analysis of range testing data, a final deployment plan was produced, and 12 receivers were deployed throughout the lemon shark habitat at the eastern end of Curieuse. Following the arrival of the 2019 cohort of neonates in October, 20 transmitters were surgically implanted. Early indications are that most if not all individuals are healthy and displaying normal behaviour. Tracking data will be downloaded from all receivers in May 2020, when a full analysis of the lemon shark spatial behaviour will be carried out.

Table of Contents

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), responsible for management and research relevant to the protection of the national parks within Seychelles.

Island Conservation Expedition

The Seychelles archipelago contains 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, the Curieuse Marine National Park (CMNP) was established in order to protect the native wildlife. Today, it is home to approximately 145 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 2019 focused on the annual giant tortoise census, Baited Remote Underwater Video surveys, beach profiling, mangrove monitoring, sea turtle sea turtle nesting success and the continuations of the Sicklefin lemon shark PIT tagging program. Two significant additional projects have now also been added, with the commencement of a rat eradication study and a passive acoustic tracking study of the neonate Lemon shark population. 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 trail 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. understand appropriate handling and measurements for tortoises, sea turtles and 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

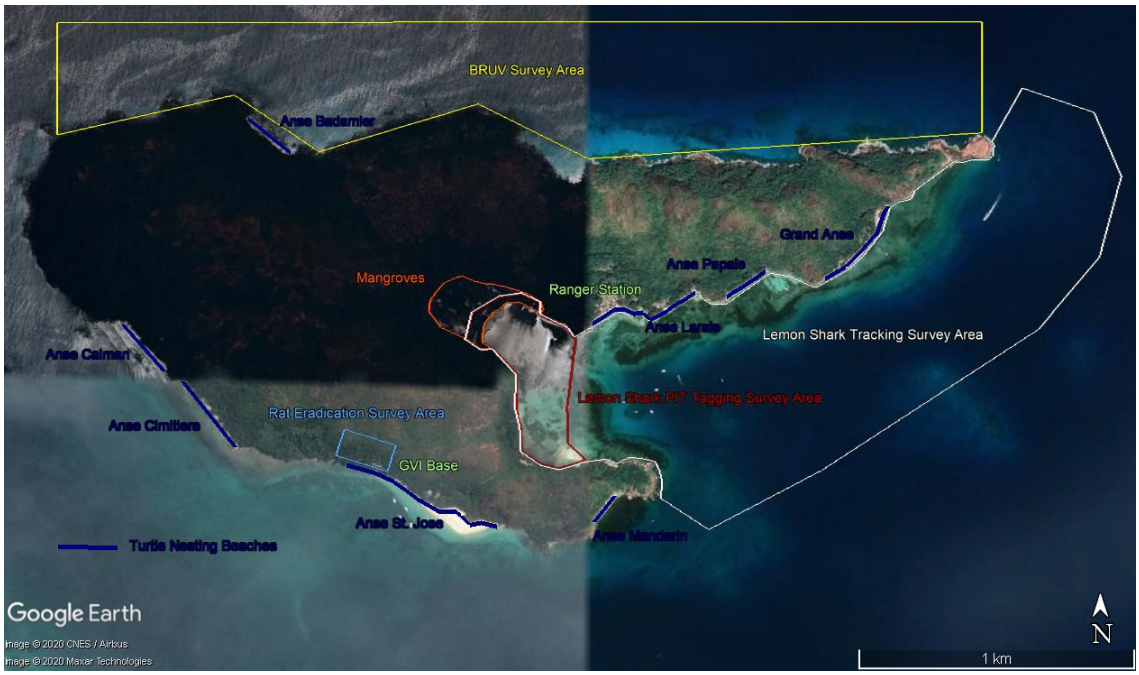

Figure 1. Curieuse Island, showing current survey sites.

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 previously. 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.

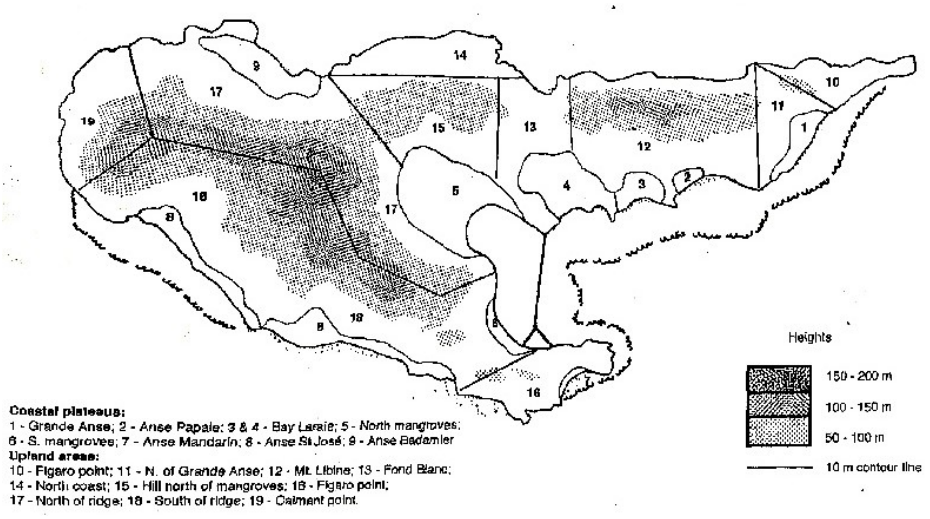

Figure 2. Map of Curieuse Island used in the annual census, originally used by the Oxford Expedition team in 1990 (Lewis et al. 1991).

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 3). This mark is not permanent and lasts only weeks or months, therefore allowing for rapid identification of tortoises already surveyed in the current census without the need to scan for internal tags. If a tortoise was unmarked, it was scanned for an existing Passive Integrated Transponder (PIT) tag (*Trovan ID 100*) using a scanner (*Trovan GR250*), and if previously untagged, a PIT tag was inserted.

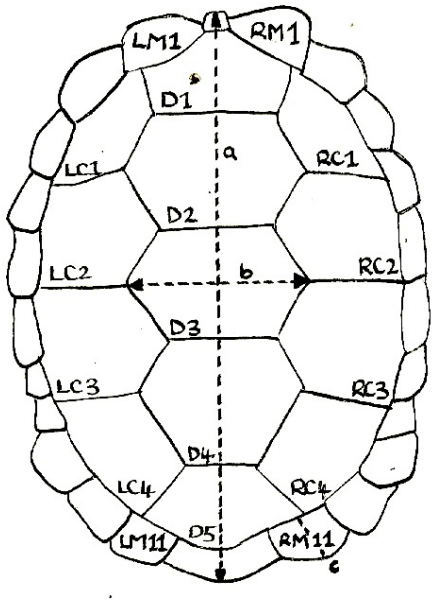

Figure 3. Tortoise carapace (upper shell). Dotted line a*:* Over-the-Curve Carapace Length (OCCL), b: width of the 3rd dorsal scute, c: point on the 11th marginal scute on either side, past which the tail is considered to be Long.

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) scute. 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 disc (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.

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. The length of the second claw from the outside of the hind right leg was measured.

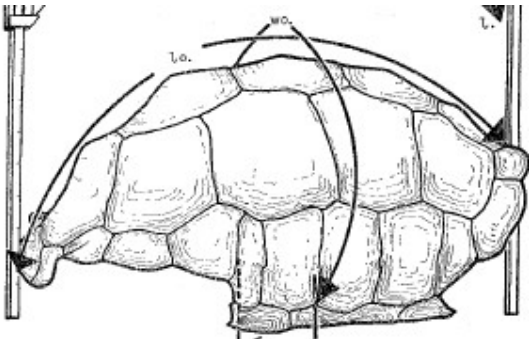

Figure 4. Width measurement as recorded over the carapace (from Gaymer 1968).

	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	*Nesting Behaviour or Cooperative Copulation	N/A	N/A

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

* indicates a characteristic that is definitive independent of other characteristics
NB - Nesting behaviour is defined as nest digging or egg laying

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 gender 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: 'Male', 'Potential Male' (those starting to display male sexual characteristics, specifically a slightly concave plastron), 'Reproductive Female', 'Juvenile', or 'Unknown' (immature male or female).

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, sides, plastron and any distinguishing marks such as extra/missing scutes or injuries.

Results

Tortoise Location

In this year's census, 134 tortoises have been encountered and identified. 14 individuals were not encountered in 2019; two of which (121 and 123) have not been found for the past six years, three (006, 038 and 115) have not been found since 2014, and one (050) has not been seen since 2015. An additional 15 tortoises were added to the Curieuse census population this year through donations of unwanted pets from locations on Praslin. As such, a population of 145 free ranging tortoises are believed to be present on Curieuse.

Tortoises were encountered primarily at the Ranger Station (n=105, 78.36%). Grand Anse had the second largest concentration (n=19, 14.18%) followed by Anse Papaie (n=4, 2.99%) and the mangroves (n=4, 2.99%). Anse St. Jose and Anse Badamier had the lowest concentrations (both n=1, 0.75%).

Gender and Size of Tortoises

This year, the census population consisted of four juveniles, 72 males, 33 potential males and 25 unknown. Table 2 displays the average width, OCCL, 3rd dorsal scute width and hind claw length for tortoises in each age/sex class in 2019. When hind claw length was considered as a percentage of the OCCL, potential males had the highest ratio (3.65%), followed by unknown gender (3.64%), males (3.50%), and then juveniles (2.53%).

	Juvenile		Potential Male		Male		Unknown	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD
3rd Dorsal Width (cm)	20.93	1.77	29.48	2.69	34.51	3.09	26.99	3.45
Width (cm)	68.60	7.91	106.75	10.01	126.89	8.69	97.85	9.01
OCCL (cm)	69.10	8.39	103.78	11.42	132.40	9.90	90.52	9.01
Hind Claw (cm)	1.75	0.29	3.79	0.80	4.63	0.65	3.29	0.91
Hind Claw to Length	0.0253		0.0364		0.0349		0.0363	

Table 2. Mean ±SD for measurements taken over each age/sex class in the 2019 census.

Average OCCL growth from 2013 to 2019 by age/sex class is shown in Table 3. Tortoises classified as potential males had the highest average growth (5.05cm), and males showed the lowest average growth (3.12cm).

	Potential Male (n=27)		Male (n=67)		Unknown (n=17)	
	Mean	SD	Mean	SD	Mean	SD
OCCL (cm)	5.05	5.70	3.12	4.14	5.03	7.12

Table 3. Mean OCCL growth for each of the age/sex classes between the 2013 and 2019 censuses.

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.

	Mean	SD
Growth (OCCL) (cm)	4.36	0.130
Growth (weight) (kg)	1.57	0.076

Table 4. Average growth in OCCL and weight for 24 hatchlings after approximately three years of growth (3/10/2016 to 19/11/2019).

Since the poaching incident, the number of hatchlings present in the nursery has been fluctuating seasonally (Figure 5). The nursery population decreased slightly between January and August 2017 and again between January and October 2018 due to predation by rats, death due to natural causes and the release of individuals 126, 128 and 135. 20 new hatchlings were found between September 2018 and November 2018, which brought the nursery population to 74. Since the beginning of 2019 to October 2019 there was a large decrease in nursery population due to rat predation and natural causes. Beginning in October and into to November 2019 12 new hatchlings were found around the Ranger Station and Grande Anse and relocated to the nursery, with a population of 43 at the end of 2019.

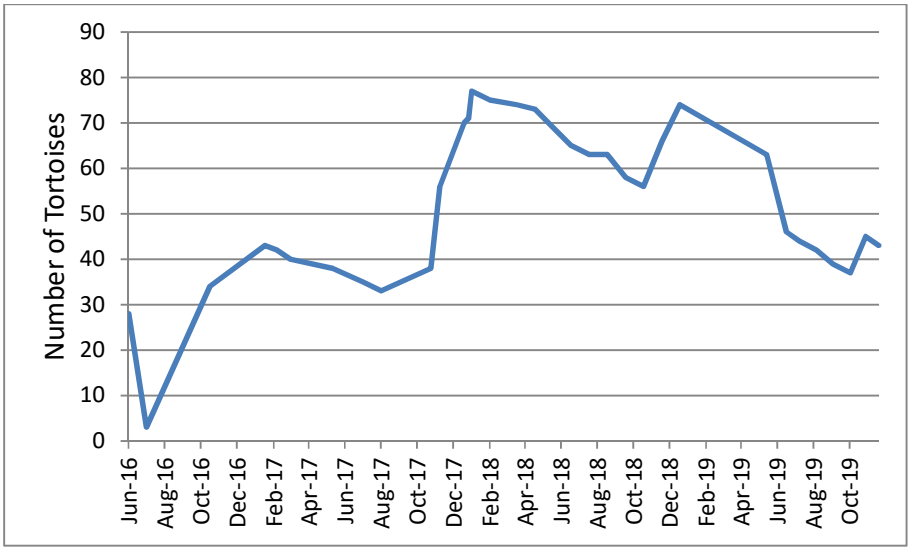

Figure 5. Number of tortoises in the nursery from June, 2016 to November, 2019.

Discussion

The 2019 annual Giant tortoise census continues to show that the methodology used for encountering tortoises remains effective, with 134 out of 148 known individuals encountered (90.5% of the free ranging population), including 13 of the 15 individuals that were donated to Curieuse. 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.

In the previous censuses and over six years, at least five tortoises are known to have died, including one from the 2016 census. Since 2017, no tortoises are known to have died, although in early 2018 tortoise 095 was in an unhealthy condition and has not been seen during the 2018 or 2019 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 in 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 105 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 19. Grand Anse is one of the largest beaches on the island, with a large stretch of wetlands/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. 12 individuals were also found there in previous censuses, indicating this may be a preferred 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 five 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 width of more than 21cm (Lewis et al. 1991). All Curieuse tortoises displaying some indications of being male (i.e. slightly concave plastron) have an OCCL of greater than 74.3cm and a 3rd dorsal scute width 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, it does not mean that they eventually 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 tortoises' exact ages are known, and after many years of consistent monitoring, 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 has been 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 potential males and tortoises of unknown gender had the highest average OCCL growth of 5.05cm and 5.03cm respectively between 2013 and 2019. The tortoises of unknown gender having a higher growth rate supports the theory that younger individuals display higher growth rates, since this class contains a mixture of immature individuals and mature females, with the immature individuals still growing to reach sexual maturity and maximum size. Males displayed an average growth rate of 3.12cm between 2013 and 2019. This is similar to previous years as confirmed males 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 five, all males, grew by more than 12 cm, the largest growth being 15.7 cm. 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, tortoises are clearly reproducing on the island and hatchlings are regularly relocated to the nursery, therefore there clearly are a number of mature females on the island. As such, it can be stated that the current classification system does not effectively identify females. It is suggested that other methods for determining gender should be explored, and from there, measurements could be continued into growth rates specific to males and females. For example, the entire Curieuse tortoise population could be assessed for gender 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, whilst 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 in any significant manner, with only a small number of sub-adult individuals having been located. 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 during 2018 allowed more tortoise nests to be found, largely explaining the increase of hatchlings in the nursery in 2018 compared to previous years, although there was not a large increase in hatchling discovery during 2019. 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. Installation of Goodnature A24 rat traps in the vicinity of the tortoise nursery during 2019 should hopefully have a positive impact on rat predation on individuals in the nursery, however to completely eliminate rat predation from the nursery, it is recommended that several A24 traps be installed in a significant radius from the nursery to provide a rat free buffer zone.

Conclusion

This study aimed to continue the annual census of the population of Aldabra giant tortoises relocated to Curieuse. This year's dataset included 134 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.

Whilst the free ranging population does not appear to be increasing significantly (apart from individuals donated from Praslin), 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 seven years is beginning to provide the basis of a good quality long term study. However, with only seven 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). Furthermore, 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 GVI 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 randomised locations along the north coast of Curieuse Island (Figure 6). Deployments were either shallow (≤ 10m depth) or deep (≥ 15m depth) site. Individual deployment

sites were chosen based on depth requirements and seabed topography, with flat substrates being ideal for deployment.

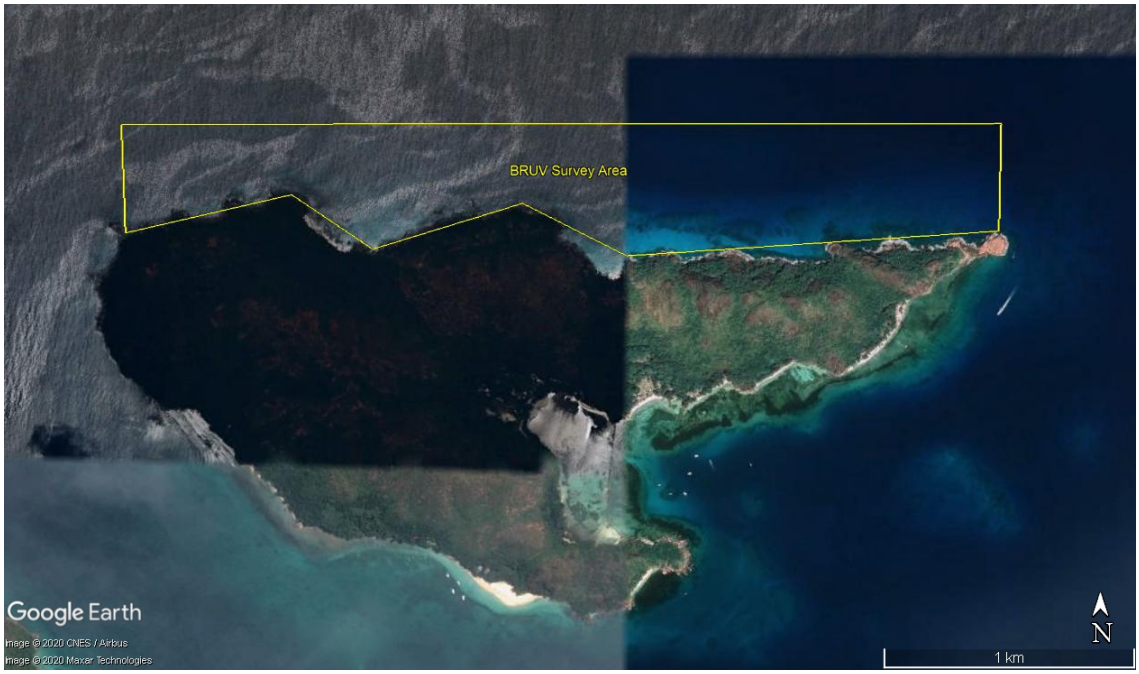

Figure 6. BRUV survey area off the north coast of Curieuse.

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 bigeye mackerel bait (*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 for shallow sites, and *Hawkeye Depthtrax 1H* depth sounder for deep sites).

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).

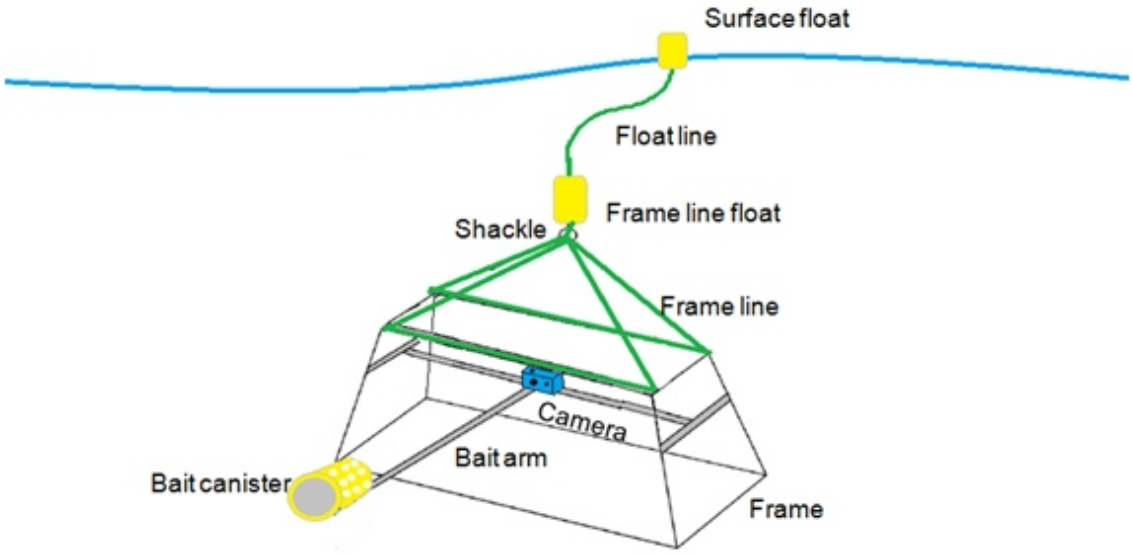

Figure 7. Depiction of a deployed BRUV unit.

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, remora 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.

Results

General

An even distribution of replicates across the study area at both shallow and deep sites were completed between the 2nd of April and the 13th of May 2019, a total of 32 deployments. A total of 109 target species were selected for monitoring and 79 were positively identified (Table 5). Water visibility averaged 10.9m (6 – 16m; ± 3.3SD) and video recordings averaged 59.4 minutes (45 – 60min ± 2.2SD). Shallow BRUV deployments were executed between 5 – 10m depth while deep

deployments were between 15 – 23m. 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.

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

Table 5. Target families and species monitored for in BRUVs during 2019. Species encountered and used in analysis are highlighted in yellow.

Relative abundance

MaxN values of species pooled over 22 categories across all sites indicate that emperor species had the highest relative abundance (pooled MaxN = 171), with the Redear emperor (*Lethrinus rubrioperculatus*) being the most commonly observed (pooled MaxN = 75, comprising 43.8% of emperor sightings). Rabbitfish species were the next highest in relative abundance (pooled MaxN = 111), with the Forktail rabbitfish (*Siganus argenteus*) seen most commonly (pooled MaxN = 81, 72.9% of rabbitfish sightings). Snapper species had the third highest relative abundance (pooled MaxN = 101), with the Red snapper (*Lutjanus bohar*) seen most commonly (pooled Max. N = 36, comprising 35.6% of snappers). Remoras, unicornfish, groupers, butterflyfish, angelfish and moorish idols were also commonly seen (pooled MaxN = 90, 90, 79, 66, 27 and 20 respectively), with all other species categories observed relatively infrequently (Max. N =< 25) (Figure 8).

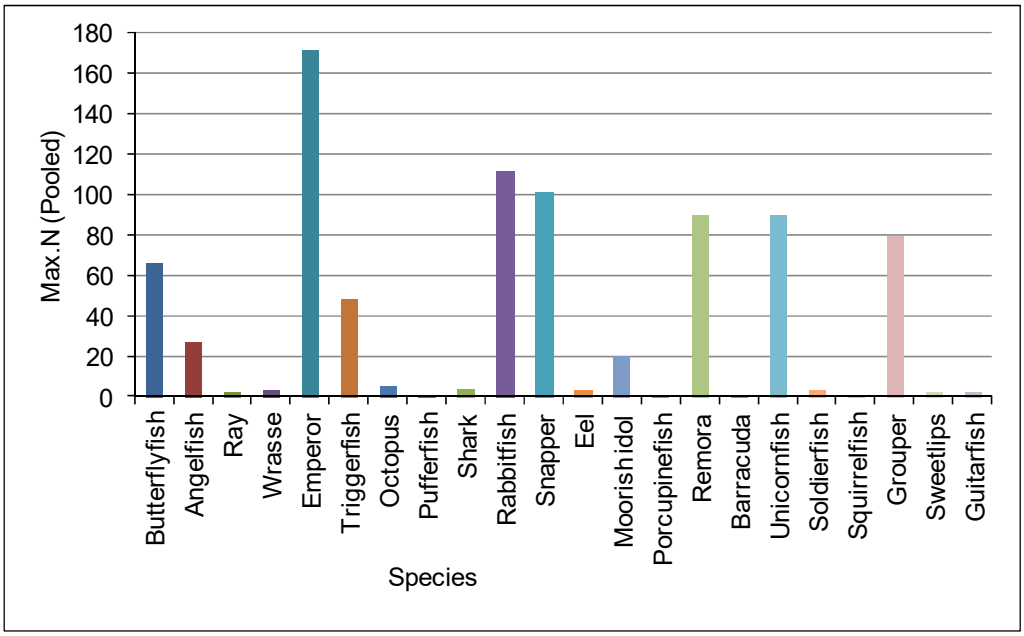

Figure 8. Relative abundance (MaxN) of species pooled over species category and BRUV deployments sites conducted along the northern shore of Curieuse Island.

By Substrate

Deployments on rocky substrate had both the highest total and average species seen (total 42; average 14.1), with rubble and sand/rock substrates having similar results (total 22, 23; average 6.3, 11.3 respectively). Deployments on sandy substrate recorded the lowest number of species (total

12; average 3.4) (Figures 9 and 10). A t-test found a significant difference between rocky substrate and others, but no significant difference between any other substrates.

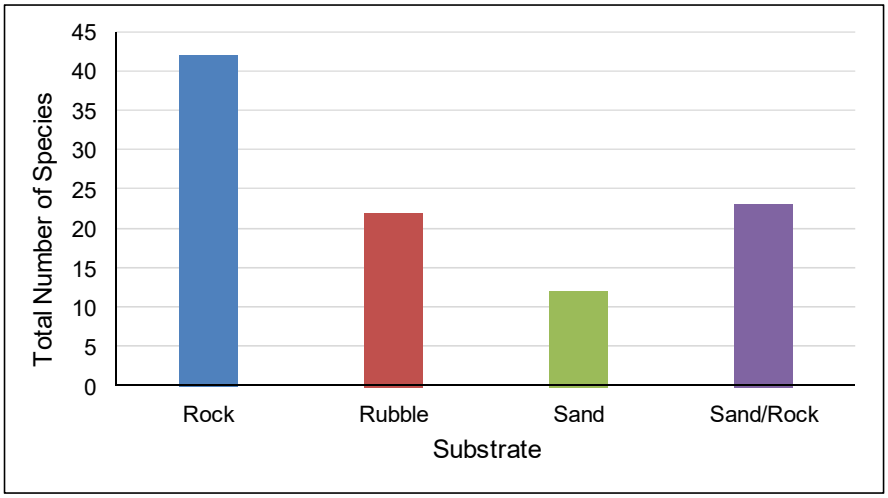

Figure 9. Total number of species found at deployments on substrates.

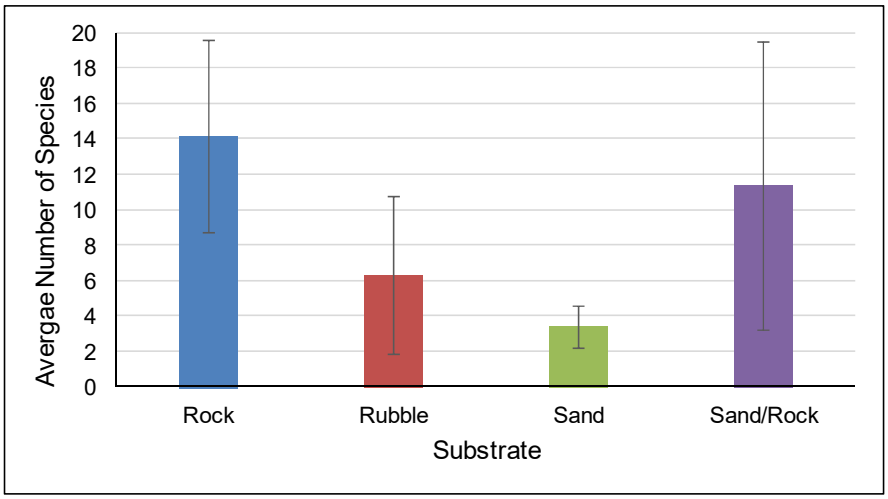

Figure 10. Average number of species found at deployments on substrates.

Emperor species showed high relative abundance across all substrate types, although rock was highest by some margin, with sand being the lowest. Rabbitfish species were more commonly observed on rubble and rock substrates, with relative low abundance on sand and sand/rock substrates, while snapper species had high relative abundance on rocky substrates and markedly less frequently on all others (Figures 11 and 12).

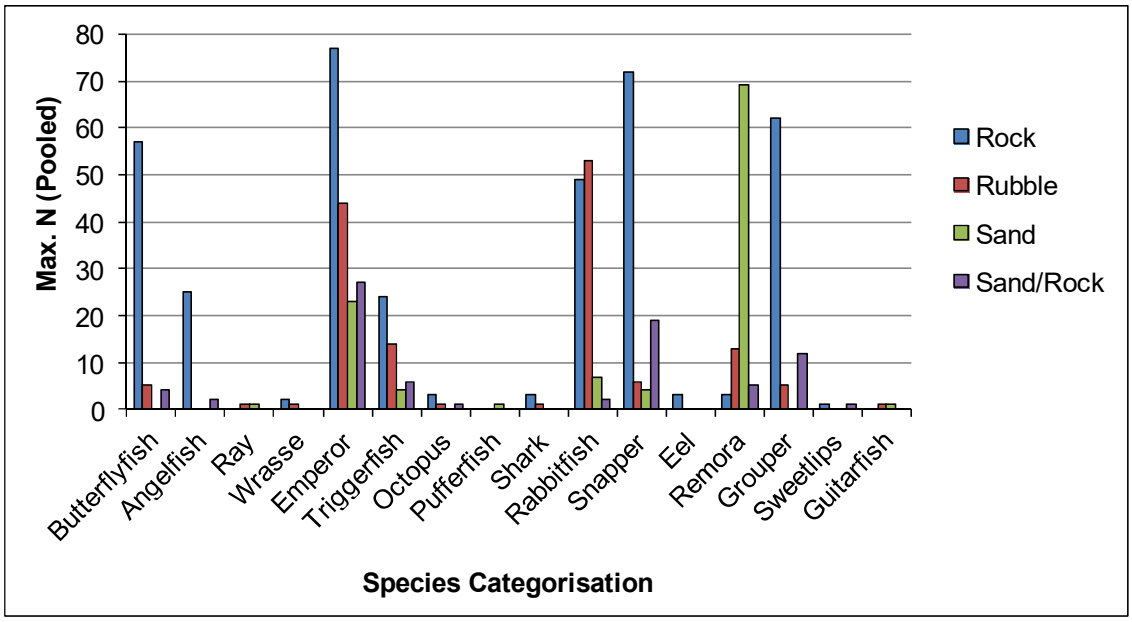

Figure 11. Total number of each species found at deployments on each substrate.

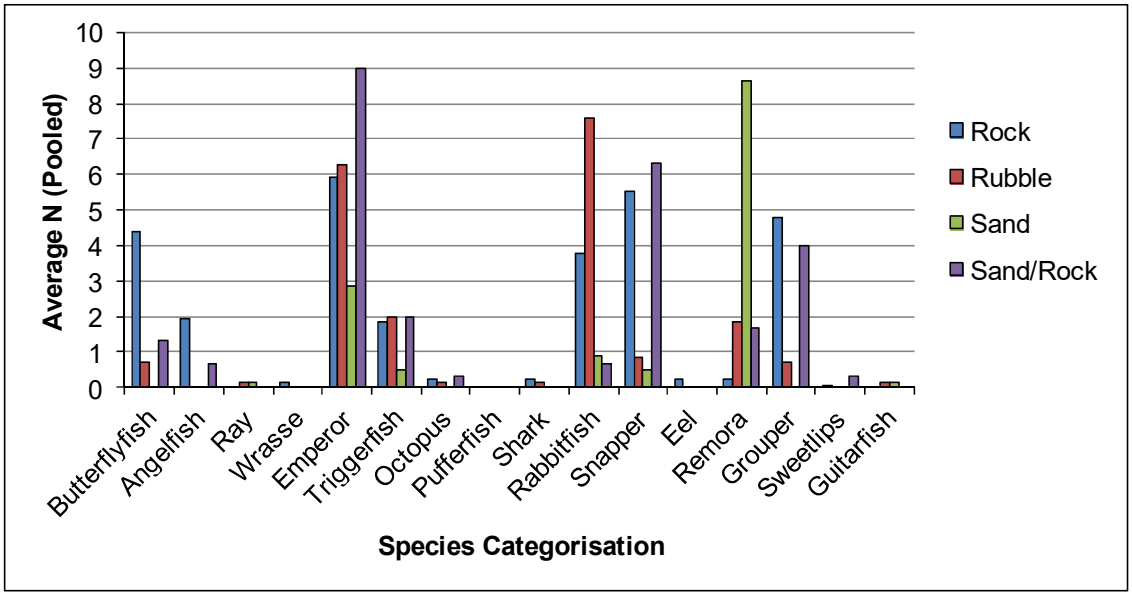

Figure 12. Average number of each species found at deployments on each substrate.

By Depth

Deployments at shallow sites showed higher relative abundances of species (total 44, average 15.5) than those at deep sites (total 32; average 6.4) (Figures 13 and 14). A t-test found a significant difference between both the total and average numbers of species at deep and shallow sites.

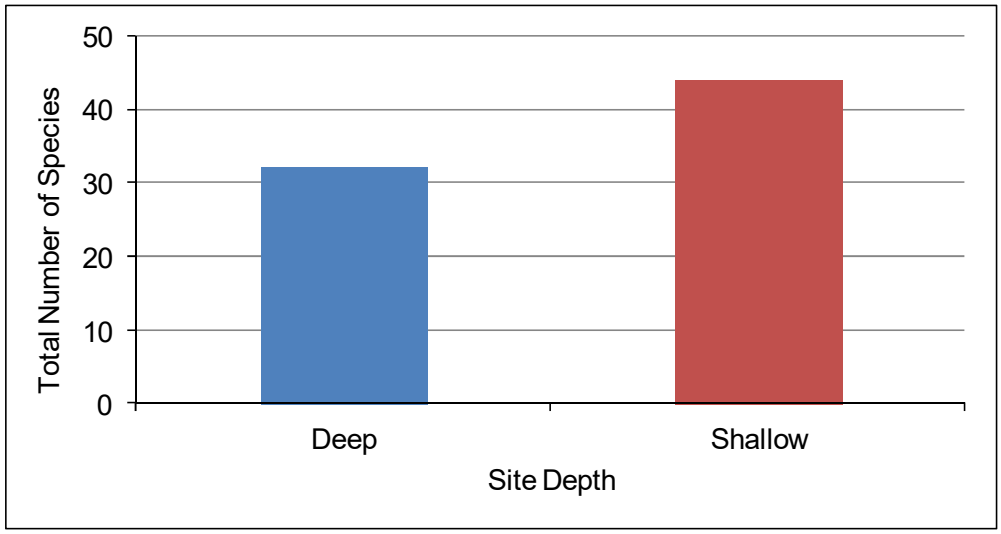

Figure 13. Total number of species found at deployments at both depths.

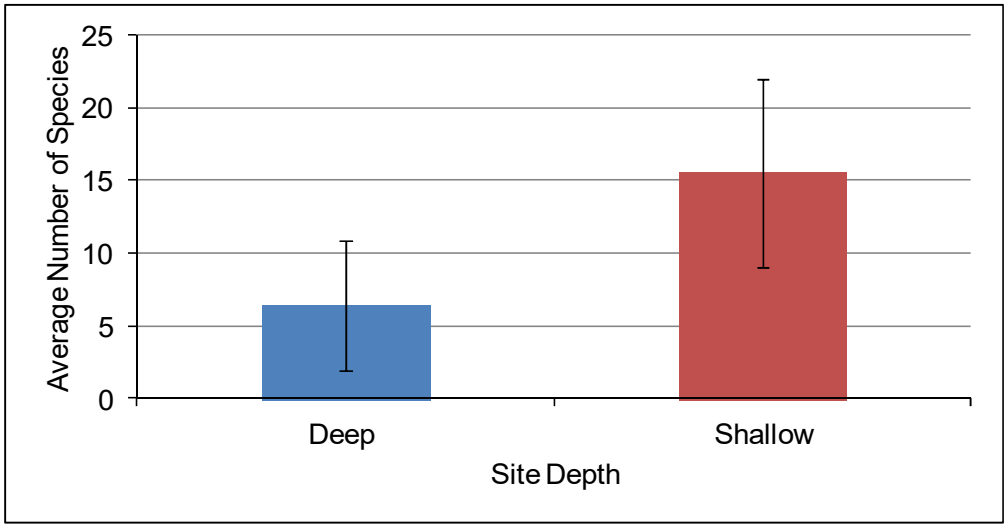

Figure 14. Average number of species found at deployments at both depths.

Emperor species showed high relative abundance at both deep and shallow sites (deep total 63, average 3.9; shallow total 108, average 6.75), while triggerfish were observed a similar amount at both (deep total 23, average 1.4; shallow total 25, average 1.6). All other species were found significantly more at one depth than the other, such as rabbitfish (deep total 16, average 1.0; shallow total 95, average 5.9) and snappers (deep total 8, average 0.5; shallow total 93, average 5.8), observed considerably more at shallow sites than deep, and remoras recorded significantly more at deep sites (deep total 82, average 5.1; shallow total 8, average 0.5) (Figures 15 and 16).

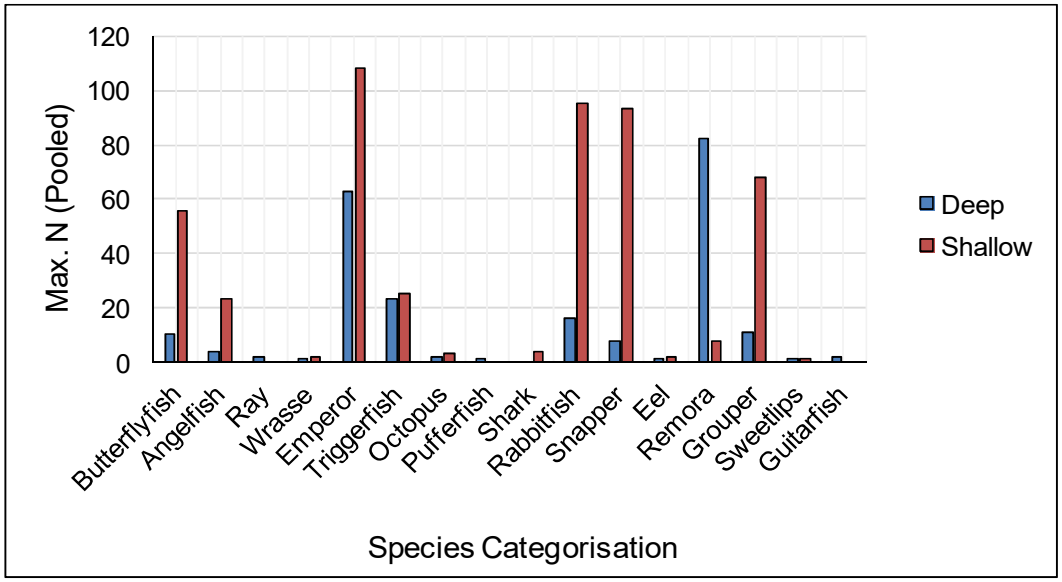

Figure 15. Total number of each species found at deployments at both depths.

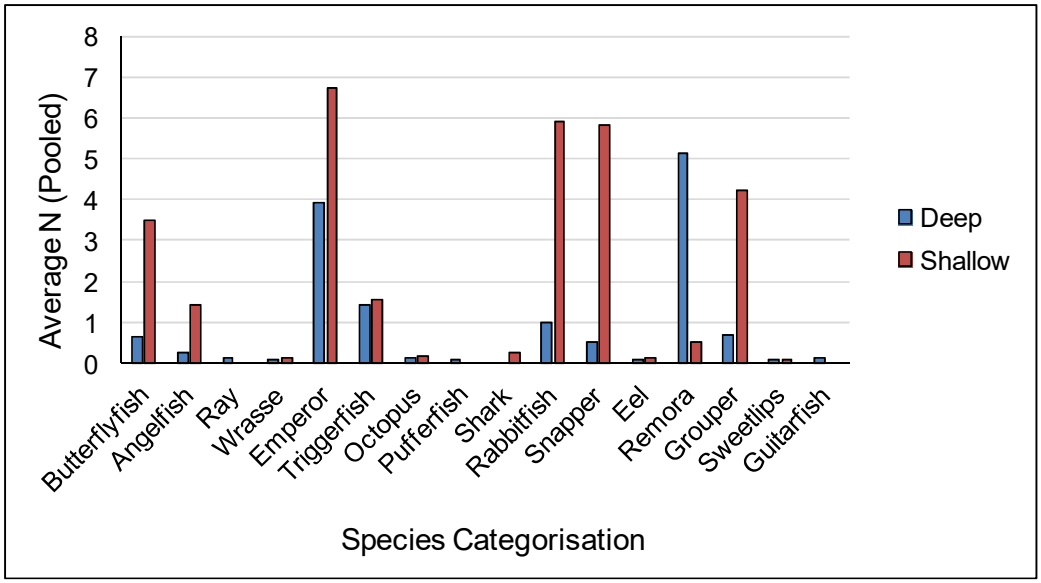

Figure 16. Average number of each species found at deployments at both depths.

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 have added to 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.

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.

This study has found that emperor species (family *Lethrinidae*) are by far the most abundant target species, which was also seen in the 2017 and 2018 surveys. 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, Seychelles fishing authority 2014); in combination with observing these species in elevated abundance in this study, it can be suggested that CMNP may play a role in promoting the overall Seychelles population stock of emperor, snapper, grouper and rabbitfish species, helping to ensure species population health and sustainable fisheries outside of CMNP. However, the lack of elasmobranchs is cause for concern as only one species was recorded in this year's surveys (Whitetip reef shark, *Triaenodon obesus*). 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. 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 was evidently the substrate as the difference in depth was relatively small. 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). As with 2018, it would therefore be inappropriate to discuss 2019 diversity results solely by depth,

while disregarding substrate, or vice versa. Consequently the results from the 2019 surveys can effectively compare species diversity over sand substrate by depth; results showing decreased species diversity over sand substrate by depth. This could suggest a lack of complex habitat (rock substrate) in neighbouring areas at deep sites because if present, 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 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 of Curieuse, 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 deep location sites and the low diversity.

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 if possible, 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 along the northern shore of Curieuse Island. A total of 79 species were positively identified out of 109 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, since 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. 2015). 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 fifth 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 17).

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. Due to the availability of four complete years of seasonal data on all the survey beaches since 2015, a decision was made to continue monitoring on an annual basis moving forward. Since the seasonal patterns of sand movement across the beaches now appear to be well understood, the accretion or erosion of the total quantity of sand on each beach should provide an adequate indication of long-term changes due to factors such as increased wave energy or sea level rise. Therefore the data presented in this report covers the period from January to June 2019 for the eastern beaches and January to July 2019 for the southern beaches.

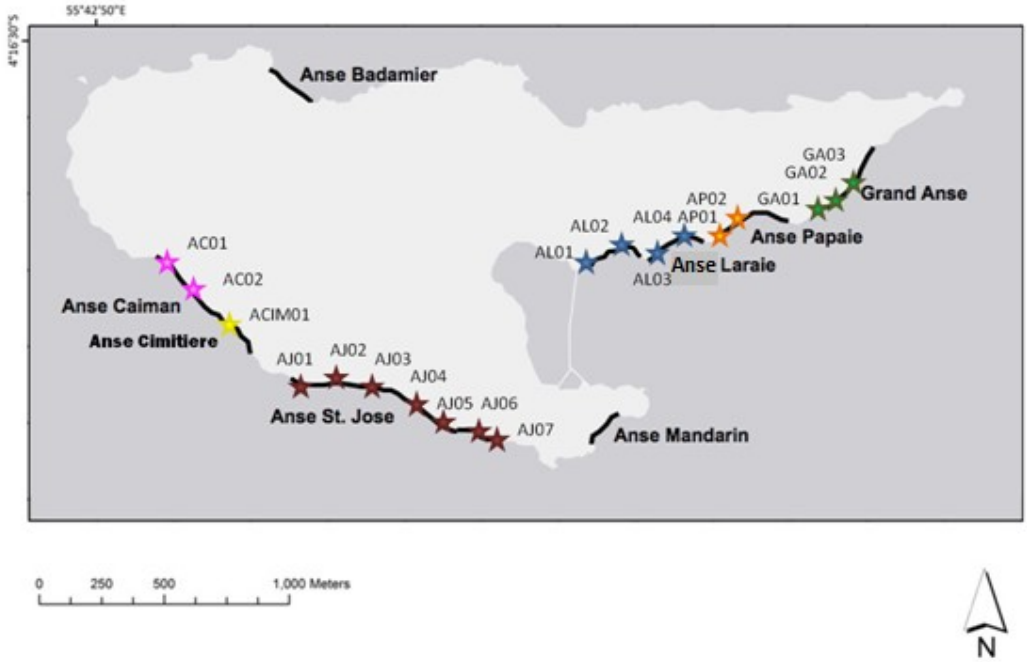

Figure 17. Approximate positions of the beach profiling transects along Anse Caiman, Anse Cimitiere, Anse St. Jose, Anse Laraie, Anse Papaie and Grand Anse in 2019 (note: AJ07 is no longer being monitored).

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. 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, 2018 and 2019 with four to six months of data collected for each of the six studied beaches. Previous reports, e.g. Beasley et al. (2018), 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 2018 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 fluctuated in area from January (25.67m²) to March (37.45m²), and then May (33.26m²) and again increasing in July (40.46m²) (Figure 18). Area progressively increased (by 57.61%) with moderate variations. Between January and May width slightly increased (by 2.08%) but dropped in July (by 6.64%). Anse Cimitiere also fluctuated in area from January (20.66m²), peaking in March (25.59m²), then decreasing (by 25.95%) in May (18.95m²). In July the area slightly increased again (by 8.92%, to 20.64m²). Width varied throughout the year (Figure 19), increasing from January (20.6m) to a peak in March (22.06m), decreasing again in May (18.39m) and further decreasing in July (17.29m). Anse Caiman and Anse Cimitiere are among the lowest for both beach width and area (along with Anse Papaie).

Anse St. Jose

Anse St. Jose decreased in area from January (52.72m²) to March (29.25m²) then increased (by 175%, with high variation) in May (80.51m²) (Figure 18). Area then further increased in July (by 29.05%, to 103.9m²). As with area, width showed similar variations from January (33.84m) to March (29.98m), then increased through May (36.06m) and July (39.72) (Figure 19). Individual transects on Anse St. Jose showed considerable variation between transects and months.

Anse Laraie

Anse Laraie displayed very little fluctuation in area with February displaying an area of 43.43m², 47.5m² in April and 37.62m² in June (Figure 18). Width declined through February (31.97m), April (28.98m) and June (25.4m) (Figure 19) representing an overall decrease of 20.5%, coinciding with the decrease in area.

Anse Papaie

Anse Papaie showed some variation between the months of beach profiling surveys. Overall, area decreased 23.35% from January (36.35m²) to April (27.95m²), then slightly decreased by 3.04% in June (27.1m²) (Figure 18). Similar to area, width was highest in January (23.7m) and decreased in April (19.35m) (Figure 19). In June width then increased by 9.04% to 21.1m.

Grand Anse

Grand Anse area values stayed moderately consistent throughout the surveyed months. Between February (49.2 m²) and April (39.73 m²) area decreased 19.24% and then increased again slightly by 7.15% in June (42.57 m²) (Figure 18). Again, width values remained relatively stable with the highest in February (32.83m) and the lowest value being recorded in June (25.83m) (Figure 19).

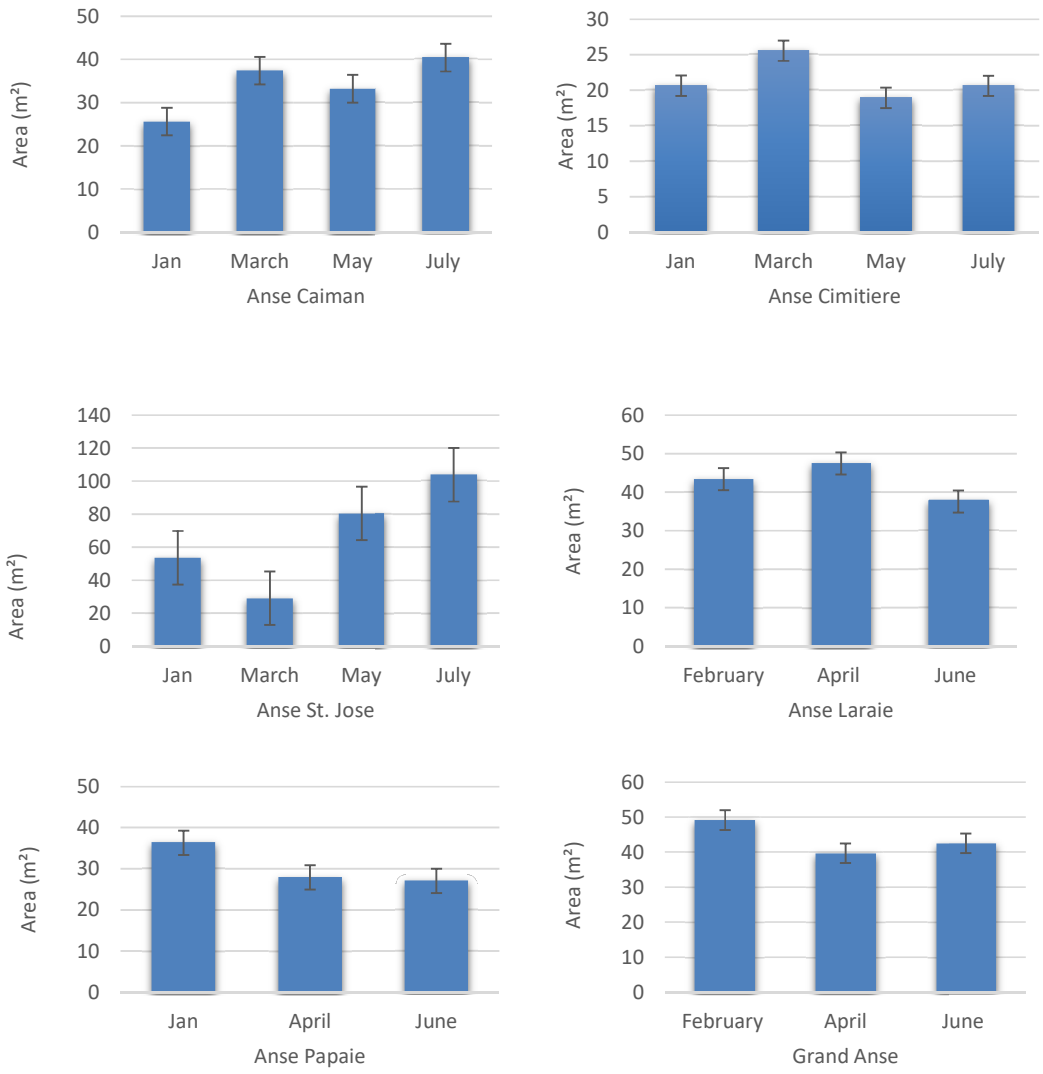

Figure 18. Mean area (m²) over 2019 for Anse Caiman, Anse Cimitiere, Anse St. Jose, Anse Laraie, Anse Papaie and Grand Anse.

Upon comparing average beach width among all beaches across 2016, 2017, 2018 and 2019 (Figure 20), the trend of decreasing width has continued on Anse Laraie and Anse Cimitiere, however the trends on the other beaches have reversed, with Grand Anse, Anse Papaie and Anse St. Jose having increased slightly in width, and Anse Caiman having decreased slightly. The greatest erosion since 2016 was observed on Anse Papaie and Grand Anse (losing 7.51m and 12.51m, respectively). Anse Papaie and Grand Anse were also among the top three in variation among transects and over time.

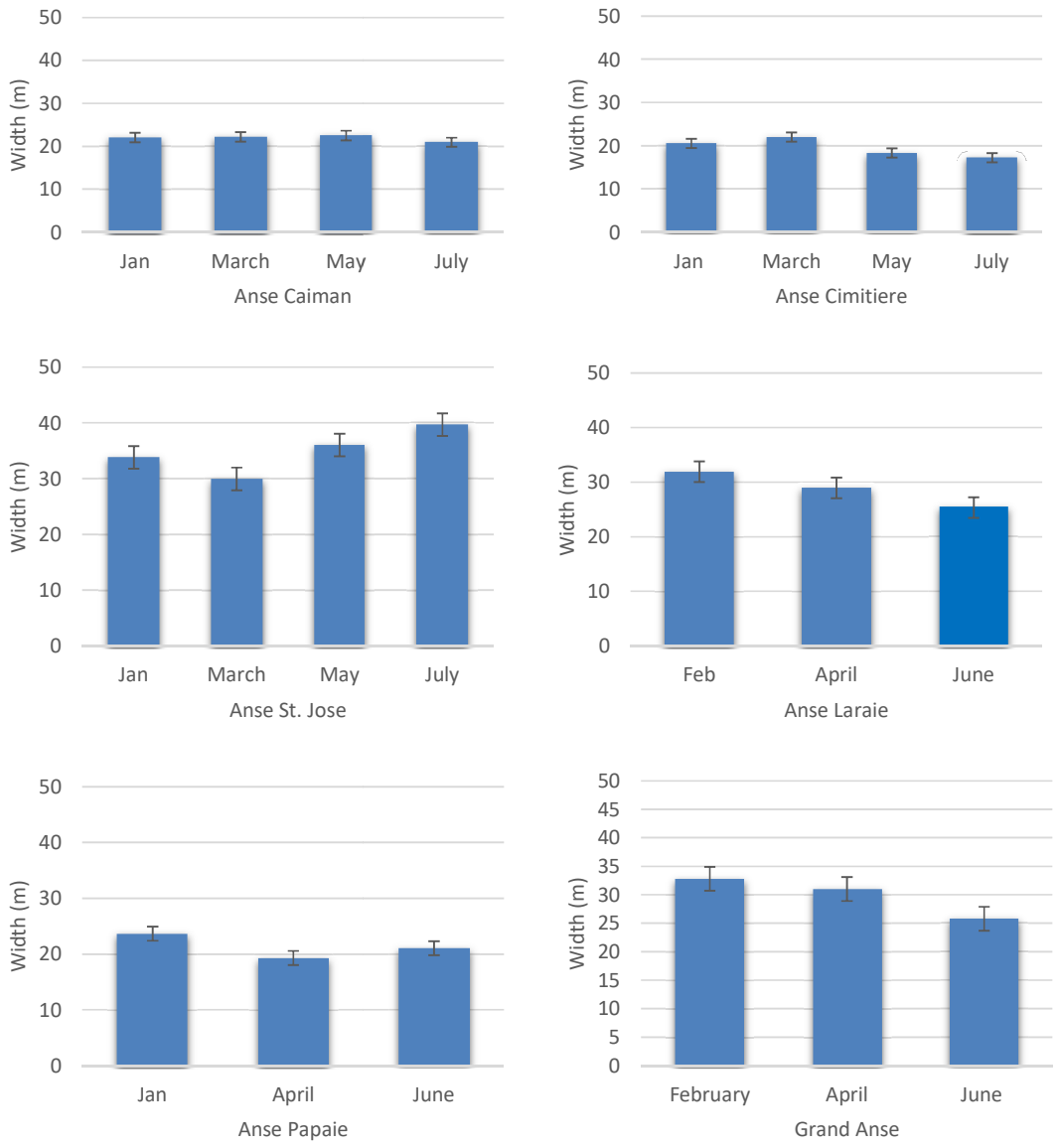

Figure 19. Mean width (m) over 2019 for Anse Caiman, Anse Cimitiere, Anse St. Jose, Anse Laraie, Anse Papaie and Grand Anse.

Mean Beach area (Figure 21) followed a similar trend, with Anse Caiman increasing by 8.99m2 since 2016 and the remaining five beaches decreasing in area on average between 2016 and 2018, however in 2019 all six of the beaches then increased in area. Overall, Anse Papaie and Grand Anse have had the leading decrease in area by 6.94m2 and 17.67m2 since 2016, respectively.

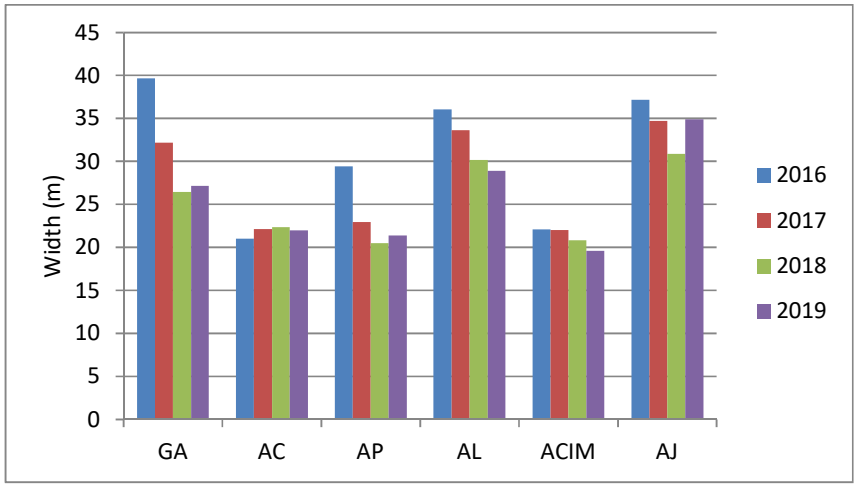

Figure 20. Mean beach width for all beaches over 2016 (blue bars), 2017 (red bars), 2018 (green bars) and 2019 (purple bars).

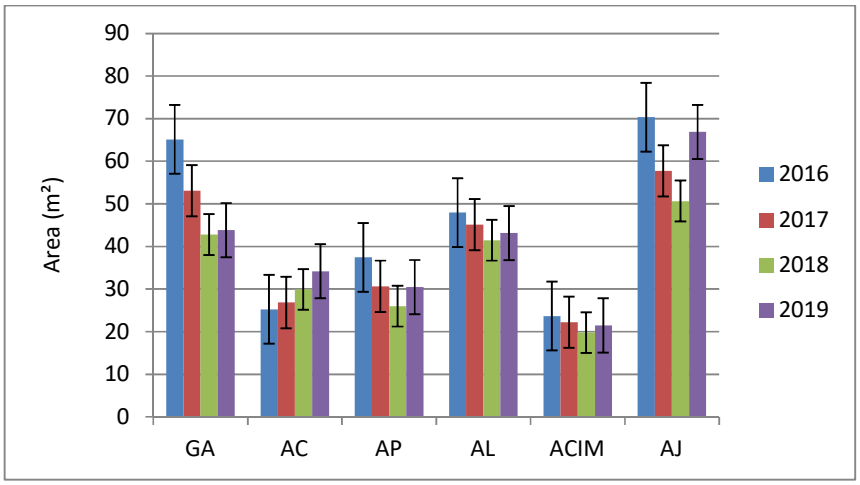

Figure 21. Mean beach area (± SEM) for all beaches over 2016 (blue bars), 2017 (red bars), 2018 (green bars) and 2019 (purple bars).

Discussion

The six beaches studied fluctuated in width and area throughout January and July 2019 due to seasonal variation in the individual transects based on the monsoon seasons. As highlighted in Beasley et al. 2018, 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 is 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, 2017, 2018 were again observed

in the first half of 2019. For this reason, beach transects were combined by beach for analysis in this report, as was done for the 2018 annual report. This way, a more concise overview of inter-annual changes to and variation in beach width and area is provided, which should be more practical for informing the management of CMNP's beaches. As of July 2019, due to having sufficient seasonal data, all beaches will now be surveyed annually to monitor overall changes in accretion and erosion. In future reports an in depth analysis will be conducted comparing years rather than months.

With the exception of Anse Caiman, in all previous years there was a general downward trend in both area and width. In 2019 however, all five beaches exhibited an increase in area whilst Grand Anse, Anse Papaie and Anse Jose also saw an increase in width. Although all six beaches increase and decrease in width and area annually, trends observed from 2016 to 2019 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. Moving forward, the beach profiling conducted by GVI will be strictly scheduled and will only take place during the months of June and July, usually over the span of two days in each month, and timed where possible to the lowest tides of the month. Storm events are often unpredictable and adding additional beach profiling surveys is not logistically possible at this time. However, dates of storm events were recorded in order to understand effects on beach profiles. 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. Following the cessation of monthly data collection in 2019, since December there has been significant and concerning erosion of many shorelines on Curieuse and elsewhere in the Seychelles, presumably as a result of higher sea levels and increased wave energy. This has been particularly prevalent on the southern beaches on Curieuse and as yet we do not have data to show whether the

total volume of sand has decreased, or whether the shoreline itself has simply receded. As such, data must be collected long-term in order to assess trends in beach width and area over time, and the continued annual monitoring will hopefully be sufficient to assess whether the volume of available nesting beach for turtles has remained relatively stable or if there should be greater cause for concern.

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 four 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 generally decreased in width and area between 2016 and 2019; however, a high level of variation is present in the data and in 2019 all beaches saw an increase in area. Nevertheless, these noticeable fluctuations in width and area highlight the importance of assessing long-term inter-annual changes in order to determine if any intervention is necessary, and the apparent increase in significant storm erosion events raises some concern regarding the viability of the Curieuse beaches as sea turtle nesting sites. 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.

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 (Hodgkiss et al. 2015).

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 22). 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 22. Location of mangrove quadrats 1 to 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. 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 2018 and 2019 have been included in this report.

10m x 10m 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 23). Between 2018 and 2019, *R. mucronata* exhibited an increase in average annual abundance in Quadrat 5, while Quadrats 1, 2, 3, and 8 showed a decrease. There was no change in average annual abundance of *R. mucronata* in Quadrats 4, 6 and 7. *B. gymnorrhiza* exhibited an increase in average annual

abundance between 2018 and 2019 in Quadrat 1 but decreased in Quadrat 4. Within Quadrats 3, 5, 6 and 7 there was no change in average annual abundance of *B. gymnorrhiza*. Between 2018 and 2019, *A. marina* exhibited no change in average annual abundance in Quadrats 1 and 2, but decreased within Quadrat 3.

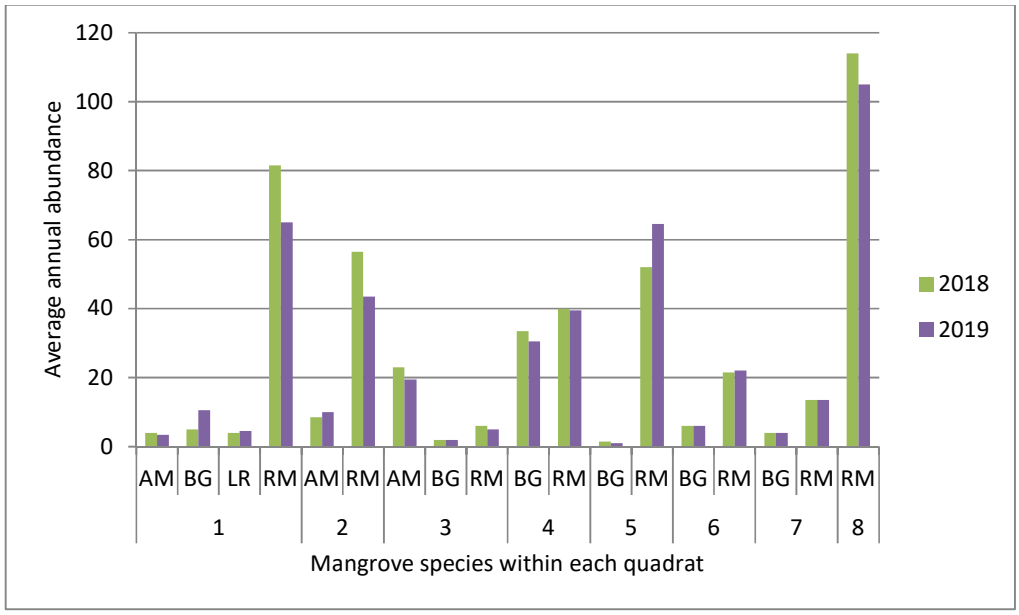

Figure 23. Distribution and average annual abundance for four mangrove species at eight 10m x 10m quadrat sites within the Baie Laraie mangrove forest during 2018 and 2019; AM= *Avicennia marina*, BG = *Bruguiera gymnorrhiza*, LR= *Lumnitzera racemosa*, RM= *Rhizophora mucronata*.

None of the quadrats contained *Xylocarpus* trees, and only Quadrat 1 contained *Lumnitzera racemosa*, exhibiting an increase in annual average abundance, from 4 to 5 plants per quadrat, between 2018 and 2019. The second most abundant tree species throughout the quadrats in the most recent survey (August 2019) after *R. mucronata* (n = 360) was *B. gymnorrhiza* (n = 56), followed by *A. marina* (total n = 31). This follows trends found in previous surveys.

1m x 1m 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 24). Quadrat 2 had the most seedlings averaging 30 and 22 (2018 and 2019 respectively), which is considerably more than the other quadrats. Quadrat 2 also contained the most saplings averaging 22 and 12 in 2018 and 2019 respectively. Overall, Quadrat 2 accounted for 63.8% and 69.5% (2018 and 2019 respectively) of all seedlings and saplings. There were no seedlings or saplings present in Quadrats 1 and 3.

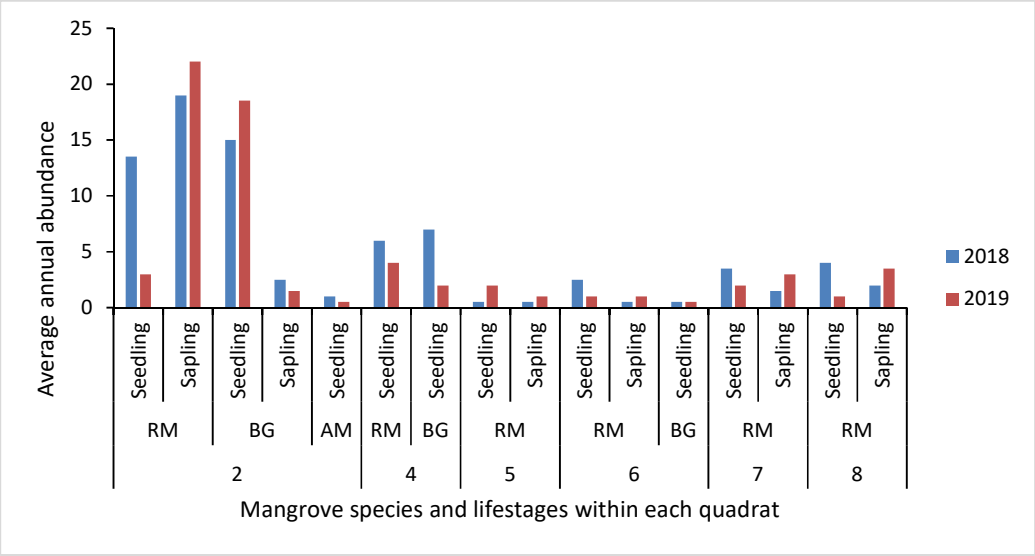

Figure 24. Seedling and sapling distribution and average annual abundance for three mangrove species at eight quadrat sites, containing four 1m x 1m quadrats, within the Baie Laraie mangrove forest during 2018 and 2019; AM= *Avicennia marina*, BG = *Bruguiera gymnorrhiza*, RM = *Rhizophora mucronata*.

R. mucronata had the most seedlings and saplings, with combined annual averages of 54 and 43 per quadrat site (2018 and 2019 respectively), compared to *B. gymnorrhiza*, with combined annual averages of 25 and 16 (2018 and 2019 respectively) (Table 6). Both *R. mucronata* and *B. gymnorrhiza* exhibited a decrease in annual average seedling presence (31 to 12, and 23 to 14 respectively). Annual average sapling presence also decreased in *B. gymnorrhiza* from three to two, but increased in *R. mucronata* from 24 to 31. None of the 1m x 1m quadrats contained any seedlings, saplings, trees or roots of *A. marina*, *R. mucronata,* or *B. gymnorrhiza*.

	R. mucronata	*B. gymnorrhiza*	*A. marina*
Seedlings			
2018	30.5	22.5	1.0
2019	12.0	14.0	0.5
Saplings			
2018	23.5	2.5	0.0
2019	30.5	1.5	0.0
Combined Average			
2018	54	25	1
2019	42.5	15.5	0.5

Table 6. Average annual abundance of seedlings and saplings of three mangrove species (*R. mucronata, B. gymnorrhiza,* and *A. marina*) at eight quadrat sites, containing four 1m x 1m quadrats, within the Baie Laraie mangrove forest.

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 2018 and 2019 was recorded in Quadrat 5, which is close in proximity to the seaward edge of the forests. These results likely indicate a continued increase in the density of this species in the seaward region of the forest following 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 a decrease in *R. mucronata* seedlings and *B. gymnorrhiza* seedlings and saplings, and an increase in *R. mucronata* saplings between 2018 and 2019. Quadrat 2 had the highest number of seedlings and saplings, for both the 2018 and 2019 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 seagrass 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 seagrass 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 in 2016 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 seagrass 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 their state, it is vital that mangrove monitoring continues in order to better understand, protect, and rehabilitate the area.

Rat Eradication

Introduction

Prior to the colonisation of Seychelles by humans in the 18th century, the islands had been isolated from the continents for 75 million years. This allowed the evolution of many varied endemic species

with few natural predators. Specifically the absence of rats, a voracious and adaptable fast reproducing omnivorous species, led to the islands providing a safe terrestrial environment for the successful propagation of animal groups such as ground nesting birds and lizards. Throughout the human history of Seychelles however, rats have been progressively introduced to many of the islands of the archipelago (Cheke 2010). This has led to a striking difference between islands free of rats such as neighbouring Cousin or Aride, both of which sustain extensive populations of species such as the White-tailed tropic bird, Wedge-tailed shearwater, Brown and Lesser noddies, Wright's skink and Giant millipedes, and islands with a rat population such as Curieuse, on which there are far fewer of such species, and of those that are present, individuals are often greatly reduced in size.

Black rats (*Rattus rattus*), also known as Ship rats, have been present on Curieuse since shortly after colonisation (Hill et al 2003), and they are known to be present in every habitat on the island. Smaller than the closely related Brown rat or Norway rat (*Rattus norvegicus*), they are excellent climbers and thrive in arboreal environments (CAB International 2014) such as the Seychelles granitic islands, usually staying high in the trees during the day and foraging on the ground at night up to 15m from their home tree.

There have been previous unsuccessful attempts to eradicate rats from Curieuse Island using poison (Merton et al 2002), which until recently was the only consistently effective method of large scale eradication, or control in localised areas. There are many risks associated with the use of poison, including potential secondary poisoning of native fauna and contamination of the environment (Courchamp et al 2003), and there have long been concerns regarding the welfare of the targets (Cowan & Warburton 2011). Furthermore, rodent poisons have a latency, which allows some continued movement and effects of rats, and there is no control over the final resting place of carcasses. Traditional trapping methods are extremely labour intensive in high density areas with daily trap inspections necessary for animal welfare reasons, and active euthanization and disposal of carcasses is required where live trapping is used. Moreover, traditional traps are single use and require to be manually reset once activated, and regularly re-baited.

However, a novel technology has recently been developed and made widely available, which does not require the use of poison, is extremely efficient in cost and labour once installed, and is extremely effective. The A24 rat trap, developed by Goodnature New Zealand (Figure 25), uses a chocolate based bait to entice rodents into a chamber where they are instantly euthanized by means of a bolt driven by compressed CO_2 gas, at which point the trap will then automatically reset. The bait is contained in an automatic lure pump, which dispenses bait over a six month period, and each

CO_2 canister will fire the trap at least 24 times. Once installed, each trap will continue to be effective with very little maintenance for up to six months in the presence of rodents, and far longer in well controlled areas. In addition, each trap is supplied with an automatic counter to monitor how many times a trap has been activated, and rodent detector cards for further monitoring of the presence or absence of rodents.

Figure 25. Goodnature A24 automatic humane rat trap.

A24 traps have already been tested and deployed in two areas in Seychelles. Some success has been experienced in the Vallee de Mai, and traps have also been effectively deployed at the Ranger Station at Baie Laraie on Curieuse. A well designed array of A24 traps has the potential to maintain an area entirely free of rodents for the duration of their deployment at very low cost and effort following installation.

The project also has unlimited scope to be expanded to eradicate rats from increasingly large areas of Curieuse Island, allowing the native flora and fauna to recover and thrive, and re-colonisation by species currently absent due to rat predation. Furthermore, the opportunity exists to examine the resulting changes in the biota and highlight the benefits of such efforts to a wide range of stakeholders.

This project represents the most comprehensive examination of the potential for A24 traps to be used to control rat populations in extended localised areas in Seychelles environments, and of their

potential for complete rat eradication on islands. With funding secured from the Environment Trust Fund, 50 A24 traps and associated supplies have been purchased and deployed in an area of productive and diverse habitats on the south coast of Curieuse.

Aims

The aims of the rat eradication project are as follows:

1. To eradicate rats from an area of coastal forest and wetlands 200m x 100m (two hectares) on the southern coast of Curieuse.
2. To maintain the rat free status of the area for a period of one year initially with intensive monitoring, and continued maintenance indefinitely.
3. Assessment of the efficacy of A24 traps in eradicating rats from localised Seychelles environments.
4. Assessment of the viability of using A24 traps for complete eradication of rats from island ecosystems.
5. To provide protected area managers and other island managers with a strategy for systematic and sustained control or eradication of rats from sensitive habitats in Seychelles.
6. To prepare the deployment area for a potential assessment of the effects of rat eradication on the flora and fauna of coastal forest on Curieuse Island.

Methodology

Study Area, Transects and Trap Deployment

Figure 26 shows the area selected for the eradication study, which was selected based on several criteria. The coastal forest contains some of the highest diversity of sensitive habitats on Curieuse, including mature forest (the site of the recent translocation of the critically endangered Seychelles paradise flycatcher *Terpsiphone corvina*), riverine habitat, wetlands containing mangroves and the critically endangered Yellow bellied mud terrapin *Pelusios castanoides*, scrubland above the forest boundary, and human habitation on the GVI base. This combination of habitats provides an extremely diverse range of areas for analysis of the effectiveness of the eradication under varying circumstances, and for the analysis of the effects of rat eradication. Furthermore, its proximity to the GVI base allows an examination of the influence of human habitation on any such attempts to eradicate rats, as well as allowing a greater frequency of data collection due to the convenience of the location. The area is 200m long by 100m wide, representing a total area of two hectares.

Figure 26. Rat eradication study area showing the habitats contained within.

66 trap stations were selected, arranged on a grid of six longitudinal transects, with trap stations and transects spaced at 20m intervals (Figure 27). Transects were prepared by first selecting a reference point (Trap Station 2) and attaching an eye, to which a 180m long transect line was tied. The transect line was deployed on a precise compass bearing of 287 degrees, and trap stations identified at 20m marks. If a suitable tree for mounting a rat trap was found within 2m of the transect line mark, that tree was selected as the trap station, and if not, a wooden stake was driven into the ground to serve as the trap station. Transects were cleared of invasive vegetation (primarily Cocoplum and Cinnamon) where necessary to allow access. A vertical reference transect was prepared from Trap Station 2 on a compass bearing of 17 degrees to provide reference points for laying out Transects 2 to 6.

The project was carried out in four phases. In Phase 1, seven A24 rat traps were deployed with bait but without gas to assess the likelihood of any effects on native species. Trap Stations 15, 24, 29, 42, 47, 50 and 63 were selected in order to assess all habitats in the study area. The distribution of traps in the first active trapping phase, Phase 2, intensively covered Transects 1 to 4 with traps spaced at 20m intervals, and extra traps placed at Trap Stations 1A, 7A, 9A, 12A, 23A and 24A (Figure 28). Upon completion of Phase 2, traps were spaced out to 40m intervals in the interior and traps relocated to Transect 5 for Phase 3, maintaining a border of two rows of traps at 20m spacings (Figure 29). During Phase 4, the final trapping phase, further traps were spaced out to 40m intervals

and relocated to Transect 6, again maintaining a border of two rows of traps at 20m spacing (Figure 30). During Phases 3 and 4, two additional traps were deployed on the GVI base, either in the volunteer kitchen, kit room or staff house.

Figure 27. Rat trap transects showing all 66 trap stations, plus extra traps deployed, with Transect 1 being at the seaward edge and Transect 6 the most landward.

Figure 28. Locations of rat traps during Phase 2.

Figure 29. Locations of rat traps during Phase 3.

Figure 30. Locations of rat traps during Phase 4.

Phase 1 – Assessment of Effects on Native Species

Between 21/1/19 and 13/4/19 at the seven locations selected, A24 traps were deployed with bait lures installed but no CO_2 canisters. Cuddeback C2 or E2 camera traps were pointed at the rat traps to record any activity from rats or native species. Traps were checked regularly and SD card data downloaded. Videos and images were analysed to confirm whether rats were indeed entering the traps, and to assess the likelihood of any bycatch once the rat traps were activated.

Phases 2, 3 and 4

Phase 2 commenced on 15/4/19 with the deployment of all 50 A24 traps along Transects 1 to 4 at 20m spacing in the arrangement depicted in Figure 24. Traps were checked daily during the first two weeks then twice weekly for the rest of the phase. Counter readings were recorded at each trap and carcasses returned to the GVI base to be sexed, measured, and stomach contents removed. Counter readings were correlated against the reading from the previous check, and any which had not recorded a trap having fired (i.e. a carcass was present but the counter hadn't advanced) had their readings corrected by flicking with a finger. Bait lures were removed to check there was still bait present in the neck of the lure, and to visually examine inside the trap. Any potential issues were rectified, and if there was any suspicion a trap may be malfunctioning, it was test fired by activating the trigger with a pencil. Any traps where the counter had reached 24 had the CO_2 canister replaced and the counter reset. Five camera traps were deployed at varying locations throughout the active trapping phases, SD cards were replaced on each trap check, and footage immediately analysed to inform the decision making process regarding trapping strategy, or to indicate any issues requiring attention. Traps were pre-baited with a small smear of bait underneath the trap chamber, in order to entice rats to investigate the trap.

Data was immediately entered into a database and examined for consistency. Traps where carcasses were found were recorded as confirmed kills, however a significant number of carcasses were not located. Formulae were devised to allocate unconfirmed trap firings for each set of traps checks to either rats or bycatch and produce estimated rat and bycatch totals:

Estimated Rat Total = (Total Trap Firings – Total Carcasses) x (Rat Carcasses/Total Carcasses)

Estimated Bycatch Total = (Total Trap Firings – Total Carcasses) x (Bycatch Carcasses/Total Carcasses)

Phase 3 was initiated on 13/5/19, with traps redeployed in the arrangement shown in Figure 29, Phase 4 was initiated on 18/6/19 with traps redeployed in the arrangement shown in Figure 30, and

is ongoing. The process of trap checks and data collection for Phases 3 and 4 were identical to Phase 2, except that pre-baiting was only conducted on the top two transects once the inner area was essentially free of rats. This was to reduce the rates of bycatch. Other methods of reducing bycatch included the development and installation of excluder devices fitted in and below the trap chamber, and the application of Vaseline to the inside of the excluder to prevent skinks in particular gaining traction and being able to enter the traps.

Rodent detector cards, plastic cards with a small amount of chocolate bait inside, were deployed at 35 trap stations towards the end of Phase 2. These cards record bite marks from any species attempting to eat the bait, with the intention that the species can often be identified by the bite patterns using a visual guide produced by Goodnature. These cards are an additional method of confirming whether any rats are still present in an area, and can also be used prior to the beginning of any control attempt to locate the best positions to install traps.

Results

Phase 1

Much rat activity was detected on all traps throughout the phase. Phase 1 was intended to last for a duration of one month, however towards the end of the month Seychelles skink interactions with the traps began to be detected, including several entering the traps. It was decided to extend Phase 1 to allow development of methods of excluding or deterring skinks from entering the traps. A total of approximately 12,500 camera trap photos and videos recording fauna interactions were analysed over the duration of the phase.

Several versions of skink excluder devices were developed and tested in a process lasting a further month. Excluders were fabricated by shrinking plastic bottles over a foam form, and were designed to fit snugly against the inside of the bait chamber to produce a smooth surface and prevent skinks gaining purchase on the inside of the chamber. The excluders also protruded 25mm from the bottom of the bait chamber in a bell shape, and were secured in the bait chamber by means of string. Discussions have been held with Goodnature regarding development of a proprietary excluder, and designs have been provided to allow them to assess the possibility of production, seemingly a relatively straightforward process once a design has been finalised. All versions of the excluders were somewhat effective at preventing skinks gaining access to the bait chamber, with the final version almost completely effective. Application of Vaseline to the inside of the excluder proved fully effective. A complication arose however, in that it appears rats will spend much time licking the

Vaseline off the excluders in preference to entering the traps to access the bait, therefore removing the effect of the Vaseline and reducing the likelihood of rats entering and triggering the traps. It was decided to proceed to the active eradication phase of the project since the ability of skinks to enter the traps appeared greatly reduced by the excluders alone, and Vaseline would be applied following the initial reduction in the rat population to reduce the incidence of rats licking the layer of Vaseline off the excluders.

Phase 2

Data for confirmed and estimated rat mortalities during each week of Phase 2 are presented in Table 7.

	Confirmed Rat	Estimated Rat
Week 1	35	54
Week 2	15	26
Week 3	4	6
Week 4	3	6
Total	57	92

Table 7. Confirmed and estimated rat mortalities during Phase 2.

The first night of trapping resulted in 18 confirmed rat mortalities, with four unaccounted trap activations, making for an estimated total of 22 rats, with the second night producing five confirmed mortalities. An analysis was conducted to determine the proportion of rats trapped in the outer two rows of traps versus the inner traps, in order to assess when eradication was complete i.e. the ratio of inside/outside trap mortalities dropping to zero, with only continuing mortalities from reinvasions on the outside traps. The results of the inside/outside trap analysis are presented in Table 8. The ratio had dropped to zero by Week 3 and remained so during Week 4, confirming that the interior of the study area was free of rats. During Week 4, 35 rodent detector cards were also deployed throughout the interior of the study area to provide additional confirmation that no rats remained. However it was immediately clear that although there were many different bite marks on the detector cards, no suitable guide exists to be able to interpret bite marks from Seychelles species since guides have only been produced by Goodnature for species from other areas of the world. Therefore in the absence of any definitive detector card data it was decided to proceed to Phase 3, and a study to produce reliable data on bite mark patterns for Seychelles species was planned to allow their use in the future.

	Outside Traps	Inside Traps	Inside/Outside Ratio
Week 1	32	3	0.0938
Week 2	14	1	0.0714
Week 3	4	0	0
Week 4	3	0	0
Phase	53	4	0.0755

Table 8. Ratio of confirmed rat mortalities at inside versus outside traps during Phase 2.

Figure 31 shows a heatmap depicting the relative rates of confirmed rat mortalities at the individual trap stations during Phase 2. This provides a clear picture of the relative densities of rats in the various habitats during the phase.

It can be seen that there were significantly higher mortalities on the outer two rows of traps, indicating that rats were continually attempting to reinvade the area, but in most cases were euthanized by traps on the outer one or two rows.

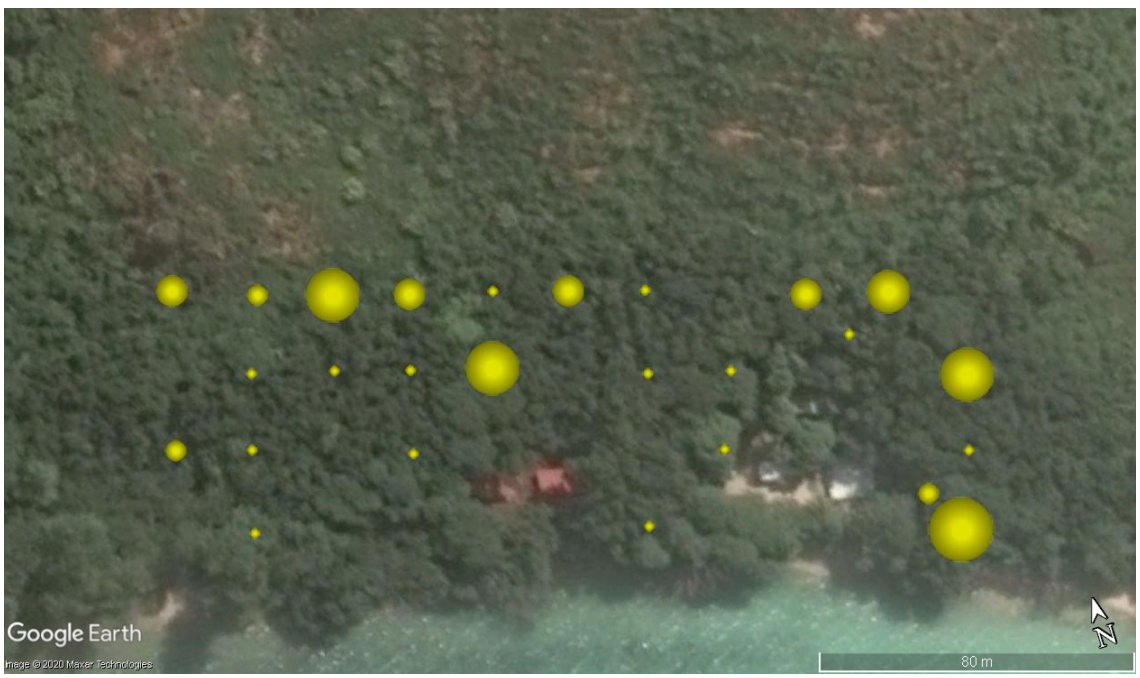

Figure 31. Heatmap of confirmed rat mortalities at each trap station during Phase 2.

Phase 3

Data for confirmed and estimated rat mortalities during each week of Phase 3 are presented in Table 9.

	Confirmed Rat	Estimated Rat
Week 1	3	8
Week 2	5	10
Week 3	7	12
Week 4	5	17
Week 5	3	12
Week 6	2	3
Total	25	62

Table 9. Confirmed and estimated rat mortalities during Phase 3.

A similar inside/outside trap mortality ratio analysis was conducted for Phase 3 to determine when eradication was complete, the results of which are presented in Table 10. As expected, the ratio was greater than zero at the beginning of the phase and dropped to zero during Weeks 2 and 3, however one individual was found at Trap Station 37 on Transect 3 during Week 4, therefore Phase 3 was extended for a further two weeks, during which the inside/outside ratio remained at zero, so it was decided to proceed to Phase 4.

	Outside Traps	Inside Traps	Inside/Outside Ratio
Week 1	1	2	2.0000
Week 2	5	0	0
Week 3	7	0	0
Week 4	4	1	0.2500
Week 5	3	0	0
Week 6	2	0	0
Phase	1	2	0.1364

Table 10. Ratio of confirmed rat mortalities at inside versus outside traps during Phase 3.

Figure 32 shows a heatmap depicting the relative rates of confirmed rat mortalities at the individual trap stations during Phase 3.

There was a similar pattern of mortalities as in Phase 2, however it can be seen that although the vast majority of mortalities were on Transect 5 and the outer two rows on the sides, there were a number of reinvasions to the interior of the study area. Of particular note is that two rat mortalities occurred on the trap in the main kitchen on the GVI base, well inside the eradication area, where food odours are constantly present due to meal preparation for all personnel on a daily basis.

Figure 32. Heatmap of confirmed rat mortalities at each trap station during Phase 3.

Phase 4

Data for confirmed and estimated rat mortalities during each week of Phase 4 up to the end of December 2019 are presented in Table 11.

	Confirmed Rat	Estimated Rat		Confirmed Rat	Estimated Rat
Week 1	8	19	Week 16	9	27
Week 2	8	23	Week 17	3	13
Week 3	19	36	Week 18	5	16
Week 4	17	28	Week 19	3	19
Week 5	13	23	Week 20	2	12
Week 6	21	25	Week 21	3	14
Week 7	15	27	Week 22	6	17
Week 8	8	19	Week 23	14	29
Week 9	5	17	Week 24	2	7
Week 10	7	13	Week 25	2	6
Week 11	8	19	Week 26	1	4
Week 12	9	16	Week 27	3	7
Week 13	7	18	Week 28	10	19
Week 14	5	23	Week 29	4	7
Week 15	7	18	Total	224	521

Table 11. Confirmed and estimated rat mortalities during Phase 4.

Continued inside/outside trap mortality ratio analysis was conducted for Phase 4, the results of which are presented in Table 12. It can be seen that the ratio for each week was usually significantly less than one, indicating that most of the rat mortalities were on traps on the outside two rows, however throughout the phase there have been continued low numbers of mortalities on inside traps. This confirms that a small proportion of reinvading rats will regularly not be euthanized by the first one or two rows of traps they encounter, even though the majority will be.

Again the pattern of mortalities indicate that the vast majority of reinvading rats were euthanized by the outer two rows of traps, with most mortalities on the traps on Transect 6, but there were clearly occasional but regular reinvasions to the interior of the eradication area, and the longer duration of Phase 4 has provided further evidence of the effects of food odours, with continued mortalities in the GVI main kitchen, and also the staff house kitchen, and in the kit room where stocks of bait were stored.

	Outside Traps	Inside Traps	Inside/Outside Ratio		Outside Traps	Inside Traps	Inside/Outside Ratio
Week 1	8	0	0	Week 16	8	1	0.1250
Week 2	7	1	0.1429	Week 17	2	1	0.5000
Week 3	15	4	0.2667	Week 18	5	0	0
Week 4	17	0	0	Week 19	3	0	0
Week 5	13	0	0	Week 20	2	0	0
Week 6	17	4	0.2353	Week 21	2	1	0.5000
Week 7	8	7	0.8750	Week 22	5	1	0.2000
Week 8	8	0	0	Week 23	11	3	0.2727
Week 9	5	0	0	Week 24	2	0	0
Week 10	7	0	0	Week 25	2	0	0
Week 11	6	2	0.3333	Week 26	0	1	1.000
Week 12	7	2	0.2857	Week 27	3	0	0
Week 13	3	4	1.3333	Week 28	10	0	0
Week 14	5	0	0	Week 29	4	0	0
Week 15	6	1	0.1667	Phase	191	33	0.1728

Table 12. Ratio of confirmed rat mortalities at inside versus outside traps during Phase 4.

Figure 33 shows a heatmap depicting the relative rates of confirmed rat mortalities at the individual trap stations during Phase 4.

Figure 33. Heatmap of confirmed rat mortalities at each trap station during Phase 4.

All Phases

A summary of confirmed and estimated rat mortalities for all active trapping phases is presented in Table 13.

	Confirmed Rat	Estimated Rat
Phase 2	57	92
Phase 3	25	62
Phase 4	224	521
Total	306	675

Table 13. Combined confirmed and estimated rat mortalities during all active trapping phases.

Figure 34 shows a heatmap depicting the relative rates of confirmed rat mortalities at the individual trap stations over the duration of the project.

It can be seen that there was a consistently relatively low rate of mortalities throughout the majority of the study area over the duration of the project, with the exception of the outer two rows, which have a consistently higher rate of mortalities, which represents continued attempted reinvasions into the eradication area.

Figure 34. Heatmap of confirmed rat mortalities at each trap station during all trapping phases.

In order to estimate the total number of rats present in the area at the beginning of the study, it is necessary to determine how long it takes before all individuals resident in the vicinity of any particular trap station at the time of deployment have been euthanized. Figure 35 shows a plot of the number of mortalities in each week following trap activation over the first eight weeks of deployment, pooled for all trap stations. It can be seen that mortalities were significantly higher during the first two weeks, and thereafter levelled off. Therefore it can be assumed that mortalities during the first two weeks of trap activation are a reasonable representation of the total population of rats present at the beginning of trapping in any given area. Fluctuations in mortality rates from Week 3 onwards are assumed to show cycles of an increasing and decreasing population outside the eradication area and continuing mortalities from reinvasions, primarily on the outside two rows of trap stations.

The total number of rats present in the eradication area prior to the commencement of trapping can be calculated as the sum of the number of confirmed rat mortalities at each trap station during the first two weeks of trapping, and applying a correction to account for carcasses not located:

Total Rat Population = Confirmed 2 week mortalities x (Total estimated mortalities/Total confirmed mortalities)

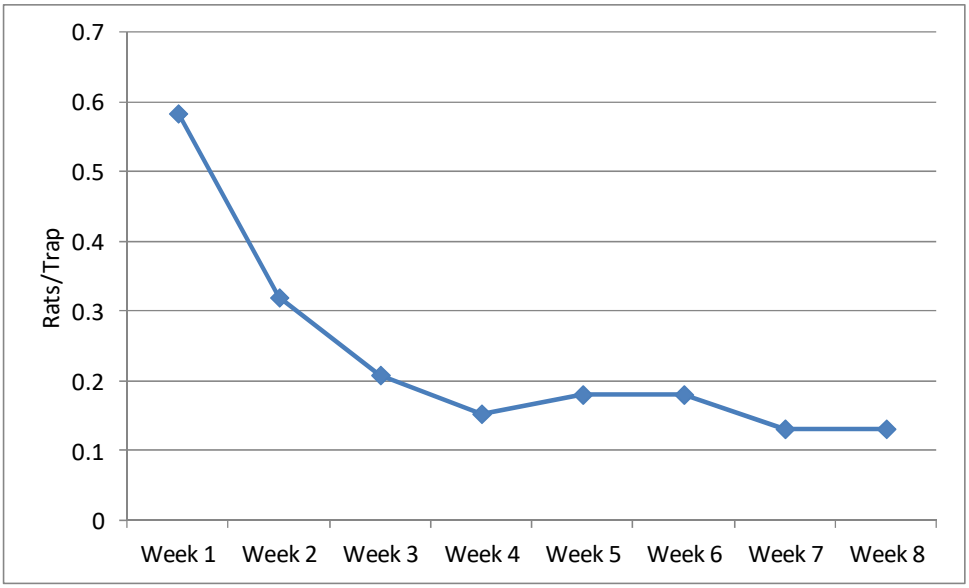

Figure 35. Plot of rats/trap for the first eight weeks following deployment.

This yields an estimate of 66 x (675/306) = 146 individuals, or 73 individuals per hectare. A preliminary estimate of the entire rat population on Curieuse would therefore be 73 x 286 hectares, or approximately 21,000 individuals. A detailed analysis of the trapping mortalities in the various habitats in the eradication area, projected across the entire island, would provide a higher resolution estimate of the island population, however Figure 36 shows the relative densities of mortalities during the initial two weeks of trapping at each trap station, giving some indication of the original rat population densities in the habitats contained in the study area.

Scavenging

Many observations have been made regarding scavenging of carcasses. It is clear that only a proportion of carcasses have been recovered during transect checks. This was expected prior to the start of the project, and rates of scavenging appear extremely high in certain areas, with carcasses often quickly being removed from the vicinity of traps, one particular observation confirming that a carcass can vanish within the space of a couple of hours. A number of observations have been made of carcasses being easily dragged and consumed by large centipedes, and observations of centipedes in the vicinity of traps or even inside traps have been relatively common. Hermit crabs have also frequently been observed scavenging carcasses, and it is assumed they may also remove entire carcasses. In the wetlands area, which has a dense population of Giant mangrove crabs, few carcasses have been discovered. Prior observations suggest that these crabs will kill or scavenge rats,

and camera trap observations of high crab activity and the density of crab burrows likely explain the lower number of carcasses recovered from this area.

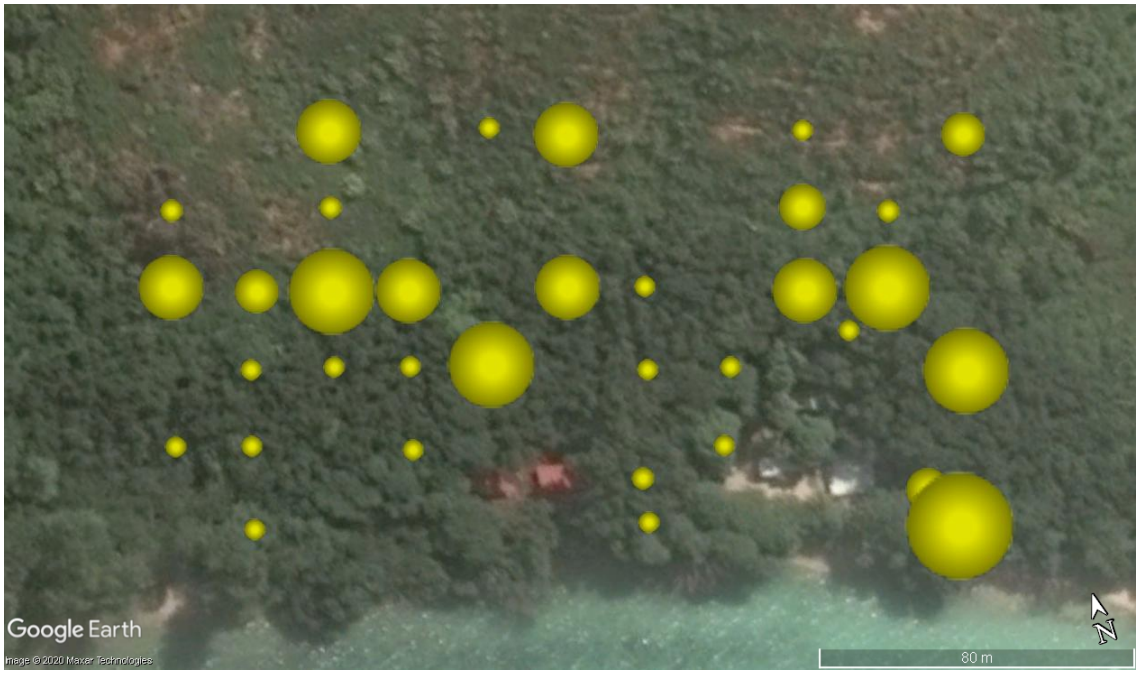

Figure 36. Heatmap of relative rat mortalities at each trap station during the first two weeks of trapping.

Stomach Contents, Size Range and Gender Ratio

A total of 76 stomach contents were collected. 10 samples were qualitatively analysed to provide an early indication of common prey items, and preliminary identifications were made to the lowest taxonomic level. All 10 samples contained significant numbers of soil dwelling nematodes of the families Diplogastridae, Aporcelaimidae and Qudsianemitidae. Significant quantities of Cocoplum fruit and Cinnamon seeds were identified, along with three species of ant, *Cardiocondyla minutior*, *Mesoponera melanaria*, and the invasive Yellow crazy ant, *Anoplolepis gracilipes*, remains of invertebrates including beetles and the American cockroach, *Periplaneta americana*, and an embryonic bird leg.

The average body size of measured rats was 13.9cm (n = 68), range 9.3 to 18.0cm, and the gender ratio was 35 males to 34 females (n = 69).

Rodent Detector Cards

A study was conducted with several species likely to leave marks on the rodent detector cards in order to produce a guide to identifying the species responsible. Individuals were contained in plastic crates with detector cards and monitored until they had attempted to eat the bait contained in the cards leaving initial bite marks. Cards were then photographed in high resolution and returned to the crate to capture further heavier bite marks, at which point they were again photographed. Species used for the study were Seychelles skink *Trachylepis seychellensis*, Giant mangrove crab *Cardisoma carnifex*, hermit crabs *Coenobita brevimanus*, *Coenobita rugosus*, ghost crabs *Ocypode cordimana*, *Ocypode ceratophthalma*, Giant centipede *Scolopendra subspinipes*, and the Giant African land snail *Achatina immaculate*. During the later stages of the project, detector cards will be used to aid in confirming the absence of rats within the study area, and photographs will be used to identify any bite marks detected. The data, photographs and original detector cards will be provided to Goodnature UK who have agreed to produce a Seychelles specific guide to interpreting detector card data.

Bycatch

Despite comprehensive monitoring prior to the activation of the A24 traps and the development and testing of skink excluder devices, there has been some bycatch. The second night of trapping resulted in four confirmed Seychelles skink mortalities and one Wright's skink. Confirmed skink mortalities numbered approximately one per day during Phase 2, which was of significant concern, especially with regards to the Wright's skink. This however was the only individual of that species lost through the duration of the study. Bycatch of skinks showed a steady decline over the course of Phase 2, and by the beginning of Phase 3 the confirmed skink bycatch had reduced to one per week, and zero by the end of Phase 3. Phase 4 has seen variable rates of confirmed bycatch with an average of less than two per week.

As the study has progressed there have been a number of incidences of bycatch of other species, primarily hermit crabs and mangrove crabs, although the majority of these seem only to have resulted in the loss of a claw which will regenerate. Hermit crab bycatch appeared to peak during the wetter month of December. Three invasive myna birds also ended up as bycatch.

Discussion

The rat eradication experiment has been tremendously successful in reducing the population of rats in the interior of the study area to zero the majority of the time. A24 traps clearly represent a significant advance in the technology available for controlling or eradication rat populations, and this experiment will provide an invaluable contribution to the knowledge of how it can be used in this type of environment. Upon the conclusion of the project we will have answered many questions in detail, which will inform the future of the project on Curieuse, and provide other NGOs or government agencies with the information required to conduct similar attempts at eradicating or controlling rat populations using this technology.

Prior to any large scale deployment of A24 traps in any new environment it is essential that studies are conducted to ensure that any negative effects on native species are reduced to a minimum. For the planned duration of Phase 1 of this study (one month), there were clearly high densities of rats present in all areas and it appeared that there would be no effects on native species, however at the end of the phase Seychelles skink interactions began to be detected, including entry into the chamber of the traps. Detailed analysis of camera trap footage suggested that individual skinks have the ability to learn, and can be very persistent in their efforts to circumvent any measures put in place to prevent trap entry. A process of excluder development resulted in a device that proved completely effective for most individuals, however it is assumed that certain individuals have more of a propensity to thoroughly investigate the traps, and for those individuals it can be difficult to prevent trap entry altogether. The skink bycatch data clearly shows that rates have been highest immediately following trap deployment in any particular area, followed by a steep decline to very low levels. It is assumed that the initially higher rates were composed of those individuals with a propensity to investigate the traps. The single Wright's skink mortality was initially a cause for concern, however discussions with SNPA and the Ministry of Energy, Environment and Climate Change resulted in a decision to continue the project, but that the situation would be reviewed again if there were any further Wright's skink mortalities. Furthermore, it was agreed that the benefits of the project vastly outweigh the risk of the loss of much smaller numbers of mostly fast growing and abundant native species. Over the course of the project the proportions of bycatch shifted significantly more towards hermit crabs, with the vast majority representing the loss of a single claw rather than mortality. Decapod limbs will regenerate at the next moult (Juanes & Smith 1995), therefore this was determined to also be an acceptable risk, and rates appeared to fluctuate with peaks during wetter months, and almost zero during drier months. Other methods of reducing bycatch included analysis of the small scale topography around each trap and modification of trap

position, including raising the position to prevent entry directly from the ground, continued analysis of camera trap footage, periodic application of Vaseline to excluders, and constant monitoring of bycatch and camera trap data to inform decision making.

The active trapping phases have produced some very interesting results, and highlighted the extreme complexity of such a deployment of traps. It quickly became apparent that only a proportion of rat carcasses were being found on each trap check, since counter readings indicated significantly higher mortality than the number of carcasses discovered. Over the course of the project many observations have been made of scavenging by several species. Hermit crabs and centipedes have provided by far the most common observations of scavenging, and it is clear that in certain locations (e.g. the wetlands) carcasses will rarely be found. The formula devised for estimating the total number of rat mortalities is assumed to be relatively accurate, since the proportion of rat mortalities to bycatch is quite accurately known. A more detailed study involving closer observation of the relative scavenging times for each individual species could increase the accuracy of the estimate, however this would require significant resources and would likely not be practical.

The data for Phase 2 clearly showed a logarithmic decline in rat mortalities over time, indicating progressive removal of rats from the area with slow rates of reinvasion, and that the distribution of traps effectively resulted in an essentially rat free area. Similar patterns were evident in the mortality rates for Transects 5 and 6 following first trap deployment. The analysis of confirmed rat mortalities at inside versus outside traps has proved extremely effective in showing when initial clearance was complete, and as a method of monitoring reinvasion rates.

The production of heatmaps has also been a very useful method of visualising patterns of mortalities and assessing the effectiveness of the strategy, and the distribution of rats throughout the study area at each stage, along with patterns of reinvasion. The Phase 2 heatmap shows a more even distribution of mortalities, although even on a timescale of one month there was already a pattern emerging of higher total mortalities at traps on the outside two rows, giving an early indication of reinvasion patterns. The heatmaps for Phases 3 and 4 show a dramatically lower mortality rate at traps on the inside of the area, confirming that the strategy has been highly effective in virtually eliminating rats from the interior of the study area. Furthermore, it can be clearly seen that the majority of mortalities over the course of the project have been on the outside two rows of traps, indicating that the majority of mortalities following the initial decline in population represent attempted but failed reinvasions.

Reinvasion to the interior of the eradication area was however a constant feature. Whilst the chocolate formula bait at 20m spacing was fully effective for most individuals, a small number of individuals did bypass the outer two rows and were then euthanized by traps on the inside. A few individuals were also euthanized by traps in the volunteer kitchen, staff house kitchen, and kit room (where stocks of chocolate bait were stored). These individuals often proved extremely difficult to trap, and we believe this is clear evidence of a well known principle in any rat control attempts, in that in the presence of alternative food sources, the chosen bait will be relatively less attractive, and also if a rat's preferred food source is available it may be impossible to trap it. If any individuals were detected in these areas and the chocolate bait was not proving effective, raw or cooked chicken was substituted with very quick success on all occasions. This suggests that a pattern of alternating bait types in field transects, particularly something seemingly as effective as chicken, could result in a vast reduction or complete elimination of any reinvasions. Those individuals which were not attracted enough to the chocolate bait were all quickly euthanized using chicken bait.

Another factor may be at play considering reinvasions. Large quantities of food are prepared on a daily basis on the GVI base, primarily in the volunteer kitchen, and it is strongly suspected that odours from cooking may be drifting downwind causing rats to ignore the odours from the chocolate bait in the outer traps as they follow the cooking odours, another example of a way in which bait may be made relatively less attractive thus reducing the likelihood of an individual being trapped. It is planned to add an expanded phase of trapping to test this theory. Upon the conclusion of Phase 4 in March 2020, the study area will be expanded from 200m x 100m to 280m x 120m, giving a total area of 3.36 hectares. Once the remaining traps in the interior of the area are spaced out to 40m this will only require the installation of a further 15 traps (65 total). It is hoped that the extra distance from the GVI cooking facilities will allow sufficient dispersal of food odours, reducing their attractiveness to a point where individuals will again be preferentially attracted to the chocolate formula in the traps. It may be however, that a very small proportion of individuals are simply not sufficiently attracted to the chocolate formula, and for them alternating bait types would be the only obvious solution. Should the expanded area still be insufficient to prevent all reinvasions to the interior, and invading individuals still appear to be attracted in by cooking odours, comparing Phase 4 and 5 reinvasion data should allow a regression to be plotted to estimate the minimum required area to entirely eliminate the effect of food odours drifting downwind.

Analysis of mortality data for the first two weeks following trap installation in each location has provided what we believe to be an extremely accurate estimate of the total population of rats present in the area prior to this study, at least comparable to, if not more accurate than traditional

index trapping or mark-recapture techniques, since this represents a removal method of population estimation rather than projections of estimated numbers via calculation following sub-sampling of a population. The preliminary estimates presented here (73 individuals per hectare, 21,000 island wide) are well within the range that may be expected from this type of habitat (Harper et al 2015, Russell et al 2011), and further analysis will be conducted to refine the estimates and assess their accuracy.

The same two week data can be used to estimate the distribution of rat density in the various habitats throughout the study area. This also followed plausible patterns (Figure 32), with relatively high densities in the more productive, moist and shaded areas of mature forest, higher estimated densities in the riverine area to the east of the study area, and lower estimated densities in the dry, less shaded and nutrient poor scrubland above the forest boundary. Data for the wetlands to the west of the study area may be compromised by the higher estimated scavenging rates here, as it might be expected to produce higher estimated densities than other habitats.

The preliminary stomach contents analysis has already yielded some interesting results. The most common prey item, present in all samples analysed, was nematode worms. Some previous studies have suspected nematodes as being parasitic, however this study has confirmed that all nematodes were in fact soil dwelling species and were therefore prey items. The fact that nematodes seem to form such a large proportion of the diet may be a strong indication of a degraded environment, since other more preferred prey items (e.g. bird eggs or chicks) are no longer in abundance due to the long established presence of rats. The preliminary analysis has also raised some interesting questions regarding the relationship between rats and other invasive species. Invasive Cocoplum and Cinnamon fruits and seeds were commonly found in the samples. Some seeds were whole but some were chewed and no longer viable. It is suspected that rats play a strong part in the spread of invasive plants on Curieuse due to the dispersal of seeds, however the picture may be more complex, with the destruction of a proportion of seeds through chewing. A further question has been raised due to the discovery of invasive Yellow crazy ants in some samples, and how much effect rats may have on controlling the populations of invasive ants. A full literature review and investigation of Black rat diet, along with detailed analysis of all stomach contents collected, should provide a much clearer picture of the typical diet of rats on Curieuse, a comparison with their diet in other environments, and will hopefully answer many of the questions raised by this study.

There have been many anecdotal early indications of potential ecosystem recovery, primarily through personal observations of GVI personnel, some of whom have had an intimate knowledge of

the local area for many years. Sightings of species such as the Wright's skink have been significantly more common in recent months, and in areas they have not previously been seen. Sightings of Giant millipedes have been more common, and species not previously reported from the area are now being seen, these include some seabird species and butterflies. Whilst it is far too early to make any definitive judgements and there could be many other factors at play such as significant changes to the shoreline in the last year due to erosion, the initial signs are encouraging, and dedicated ecological study over time will hopefully show clear evidence of ecosystem recovery and the return of native species.

Conclusion

This project represents the first large scale intensive ecosystem level study of the use of Goodnature A24 rat traps in order to completely eradicate Black rats from a localised area in Seychelles. The information learned from the study has the potential to be used to infinitely scale up the size of eradication area, even to achieve complete island eradication. Regardless of the outcome of the study with respect to the limits of the technology when deployed in this manner, the results will be incredibly useful in informing the design and management of other similar future eradication or control attempts.

Upon the conclusion of the study it will be important to continue to maintain the array of rat traps to maintain the study area effectively free of rats. It is hoped that by providing a long term rat free environment in the sensitive and diverse coastal forest and wetland habitat on Curieuse that long term ecosystem recovery will follow.

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 – 1982 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.

The waters surrounding Curieuse are home to both Green and Hawksbill turtles, as the reefs and seagrass beds provide ample food resources. 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 Curieuse 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 to 24 hours old, and 2 being 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 14). 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.

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 37).

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

Table 14. Nine categories of sea turtle emergence types.

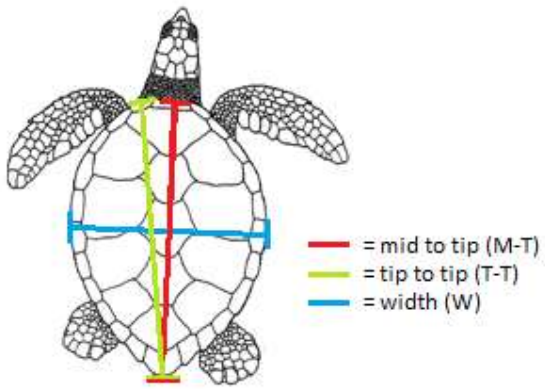

Figure 37. Measurements taken for each turtle encountered.

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 was carried out when the turtle had completed laying and the covering process had begun. Tags fitted during both the 2014 – 2015 and 2015 – 2016 seasons were 'SCA' series, while mostly 'E' series tags were fitted during the 2016 – 2017 season. Tags fitted during the 2017 – 2018 season were mostly 'SXX' series, as were all during the 2018 – 2019 season. The location of the eggs was triangulated as above. Once an activity had been recorded, marks were scored through all tracks so they would not be mistaken for new tracks in subsequent surveys.

Nesting female population size estimate was calculated by dividing total number of nests by a bracketed mean of three to four clutches per female per season for Hawksbills, and three to five 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 15. 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.

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

Table 15. Nest excavation categories and definitions.

Results

This report contains a summary of the complete 2018 – 2019 nesting season. The 2020 annual report will contain a summary of the 2019 – 2020 season, once it has been completed.

Nesting Adults – Hawksbills

The peak of the nesting season occurred in December, with 34% of all activities recorded (Figure 38). The total number of activities for this season was 506, of which 234 were nests, resulting in a population estimate of 59 – 78 individuals (Table 16). Grand Anse had the highest percentage of laid nests of the 2018 – 2019 season with 68% (Figure 39). The lowest percentage of laid nests was on Anse St. Jose with only 2%. These findings follow previous season trends.

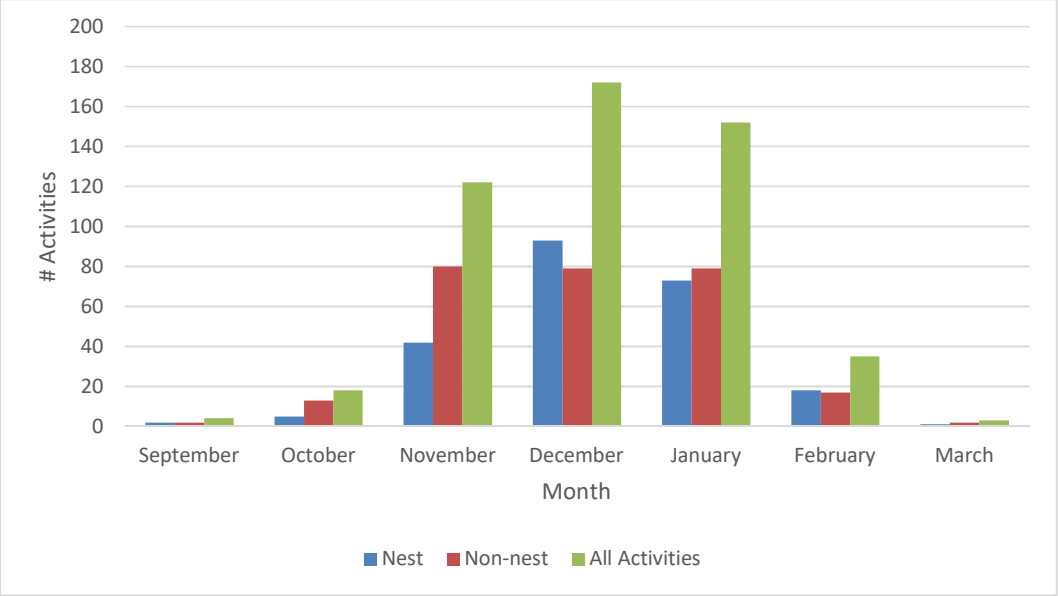

Figure 38. Number of activities by month (nest, non-nest and all activities) for the 2018 – 2019 Hawksbill season.

Beach suitability was compared by assessing nesting success on each beach (Figure 40). The higher the proportion of successful nesting activities compared to aborted nesting activities (i.e. non-nests), the higher the beach suitability. The highest percentage of successful nesting rates were on Anse Papaie, Anse Laraie and Anse Cimitiere, with each beach concluding with a success rate of 100%. The least suitable beach was Anse Badamier, with a nesting success rate of 27%. It should be noted however that small sample numbers from Anse Papaie, Anse Laraie and Anse Cimitiere (n = 35, n = 9 and n = 9, respectively) are likely skewing the results for these three beaches.

	Nesting Season	2010 - 2011	2011 - 2012	2012 - 2013	2013 - 2014	2014 - 2015	2015 - 2016	2016 - 2017	2017 - 2018	2018 - 2019
Hawksbill	Activities	312	367	522	323	428	596	479	445	506
	Total Nests	151	186	282	128	225	368	182	148	234
	Population Estimate	38 - 50	47 - 62	71 - 94	32 - 43	56 - 75	92 - 123	46 - 61	37 - 49	59 - 78
Green	Activities	8	14	9	6	53	47	21	56	24
	Total Nests	0	8	2	4	22	27	10	21	18
	Population Estimate	1 - 2	1 - 2	1 - 2	1 - 2	5 - 7	5 - 9	2 - 3	4 - 7	4 - 6

Table 16. Total number of activities and nests for Hawksbill and Green turtles recorded for the past eight nesting seasons, 2010 – 2019, and population size estimates. Numbers in red indicate years where monitoring was not consistent.

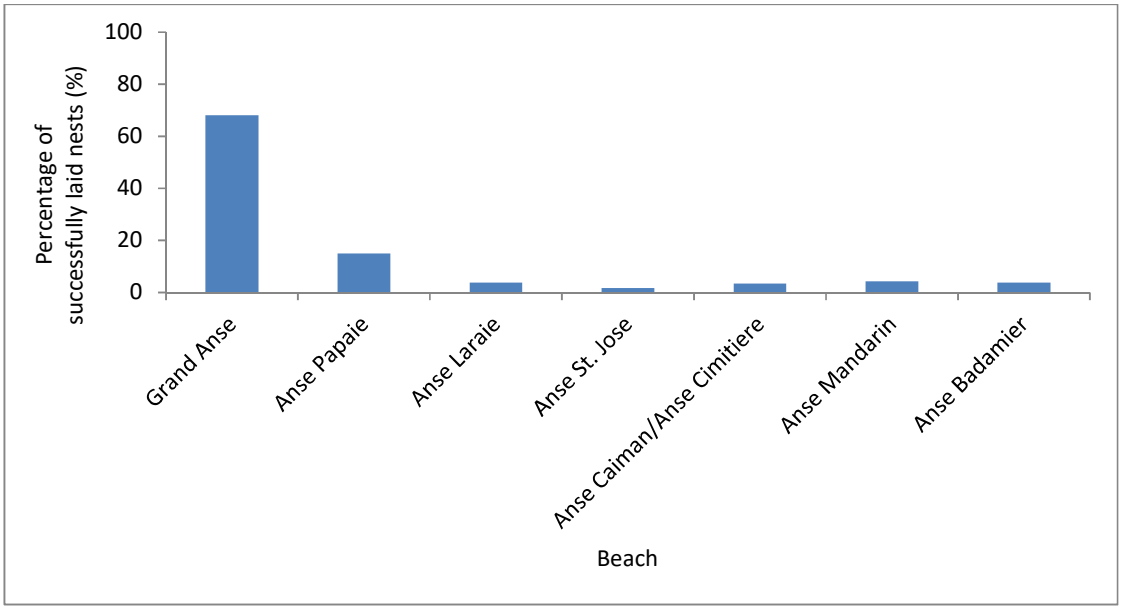

Figure 39. Percentage of total nests laid on each of the seven nesting beaches on Curieuse during the 2018 – 2019 Hawksbill season.

Nesting Adults – Green Turtles

For the 2018 – 2019 season, there were a total of 24 activities, of which 18 were nests. 11 of the nests were laid on Grand Anse whilst the rest were laid on Anse Papaie and Anse St. Jose. The peak of nesting season occurred in January with 41% of all activities recorded. The population estimate for Green turtles was four to six individuals (Table 16).

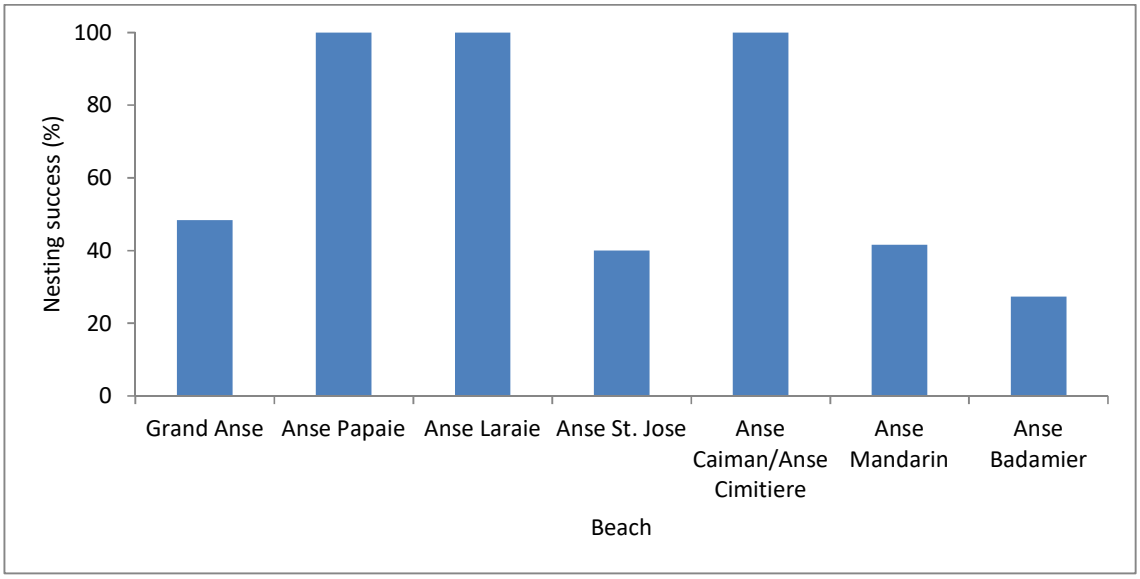

Figure 40. Beach suitability for the 2018 – 2019 Hawksbill season expressed as nesting success (number of nests divided by total number of activities) for each beach.

Hatching Success

A total of 84 Hawksbill nests and six Green nests were successfully excavated in the 2018 – 2019 season (Table 17). Hatching success was higher for Greens (98.7%) than for Hawksbills (90.1%). Grand Anse (93.2%, n = 84) had the highest Hawksbill hatching success, followed by Anse Papaie (91.7%, n = 12), where enough excavations were done to obtain a reliable result. Seven excavations were done on Anse Mandarin (87.5% hatching success), three on Anse Laraie (89.7%) and four on Anse Badamier (90.0%). The two beaches with the lowest hawksbill hatchling success were Anse St. Jose (27.7%, n = 3) and Anse Cimitiere (0.0%, n = 2). Anse Cimitiere suffered losses to both nests due to erosion. Anse St. Jose is the most visited beach on Curieuse by tourists so this will certainly have an effect on the nesting choice, and the reasons for low hatching success here were erosion as well as vegetation roots present and crab/worm activity. As mentioned previously, six Green turtle nests were found, however only four were successfully excavated. Two were omitted from analysis due to either no egg chamber being located, or high crab activity. A total of seven triangulated Green and Hawksbill nests were lost to erosion over their incubation period.

Nesting Hawksbill Identification

There were a total of 76 encounters during the 2018 – 2019 season. Of these, 24 turtles had tags fitted by GVI and were recorded as newly tagged turtles. 51 already tagged turtles were encountered. Four turtles had a tag missing from one flipper; new tags were applied. The majority of turtles were only encountered once, however six were encountered twice and another six were

encountered three times. One turtle was encountered five times, laying each time within a period of two months. No turtles encountered in the 2018 – 2019 season were encountered in the previous season.

	Hawksbill	Green
Number of Excavations	84	6
Hatching Success (%)	90.1	98.7
Emerging Success (%)	89.2	98.7
Average Clutch Size	157.0	89
Average Nest Depth (cm)	46.0	64.1
Total Hatched Eggs	11,562	356

Table 17. Hawksbill and Green turtle excavation parameters collected for the 2018 – 2019 season.

Discussion

The 2018 – 2019 nesting season experienced a slow start, however more Hawksbill activities and nests were recorded compared to the previous season, and the encounter rate was higher during the 2018 – 2019 season than the 2017 – 2018 season (n = 76 and n = 34, respectively). This can be expected due to the nature of biennial nesting populations of hawksbill turtles. Higher encounter rates enable us to look more closely at inter-nesting behaviour of Hawksbills.

Table 16 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 individuals are higher. Our understanding of Hawksbill population trends will be aided with the completion of the 2019 – 2020 season.

This is the seventh 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 are based on the assumption that Hawksbills lay an average of three to four 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 underestimate of the number of females nesting on Curieuse annually. The continuation of metal flipper tagging and recording of tag numbers will hopefully allow a greater understanding of the degree of inter-nesting intervals and changes in nesting site selection. This may also lead to a more accurate estimate of the nesting

female population, though this requires collaboration and sharing of data between islands. 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 2018 – 2019 season six Hawksbill turtles were encountered on three separate occasions, another six where encountered on two separate occasions and one individual was encountered five times with 14 – 16 days between each encounter. During the 2017 – 2018 season, the highest number of repeat encounters was three. With a higher proportion of turtles encountered more frequently, we may gain a better understanding of their inter-nesting behaviour. The 2018 – 2019 season had much more conclusive data than previous years to interpret inter-nesting behaviour. Turtles encountered three times during the 2017 – 2018 season had on average a 44 day inter-nesting periods followed by 14 days between nesting. Nevertheless, we must consider that these turtles may have nested, or attempted to nest, between these days without being encountered. The average number of days between nesting was 14, which supports the known inter-nesting behaviour of the species.

Green sea turtles lay nests throughout the year in the Inner Granitic Islands, though the low number of encountered activities 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, 2015 – 2016 and 2017 – 2018 seasons were remarkably higher than previously seen on Curieuse. The 2018 – 2019 season saw a slight decrease in turtle nesting activities with 18 nests being identified compared to the previous year of 21. 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 population 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 2014 – 2015 and 2015 – 2016 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 females. The 2018 – 2019 season documented 24 activities, of which 21 were nests producing a population estimate of four to six females. 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 90.1% for the 2018 – 2019 nesting season (n = 84) seems average when compared to previous data and other islands. Hatching success for Green turtle nests was higher (98.7%). However, the small sample size (n = 4) 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). This 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 proved 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 the recent discussions regarding increased development on Curieuse for tourism, it is imperative that Grand Anse and Anse Papaie remain safe, undisturbed areas for nesting Hawksbills and Greens in the inner islands. 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, with the help of SNPA rangers, will be attempting 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 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 subsequently 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 same 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 41). 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.

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).

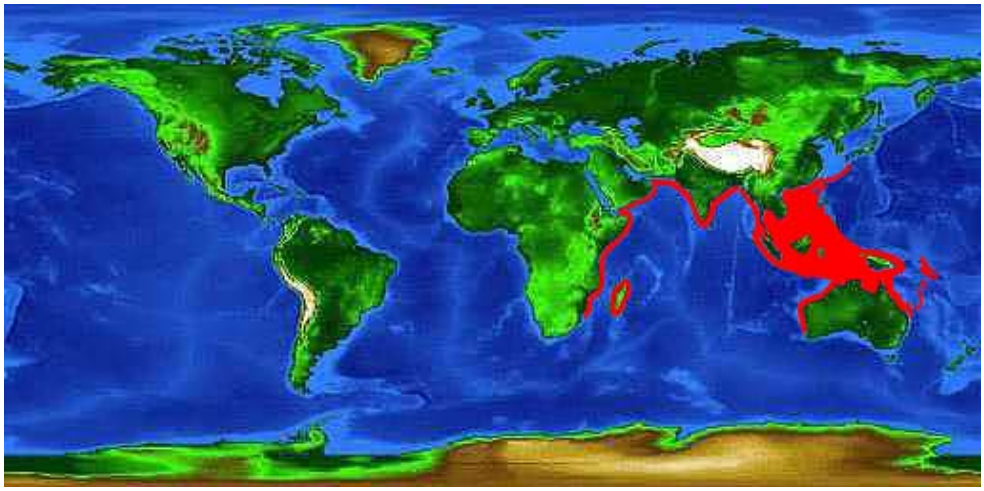

Figure 41. Distribution of the Sicklefin lemon shark, *N. acutidens* (Image from IUCN).

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 sixth research season. 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 detections. However in 2019 a passive acoustic tracking study was initiated and is ongoing (see following chapter). Throughout the six year study on Curieuse, Sicklefin lemon sharks were the sole shark species captured in the lagoon until last season, when for the first time the IUCN Near Threatened juvenile Blacktip reef shark, *Carcharhinus melanopterus,* was also caught in the lagoon. During the 2018 – 2019 season juveniles of two further species were captured for the first time, the Near Threatened Blacktip shark, *Carcharhinus limbatus*, and the Critically Endangered Scalloped hammerhead shark, *Sphyrna lewini*, were also captured using identical methodologies to the lemon shark study.

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 42). 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.

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 an average of 89cm deep during spring tides (Hodgkiss et al. 2015). 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 0.8m above chart datum, the southernmost section of the lagoon is isolated, forming a pool approximately 25 x 50m, which is referred to as "Pat's Pool".

Figure 42. *N. acutidens* study area within the Turtle Pond at Baie Laraie, CMNP, Seychelles.

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 – 90 x 0.75m and 10 x 1.5m (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 – 25 x 1.5m, 18 x 1.5m, and 10 x 1.5m, with a stretched mesh of 60mm (the 10m net was decommissioned in August 2016 and replaced with the 25m net, and the 18m net was decommissioned in November 2019 and replaced with a second 25 x 1.5m 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 (2.12 x 12mm *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, 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. A sibling analysis using genetic samples from 2014 to 2017 has been carried out and full results will be available in the next GVI Annual Report. 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 8.1 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): $CF = (\text{weight [kg]}/PCL^3) \times 10^5$). 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 basic results for the other species captured over the 2018 – 2019 research season.

2018 – 2019 Lemon Shark Study

Research Effort

The first *N. acutidens* neonate capture of the 2018 – 2019 cohort was on the 10th of October 2018. This marked the starting point for captures of the 2018 – 2019 research season. Research activities for this season comprised a total of 48 survey sessions over 370 days. Sampling effort varied throughout the season (Figure 43), with an average of four (± 2.5SD) surveys per month. The highest sampling effort by number of surveys was in October and November 2018 (n = 7, n = 9 surveys/month respectively). 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.

Capture Overview

A total of 169 captures were made over the 2018 – 2019 season, comprised of 132 new captures and 37 recaptures (of 29 individuals). No individuals initially captured during the 2017 – 2018 season were recaptured during the 2018 – 2019 season.

Compared to the 2017 – 2018 season, the total number of captures in the 2018 – 2019 season was 19.9% higher, with 15.8% more new captures and 32% more recaptures. The total number of captures varied among sampling sites (Figure 44), with the highest number of captures occurring in the North Turtle Pond and Pat's Pool (both n = 71), and lowest in the Mangroves (n = 4). Over the entire season, the mean capture rate was 3.5 (± 4.6SD) captures per survey, however capture rates varied by month (Figure 43). Initial neonate capture began in October, with a mean capture rate of 2.3 individuals per survey (± 1.98SD). The mean capture rate peaked in November, with a capture rate of 8.3 individuals per survey (± 8.0SD). Mean capture rate then declined in December (5.0 ± 2.83SD captures per survey) until the end of the season.

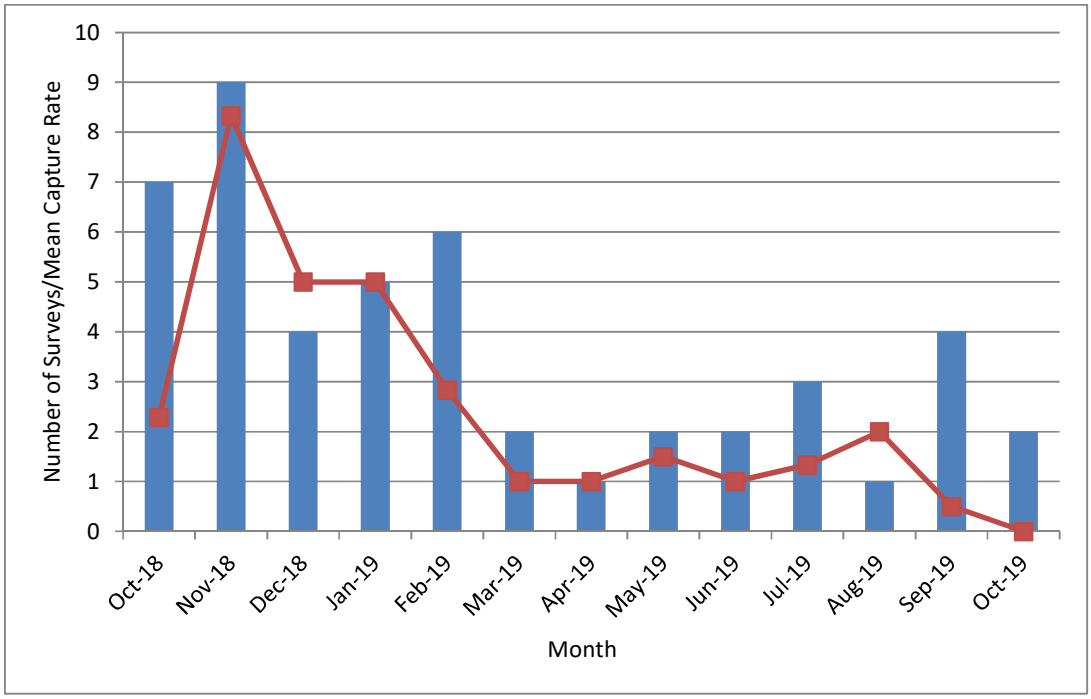

Figure 43. Total monthly sampling effort in number of surveys (blue bars) and mean *N. acutidens* capture rate per survey (red line) over the 2018 – 2019 season.

The most successful capture method involved the use of gill nets, which accounted for 97.6% of captures, followed by the seine net, which accounted for 2.4% of captures. The dip net and cast net methods accounted for 0% of captures. This reflects an evolution of sampling strategy with improved capture rates, however although the gill nets were responsible for the final capture of almost all individuals a combination of nets was frequently employed, with the seine net being extremely useful in containing sharks in an area for final capture with gill nets, and the dip nets were used in most captures to secure the individuals.

Population Estimate

The size of the neonate *N. acutidens* population for the 2018 – 2019 season was estimated at 337 (± 65.7SE, range: 242 – 508, 95%CI). This population estimate is 8.4% and 31.6% higher than 2014 – 2015 and 2015 – 2016 respectively, and 7.4% and 49.0% lower than 2016 – 2017 and 2017 – 2018 respectively (Figure 45).

Sex Ratio

A total of 59 females and 73 males were captured, resulting in a ratio of 109 males: 100 females. This compares to the 2014 – 2015 and 2017 – 2018 seasons which were also male biased (126 males:

100 females and 109 males: 100 females respectively). Both 2015 – 2016 and 2016 – 2017 cohorts were female-biased (83 males: 100 females and 98 males: 100 females respectively).

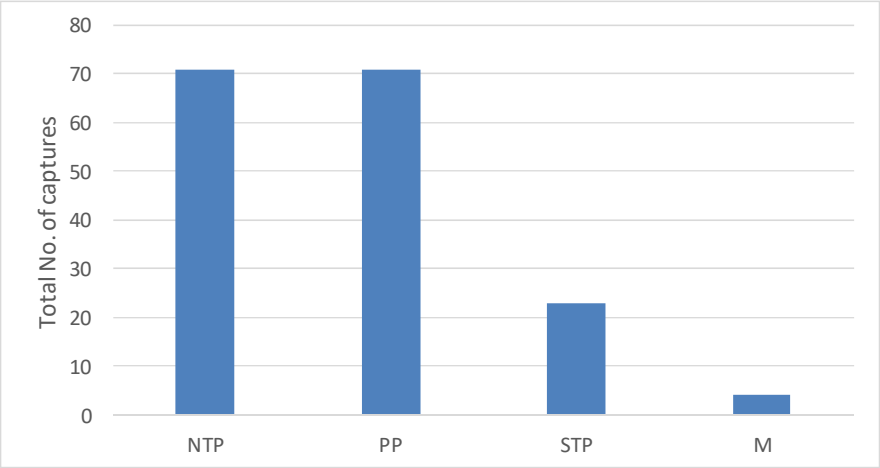

Figure 44. Total number of *N. acutidens* captured by sampling location, over the 2018 – 2019 season. M = Mangroves, NTP = North Turtle Pond, PP = Pat's Pool, STP = South Turtle Pond.

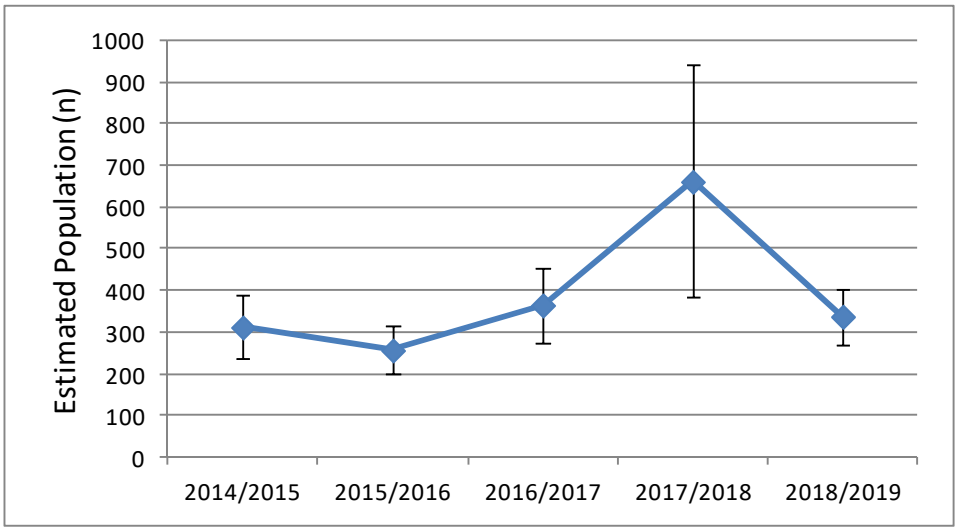

Figure 45. Population estimates (n ± SE) for *N. acutidens* in the study area over five seasons.

Pupping Season

The first observed neonate of the 2018 - 2019 cohort was captured on the 10th of October 2018, 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 almost fully open umbilical scar was captured on the 24th of December 2018. Therefore, the 2018 – 2019

pupping season was an estimated 75 days long, one day longer than the 2016 – 2017 season and the same as the 2015 – 2016 season.

Size at First Capture

Size at first capture data was used to produce the following summary statistics for the 2018 – 2019 season. With regards to size, mean PCL was 51.8cm (± 4.4SD), range: 44.3 – 75.1cm. Mean FL was 57.0cm (± 4.9SD), range: 44.9 – 82.0cm. Mean TL was 66.5cm (± 5.61SD), range: 56.3 – 96.6cm. 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 1.67kg (± 0.6SD), range: 0.85 – 5.8kg. 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 46), the length of captures appears to slightly increase between October 2018 and February 2019, with a relatively low variation among individuals. Large fluctuations in the average length of captures were evident from March onward, primarily due to low capture numbers (zero to three captures/month).

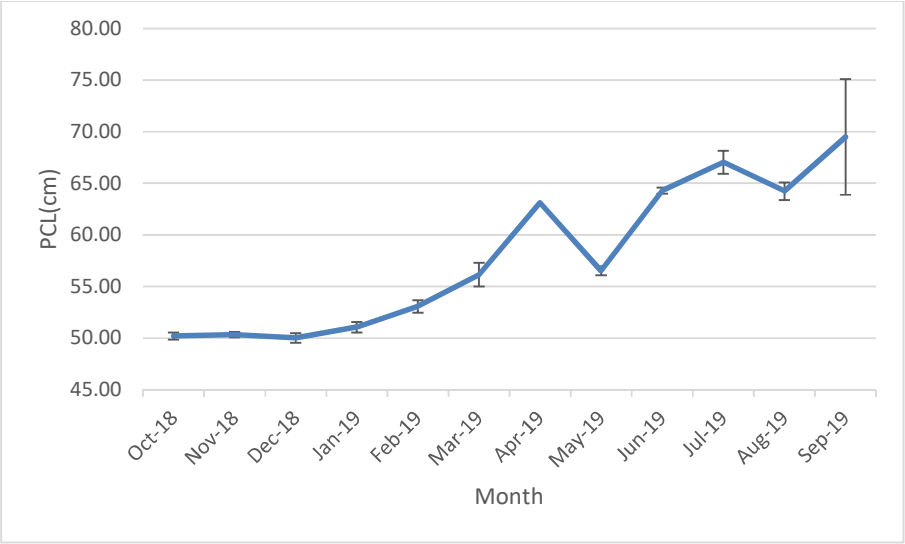

Figure 46. Mean PCL (± SEM) by month for *N. acutidens* captures over the 2018 – 2019 season.

The mean weight of captures by month (Figure 47) was steady from October 2018 to February 2019. From March 2019 onward there was a consistent increase in mean weight, with the exception of May and August 2019, although again low capture numbers at that time of year likely influence the results.

Condition Factor

The mean condition factor of the population over the 2018 – 2019 season was 1.20 (± 0.08SD), range: 1.09 – 1.3. Upon comparing mean values by month (Figure 48), condition factor appeared to decrease between October 2018 and February 2019. Large fluctuations in the condition factor of captures are evident from February onward, most likely due to low capture numbers (0 – 1.5 captures/month). No significant difference (α= 0.05; t-test) in condition factor was detected between the sexes or seasons.

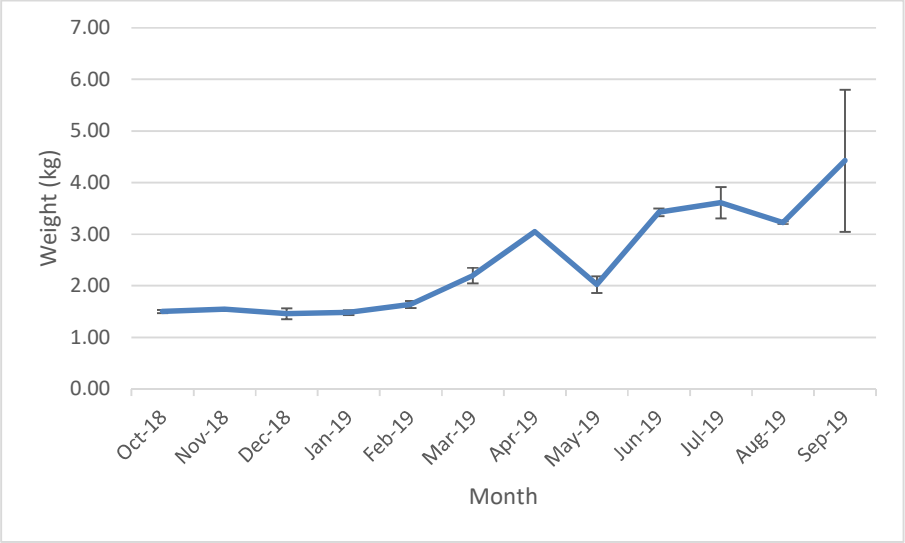

Figure 47. Mean weight (± SEM) by month for *N. acutidens* captures over the 2018 – 2019 season.

Recaptures and Growth

2018 – 2019 cohort

A total of 37 recaptures were made of 29 individuals, which accounted for 27.8% of total captures in the 2018 – 2019 season. Of these recaptures, no individuals were initially captured during previous seasons.

Individuals of the 2018 – 2019 cohort were recaptured between one and two times. Recaptured individuals were at large for an average of 79.6 days (± 71.8SD), range: 6 – 296. Mean growth in TL per day was calculated at 0.03cm/day (± 0.04SD), range: -0.01 – +0.2. Mean annual growth in TL was calculated at 15.64cm/year (± 15.7SD), range: -3.88 – +73.00. Mean growth in weight per day was calculated at -0.0001kg/day (± 0.0078SD), range: -0.03 – +0.013. Mean annual growth in weight was calculated at -0.042kg/year (± 2.8SD), range: -11.2 – +4.8.

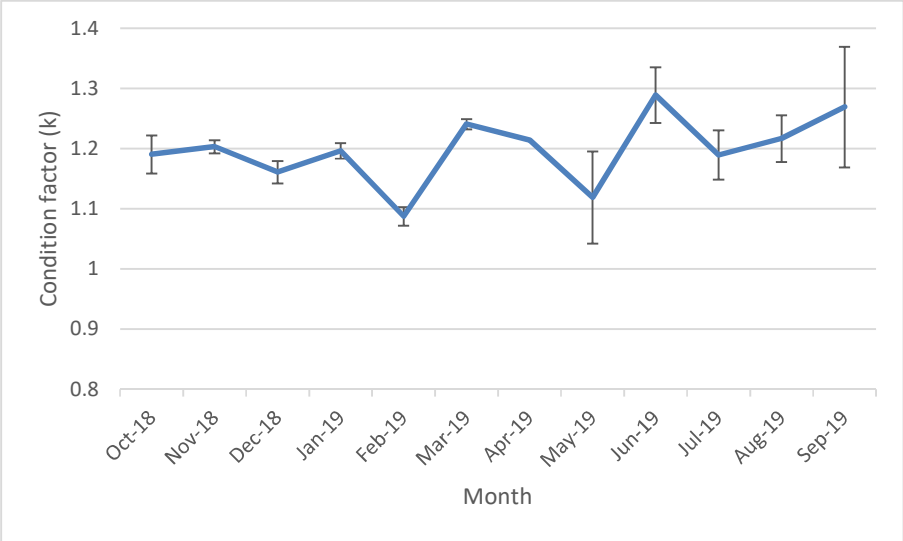

Figure 48. Mean condition factor (k) by month (± SEM) for *N. acutidens* captures over the 2018 – 2019 season.

Sampling Mortality and Natural Injuries

Sampling mortality for the 2018 – 2019 season was 2.96% (n = 5). Natural injuries were observed in 4.7% (n = 8) of captured sharks.

Natural Mortality

During the 2018 – 2019 season there were no recorded natural mortalities.

2019 – 2020 Mark-Recapture Study

The current season for the mark-recapture study began on the 15th of October 2019 with the capture of the first neonate of the pupping season, as evidenced by the presence of a partially open umbilical scar. By the end of the 2019, a total of 142 year-0 sharks had been captured, with a total of 56 recaptures. A preliminary calculation suggests a population of 303 individuals (± 49.2SE, range: 231 – 430, 95%CI).

Juvenile Blacktip Sharks

Eight juvenile Blacktip shark (*Carcharhinus limbatus*) captures were made between the 3rd of September and 7th of October 2019. All eight were PIT tagged and confirmed as new captures. The eight captures were made over five surveys, averaging 1.6 (± 0.89SD) individuals per survey. Gill nets accounted for 100% of captures; all eight were captured in the south Turtle Pond. A total of three

males and five females were captured, resulting in a sex ratio of 60 males: 100 females. Every individual captured had either a freshly closed or closed umbilical scar. The size at first capture showed a mean PCL of 62.9cm (± 7.6SD), range: 57.5 – 80.5cm, mean FL of 70.8cm (± 10.4SD), range: 63.5 – 96.0cm, and mean TL of 84.7cm (± 11.2SD), range: 78.4 – 112.0cm. 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 3.4kg (± 0.49SD), range: 2.6 – 4.1kg. 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 1.4 (± 0.56SD), range: 0.0 – 1.8. No significant difference (α = 0.05; t-test) in condition factor was detected between the sexes.

Juvenile Blacktip Reef Sharks

Two juvenile Blacktip reef shark (*Carcharhinus melanopterus*) captures were made on the 15th of October 2019. Both were PIT tagged and confirmed as new captures. Gill nets accounted for 100% of captures; both were captured in the north Turtle Pond. Both individuals were females. One individual had a ¼ open umbilical scar and the other closed, therefore no conclusions can be drawn regarding the pupping season from this year's data, except that pupping occurred in the month of October. For size at first capture both individuals showed a mean PCL of 42.9cm (± 0.0SD), mean FL was 48.2cm (± 0.14SD), range: 48.1 – 48.3cm, and mean TL of 58.7cm (± 0.49SD), range: 58.3 – 59.0cm. The mean weight at first capture over this season was 1.3kg (± 0.14SD), range: 1.2 – 1.4kg. The mean condition factor of the population over this season was 1.6 (± 0.18SD), range: 1.5 – 1.8.

Juvenile Scalloped Hammerhead Shark

One juvenile Scalloped hammerhead shark (*Sphyrna lewini*) was captured on the 8th of August 2019 and again on the 24th of September 2019. A PIT tag was inserted and it received the same workup procedures as for the Lemon sharks. On both occasions it was captured using gill nets in the south Turtle Pond. It was a male with a freshly closed umbilical scar on first capture, which was then closed on second capture after being at large for 49 days. The daily growth in TL was calculated to be 0.11cm/day, and annual growth in TL at 40.2cm/year. The daily growth in weight was calculated to be 0.002kg/day, and annual growth in weight at 3.72kg/year.

Discussion

Currently in its sixth 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 (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 since the 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 24th of December (2018), equalling a maximum estimated range of approximately 91 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 2018 – 2019 season has resulted in data collection over four 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. Similarly the mean weight of the population generally remained stable between September and January with a slight increase in January. The population exhibited positive growth throughout the remainder of season (bearing in mind sample size ranged from zero to two individuals per month). There was a significant difference between the mean weight from the 2017 – 2018 and 2018 – 2019 seasons, with 2018 – 2019 having a larger mean weight. This difference in weight between seasons could be due to an increase in resources in the Turtle Pond but also likely due to an increased number of individuals captured later in the 2018 – 2019 season compared to 2017 – 2018. During 2017 – 2018 there were eight pups captured from February onwards, and in 2018 – 2019 there were 33 captured during the same time period. Following several months of growth the pups caught later in the season tend to weigh more so an increased proportion captured later in the season would likely significantly increase the mean weight. The population exhibited positive growth throughout the remainder of season from February onward, however it should be borne in mind that sample sizes from this time of year are generally very low. The three individuals captured in May displayed a lighter weight and shorter PCL, as well as a lower condition factor than those caught during the months before and after. This increase in weight and length dissimilarity over the later potion 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 seemed to increase 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 (15.64cm/year; ± 15.7SD) is comparable to that of other studies of *N. acutidens* populations (12.5 – 15.5cm/year) (Stevens 1984) and Curieuse individuals from the 2017 – 2018 cohort (21.17cm/year; ± 18.0SD). 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; intermediate life stages have rarely been caught in this study, limiting the scope of the growth data primarily to year-0 sharks, however continuation of this study should steadily increase the volume of data for older individuals and allow a meaningful analysis.

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%, 0% and 2.96% over the 2015 – 2016, 2016 – 2017, 2017 – 2018 and 2018 – 2019 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.

The capture of juvenile Blacktip sharks emphasises the importance of Curieuse Island as a nursery to multiple juvenile shark species. Blacktip results showed no major differences from other studies conducted on *Carcharhinus limbatus*, such as Capapé et al. (2004). Curieuse Blacktip pupping season was within the scope of known Blacktip parturition in Africa. It is important to note the small number of Blacktips caught and the difficulties of determining significant results from such a low capture rate. Based on the lack of neonates with open umbilical scars, it is difficult to estimate if the pups come from one or multiple mothers. The continued capture of Blacktips in the Turtle Pond 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 inaugural capture of juvenile Blacktip reef sharks and a critically endangered Scalloped hammerhead shark further highlights the importance of CMNP as a significant nursery area for several shark species, and the need to continue monitoring in order to provide scientific data to enable its effective management and ensure its ecological functions are maintained.

Conclusion

The Sicklefin lemon shark monitoring program on Curieuse, now in its sixth 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.

The continued capture of multiple juvenile Blacktip reef sharks, and the inaugural capture of juvenile Blacktip and Scalloped hammerhead sharks in the Turtle Pond highlights the continued importance of CMNP in protecting multiple shark species in their vulnerable life stages. It is therefore recommended that an increased focus is placed on the protection of this critical shark nursery area, especially 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.

Sicklefin Lemon Shark Acoustic Tracking

Introduction

Since 2014 the mark-recapture study of the neonate *N. acutidens* population within CMNP has been collecting data on basic population parameters and is now in its sixth year of research. Data collected through the project has already provided significant results relating to the duration of the pupping season, neonate population size, growth and habitat use within the CMNP, and led to the publication of a research paper along with the presentation of a poster at the WIOMSA conference in 2015, and subsequently the publication of a peer reviewed research paper in 2017.

Despite the success of the PIT tagging study however, there are clear limitations to the types of data that can be collected and information that can be gleaned from such a study. In particular, it is not possible to conduct any analysis on the spatial movement patterns of the species, and the restricted survey area for the PIT tagging study (the Turtle Pond) raises questions regarding the accuracy of the population estimates. Furthermore, the overall range of the species on the east side of Curieuse is believed to extend from the Turtle Pond in the south to Point Rouge in the north, encompassing many different habitats, and there is currently no information regarding the use of these habitats on Curieuse by neonate *N. acutidens*. Previous attempts to study these aspects of the population using active acoustic tracking proved to be impractical for a variety of reasons, so in 2019 a passive acoustic tracking project was initiated. The project should greatly improve our understanding of the species and its use of the habitat within CMNP.

The overall rationale of the project approach is that by collecting data on the movement and habitat use of neonate *N. acutidens*, a species that is vulnerable to over-fishing, the effectiveness of CMNP in protecting the critical life stage can be assessed. Furthermore, in order to effectively conserve, manage, maintain or increase populations of sharks both inside and outside of protected areas, it is

essential to possess knowledge regarding the status of a population, involving an assessment of life history patterns and population trends over time, in particular critical early life stages. Therefore, any new information resulting from the scientific study of sharks can aid in their management and conservation.

The installation of an acoustic receiver array along the east coast of Curieuse Island, together with the acoustic tagging of 20 neonates, will enhance our knowledge of *N. acutidens* spatial ecology. Furthermore, results from this project will enable MPA practitioners to make better informed future recommendations with regards to the protection of this particular species, and allow future studies of *N. acutidens* and potentially other species.

The acoustic tracking project is currently in its data collection phase, therefore full results will be presented in the 2020 GVI Annual Report, however a summary of progress to date will be presented here.

Aims

The main aims of the acoustic tracking project are as follows:

1. Gain a greater understanding of the movements of neonate *N. acutidens* within CMNP and the efficacy of the park size in their protection.
2. Obtain an improved understanding of spatial behaviour and habitat use of 20 neonate *N. acutidens* within CMNP by monitoring their movements for six months using acoustic transmitters.
3. Refine existing mark-recapture population estimates of neonate *N. acutidens* within CMNP.
4. Contribute to national efforts to protect biodiversity by identifying areas of critical habitat within and outside CMNP essential to the survival of neonate *N. acutidens.*
5. Provide critical habitat data to national park managers and stakeholders within CMNP and other protected areas to better inform management decisions regarding the conservation of *N. acutidens*.

Methodology

The core of the tracking project is an array of 12 *Vemco VR2W* acoustic receivers (Figure 49) located on the east coast of CMNP installed at specific locations underwater. The receivers automatically detect ultrasonic coded pings from *Vemco V13* transmitters (Figure 50), and decode the ping pattern

to an individual transmitter according to a code map before storing the detection in internal memory. The data can then be collected later by retrieving each receiver and downloading to a laptop via bluetooth. The original installation locations are shown in Figure 51. The array consisted of six receivers in shallow locations in the known habitat of neonate *N. acutidens*, and six positioned in deeper water from north of Point Rouge to east of Anse Mandarin. The array was designed to capture detailed movement patterns of the species in their usual range with the shallow receivers (R3, R5, R7, R8, R9 and R10), and attempt to detect any movement towards the boundary of the national park to the east with the deep receivers (R1, R2, R4, R6, R11 and R12). Shallow receivers were installed close to the substrate by means of cable ties attached to a length of steel rebar driven into the substrate, and deep receivers were attached with cable ties to a rope secured to a mooring of concrete blocks, and suspended in the water column with a foam float above.

Figure 49. Vemco VR2W acoustic receiver.

In order to determine the effectiveness of the receiver array it was first necessary to conduct range testing. The generally accepted minimum detection percentage for a reasonable analysis of this type is 80% of emitted ping sets. The receivers selected for the range testing were R3, R4, R5, R6, R9 and R10, this was intended to cover as many conceivable conditions as possible i.e. deep water (R4, R6), shallow exposed locations (R3, R5), and the sheltered shallow water of the Turtle Pond (R9, R10). Transmitters were installed at precise locations at known distances from each receiver (Figure 52), fixed close to the substrate by similar means as the shallow receivers.

Figure 50. Vemco V13 acoustic transmitter.

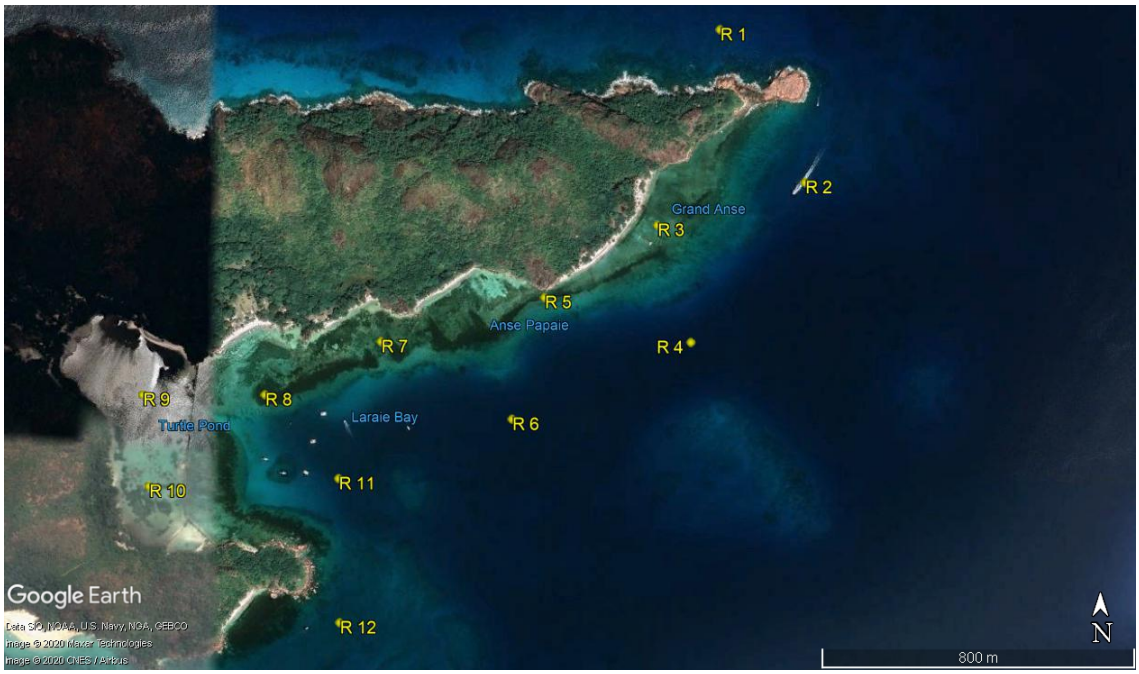

Figure 51. Original planned positions of acoustic receivers.

Initial range testing was conducted between the 20th of August and the 1st of October 2019, and further testing was continued until the 26th of November 2019. Receivers were retrieved periodically to download data and analyse the detection ranges, and adjustments made to the positions of receivers to ensure adequate detection percentages before redeployment and collection of further data to assess the modified receiver locations.

Once the receiver positions were judged to be appropriate for a minimum 80% detection range, all 12 receivers were deployed in their final locations on the 10th and 11th of October 2019 in preparation for the arrival of the 2019 cohort of neonate *N. acutidens*. Following their arrival, during sampling sessions for the PIT tagging study and using identical capture and workup methods, 20 individuals were selected for surgical implantation of transmitters, based on their apparent health and stress levels. Following workup individuals were transferred to a separate water filled trough and inverted to induce tonic immobility, prior to making a small incision with a scalpel in the ventral side in the posterior third of the body cavity, and transmitters inserted. Incisions were then closed with three sutures before being released and monitored to ensure recovery. At all times stress levels were monitored, and water in the trough frequently refreshed to maintain oxygen levels. The entire process for each shark was completed in five minutes or less.

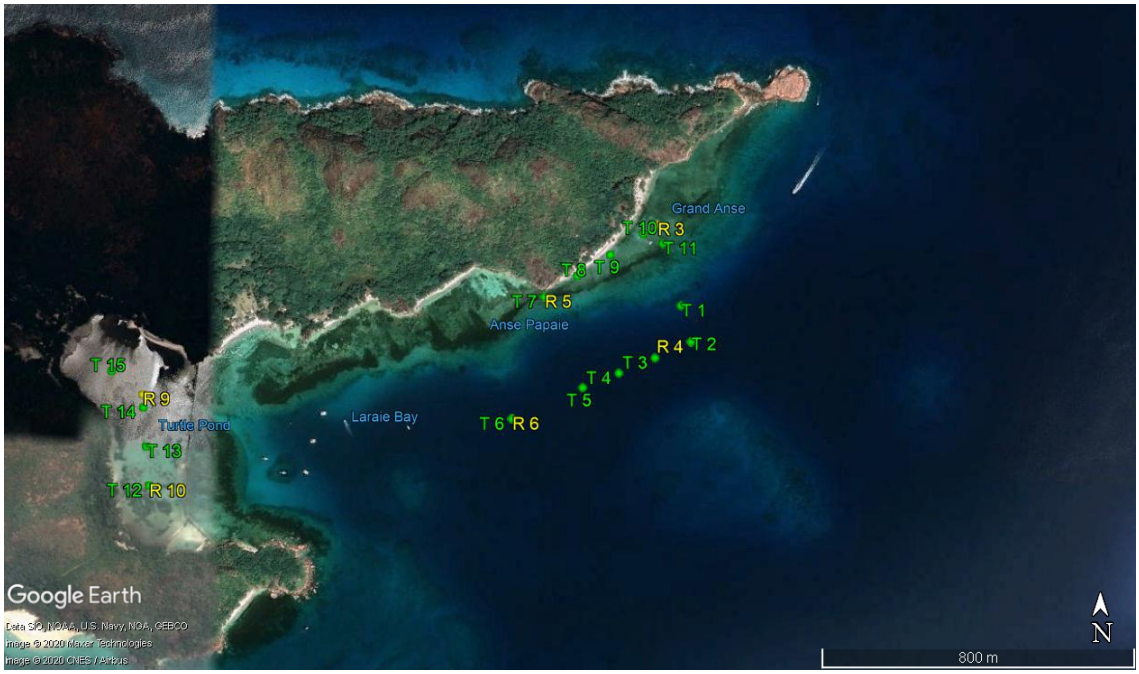

Figure 52. Positions of receivers and transmitters during range testing.

Results

Range Testing

Analysis of the initial download of range testing data showed mixed results. The deep receivers showed the best detection ranges with both R4 and R6 having approximately an 80% detection rate at 200m range (Figure 53). The shallow receivers in the Turtle Pond also showed reasonable detection rates of over 80% at 100m range for R10, and 60% at 100m for R9 (Figure 54). R9 was positioned in the very shallow north Turtle Pond, and at low tide some of the range testing transmitters would not have been detectable. Filtering the detections to remove periods of low tide would increase the detection percentage to at least 80%. This would be entirely appropriate, since Lemon sharks are not commonly seen in that part of the Turtle Pond at such low tides, and detection rates would more closely match the reality of shark distribution.

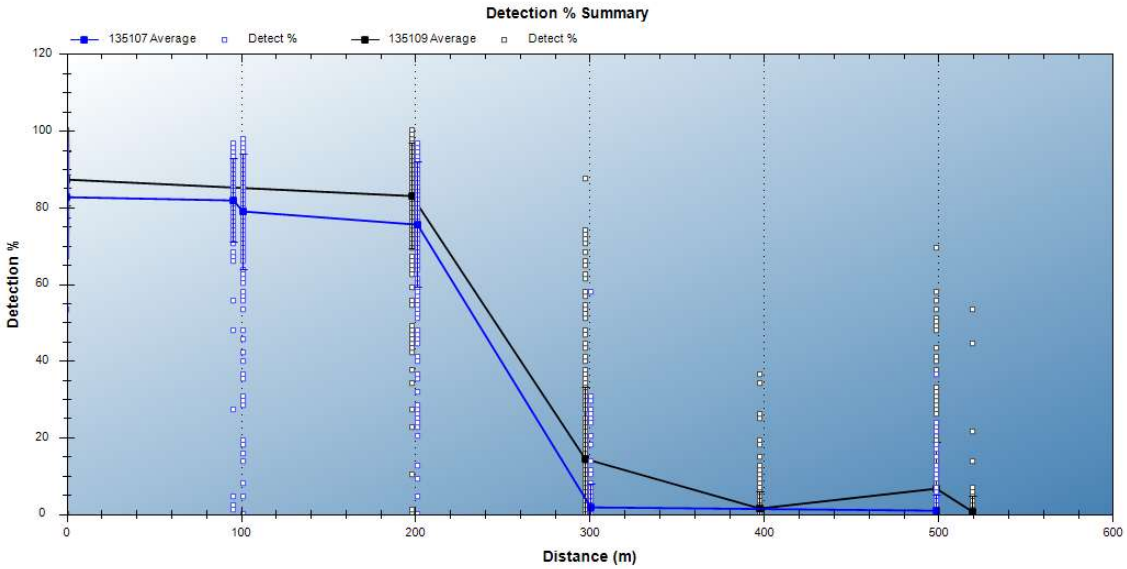

Figure 53. Detection rates for deep receivers R4 and R6 during range testing.

The shallow receivers positioned around Grand Anse in the north did not produce acceptable results, with the detection rate for R3 dropping below 80% by 40m range, and R5 recording very few meaningful detections (Figure 55). This is most likely due to interference from significant wave action, especially for R5, being positioned on the highly exposed point between Grand Anse and Anse Papaie.

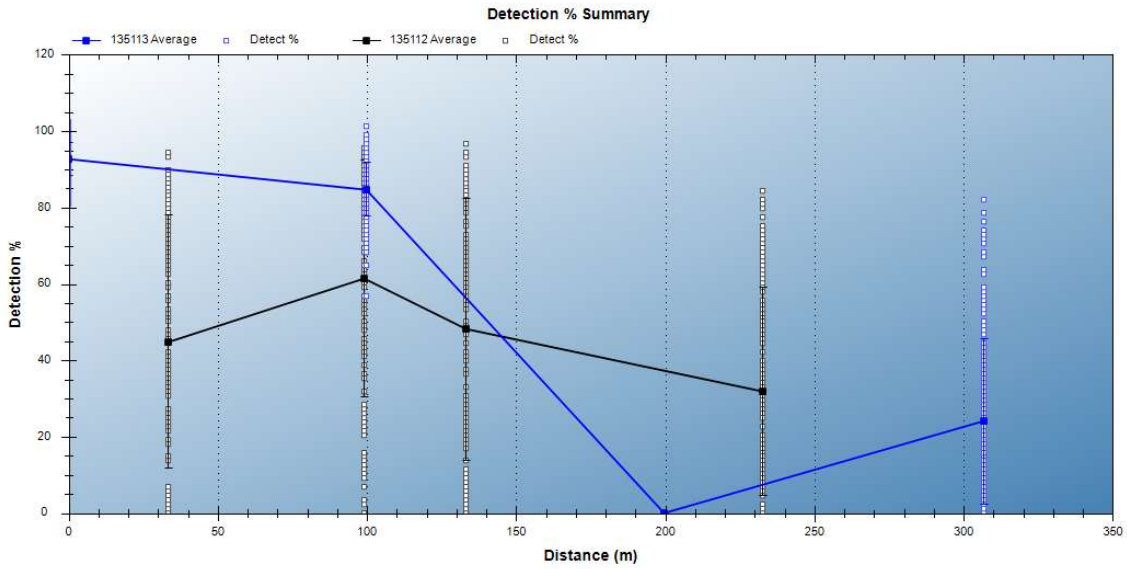

Figure 54. Detection rates for Turtle Pond receivers R9 and R10 during range testing.

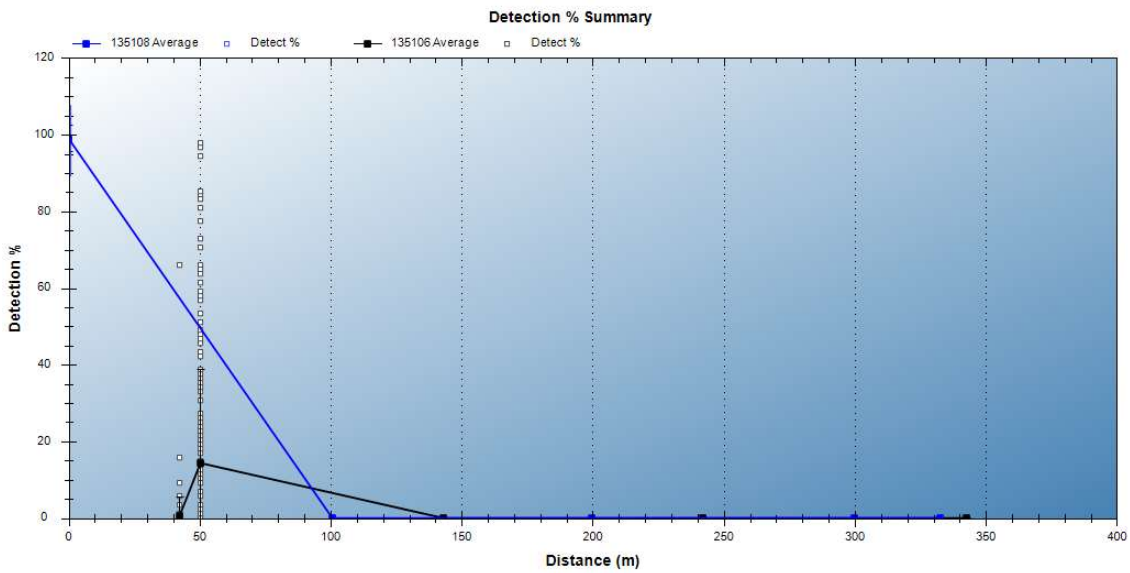

Figure 55. Detection rates for shallow receivers R3 and R5 during range testing.

Following analysis of the range testing data, the proposed deployment plan was modified to take account of the observed detection ranges. Figure 56 displays the final receiver array positions, along with 200m range radii for the deep receivers. Adjustments were made to the positions of the deep receivers from a spacing of 500m apart to 380m, with R12 remaining in its original planned location and the others being moved further south. This provided an overlap of detection ranges, and is assumed to effectively create an acoustic barrier across which if any individuals cross they will be

detected by one or more deep receiver. Receivers were deployed in depths ranging from 10.9m to 27.2m.

The positions of R9 and R10 in the Turtle Pond were deemed to be acceptable, therefore they remained in their original planned locations. It was assumed that the detection ranges of R7 and R8 would be greater than the original results for R3 and R5, since although shallow, they are positioned on more sheltered seagrass beds rather than the more exposed reef environment in front of Anse Papaie and Grand Anse, therefore interference would be expected to be lower, so their final positions remained essentially unchanged. The positions of R3 and R5 however were adjusted in an attempt to maximise detections. R5 was moved from the highly exposed point between Grand Anse and Anse Papaie to the centre of the more sheltered back reef in front of Anse Papaie. R3 was moved south west from its original location to a new location with a 70m distance between the reef fringe and the shore, based on an 80% detection range of 40m. This is expected to capture sufficient detections from any individuals traversing between Anse Papaie and Grand Anse.

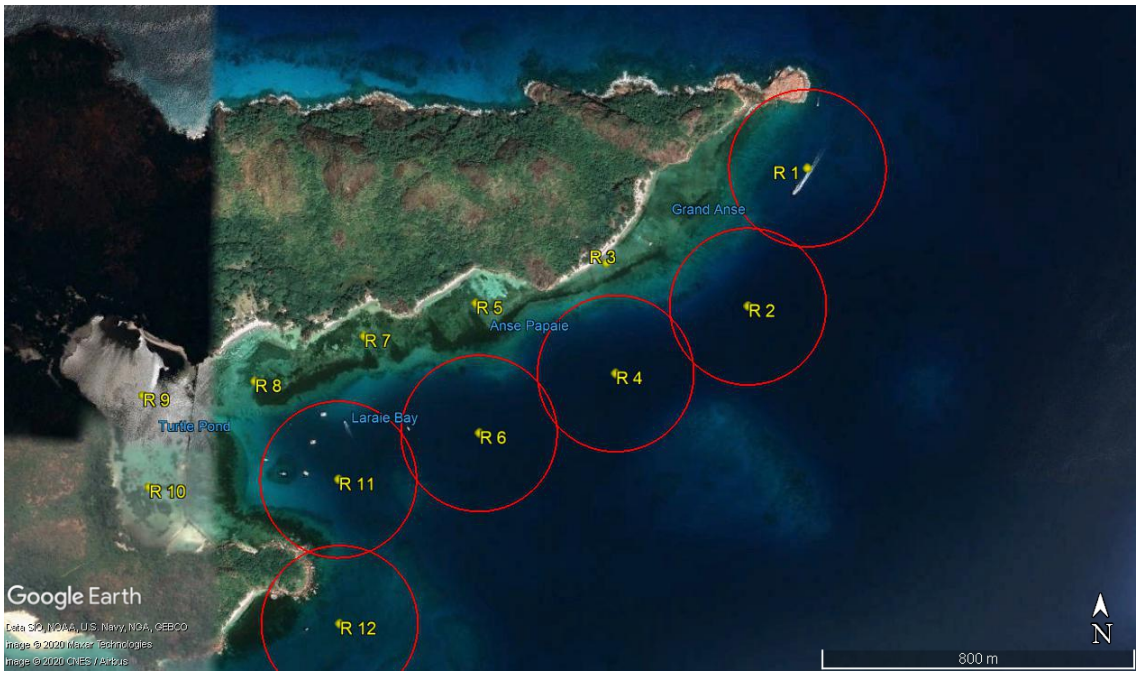

Figure 56. Final receiver deployment locations showing 200m detection radii for deep receivers.

Transmitter Implantation

V13 transmitters were surgically implanted in 20 neonate *N. acutidens* between the 24th of October and 11th of November 2019. 10 males and 10 females were selected, with a mean TL of 65.7cm,

range 61.4 – 72.1cm, and a mean weight of 1.61kg, range 1.325 – 2.05kg. Individuals were followed for as long as possible to ensure they recovered sufficiently from the workup procedure followed by implantation surgery. All individuals recovered and showed no immediate signs of negative effects.

Seven of the 20 individuals were recaptured at intervals ranging from eight to 35 days at large. All incisions appeared to be healing well, and all individuals appeared generally healthy. A further six individuals were also detected on at least two separate receivers, following the download of further range testing data. This confirms the survival of at least 13 of the 20 sharks implanted with transmitters.

Periodic monitoring of pings using a *Sonotronics Mantrak* manual tracking kit produced regular detections of individuals, often multiple individuals in an area at the same time. Changing patterns of detection with no static and unchanging detections provide further strong evidence that survival of individuals with implanted transmitters was high, and combined with recapture and receiver data, suggests that most if not all individuals having undergone surgery survived with no ill effects.

Discussion

This report presents the acoustic tracking project up to the point of transmitter implantation, and as such, only range testing data has been presented and will be discussed. The receiver tracking data will be analysed and presented fully in the 2020 GVI Annual Report.

Following analysis of the range testing data and modification of the planned acoustic receiver positions, it is believed that the project will achieve its aims. The detection ranges of the deep receivers will be sufficient for full coverage of the deeper offshore areas. This will allow us to answer the question of whether or not neonate sharks are likely to travel outside of CMNP, therefore confirming whether or not the boundaries of the national park are sufficient to fully protect this life stage of the species. Any individuals within the Turtle Pond should be detected the majority of the time they are in the area. The receivers deployed in front on Anse Laraie should provide an acceptable detection rate based on the similarity of the habitat to the Turtle Pond. The receivers in front of Anse Papaie and Grand Anse may have lower detection ranges, however strategic final positioning should yield sufficient data to allow a full analysis of the sharks' spatial movement patterns in that area. It is expected that there will be a gap in coverage towards the north of Grand Anse, however this is not expected to significantly affect the overall data set, and we expect to be able to produce a relatively detailed overall spatial analysis.

The apparent high overall survival rate of neonates having undergone surgery suggests that there will be a wide range of data on individuals available for analysis. Further recaptures of tagged individuals over a longer time period and comparison of their growth rates with the average growth of the rest of the neonate cohort should also provide an indication if there was any other significant effect of transmitter implantation. For the first time it should also be possible to provide some indication of neonate natural mortality rates, since it should be clear from receiver data when an individual is no longer moving around the study area, or in the case of predation, a sudden change in movement patterns may be detected.

The data from all receivers is due to be downloaded during May 2020, following which an extensive analysis will be conducted.

Conclusion

Passive tracking of neonate *N. acutidens* within CMNP will significantly expand the range of knowledge of the species, and identify the critical habitat for their early life stages. It will also provide detailed data on their spatial movement patterns, and determine whether the boundaries of CMNP are sufficient to fully protect the early life stages of the species. Implementation of the project has proceeded as planned, with acceptable positioning of receivers, and high apparent survival of neonates implanted with transmitters. Retrieval of receiver data is planned for May 2020.

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