Preventing Costs of Invasive Alien Species (IAS) in Barbados and Countries of the OECS Project

Assessment and Management of Lionfish and Status of Other Marine Invasive Species of Threat to High Biodiversity-value Reef Ecosystems

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Final Report:

February 2023

Citation:

Vallès, H., J. Walcott and H.A. Oxenford. 2023. Assessment and Management of Lionfish and Status of Other Marine Invasive Species of Threat to High Biodiversity-value Reef Ecosystems. Draft Final Report. Preventing Costs of Invasive Alien Species (IAS) in Barbados and Countries of the OECS Project. CERMES, UWI, Cave Hill, Barbados, 53pp.

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ACRONYMS

CABI	Centre for Agriculture and Biosciences International
CBD	Convention on Biological Diversity
CERMES	Centre for Resource Management and Environmental Studies
CZMU	Coastal Zone Management Unit
FD	Fisheries Division
FL	Fork Length
GEF	Global Environment Facility
IAS	Invasive Alien Species
MANOVA	Multivariate Analysis Of Variance
NHD	Natural Heritage Department
OECS	Organisation of Eastern Caribbean States
SIDS	Small Island Developing States
UWI	University of the West Indies (Cave Hill Campus)

ACKNOWLEDGEMENTS

This project was funded by the GEF Trust Fund, through the Ministry of Environment and National Beautification, Government of Barbados, as a component of the regional GEF Project # 9408 'Preventing Costs of Invasive Alien Species (IAS) in Barbados and Countries of the OECS'. We are extremely grateful to the team of competent SCUBA divers who undertook all of the fish surveys over a period of one year. Apart from the authors this team also comprised: Caroline Bissada, Holly Trew, Annabel Cox, Robert Bourne, Micaela Small and Joseph Weekes. We thank Barbados Blue and Randall Armstrong for their patience, expert boatmanship and use of their dive boats and crew for day and night surveys. We also appreciate the willingness and patience of trap fishers and spear fishers from Pile Bay and Oistins in allowing their catches and fishing effort to be recorded, and data collectors Caroline Bissada and Julian Walcott. We are extremely grateful to fishers Sherwin Batson and Andrew Babb of Pile Bay for their willingness to participate in the deep trap fishing trials. Special thanks are due to Robert Bourne who undertook all of the recreational diver and dive shop operator interviews, and was responsible for the sun coral surveys and removal experiments which he undertook for his supervised research project as a UWI student of Biology. We further appreciate the oversight and review of the Biodiversity Working Group.

EXECUTIVE SUMMARY

The red lionfish, *Pterois volitans*, an invasive alien marine species with potential to cause significant damage to valuable high-biodiversity coral reef ecosystems, arrived in Barbados at the end of 2011. In anticipation of its eventual arrival in what was a Caribbean-wide invasion, the Biodiversity Working Group of the Barbados Government drew up a Lionfish Response Plan and undertook pre-invasion baseline surveys of the coral reef fish community and of the reef fishery. A decade later, the current study, funded through the Ministry of Environment and National Beautification as a part of their GEF-funded project 'Preventing Costs of Invasive Alien Species in Barbados and Countries of the OECS' seeks to examine the impacts of lionfish on reef fish communities and catches of reef fishers in Barbados. This study also examines the status of two other marine invasive alien species in Barbados that are known to be present in many locations across the Caribbean, the sun corals, *Tubastraea* spp., and the seagrass, *Halophila stipulacea*, and are considered potential hazards to native marine biodiversity. The ultimate aim of this study is to provide appropriate management advice to the Government with regard to these three marine invasive alien species.

The impacts of lionfish, the focal species of this study, were examined using several approaches. These included re-surveying the coral reef fish communities at 10 reefs every four months for a year, and re-surveying the catches and catch rates of reef fishers at the two main landing sites for this fishery during the pelagic fishery season and during the off-season. The results were then compared with the pre-invasion baseline data collected in 2012 using the same methods and study sites to look for any changes in the native reef fish communities, or in the catch composition and yields of reef fishers. Additional roving surveys to record lionfish were also conducted on selected reefs during the day and again after sunset to make sure that lionfish was not being underestimated in daytime surveys. The study also collaborated with local fishers to conduct pilot fishing with deepwater traps to test their efficacy in controlling lionfish populations in depths beyond the reach of divers. Lionfish management efforts implemented to date were ascertained through the authors' personal knowledge and through interviews with Government technical officers, dive shop operators, and local recreational divers.

A comparison of coral reef fish diversity and abundance pre- and post-invasion indicated no detectable effects of lionfish on native fish assemblages. A comparison of day and night surveys indicated little difference in the lionfish abundance estimates at most sites, with the exception of Holetown bank reef where more lionfish were detected at night. This is likely due to lionfish reducing their activity during the daylight hours because of the particularly heavy spearfishing pressure known to occur at this site.

We found no evidence of differences in catch rates of reef fishers between the pre- and post-invasion periods suggesting no significant impacts on their earnings. We also found no evidence that the species composition of the catch in traps differed between the pre- and post-invasion periods, even though lionfish were occasionally recorded in the catch. In contrast, we found very strong differences in the catch composition of spear fishers pre- and post-invasion. Lionfish (now sought after by consumers) have become the second most important fish group taken by spear fishers. This suggests that the targeting of lionfish has helped relieve some of the spearfishing pressure on other reef fish groups, including parrotfishes, thus potentially benefitting the reef community.

The different lines of evidence presented support a particularly important role of spearfishing by

both commercial and recreational divers in helping to control the lionfish population at shallow to moderately deep sites (<30 m deep) to levels where they have no apparent impact on local reef fish communities or reef fisher yields/livelihoods. Sites where culling occurs regularly by recreational divers seem to be replenished by lionfish several weeks after culling, likely from lionfish immigration from deeper sites. Fishing trials with regular and modified deepwater traps were found to be ineffective at catching lionfish and thus of no value as a management tool in controlling the deepwater populations.

In summary, spearfishing, by both commercial and recreational divers, plays an important role in helping to control the lionfish population at shallow to moderately deep sites and as such there should be a continued effort to promote a spear fishery that targets lionfish. This should be bolstered through organized lionfish derby events which involve the Tourism Sector to help promote such events as opportunities to diversify the tourism product. The development of markets for lionfish meat (which can also be sold in restaurants) and for lionfish byproducts such as the fins (which can be used to make jewelry) should also be encouraged to help supplement local incomes.

An initial investigation into the sun corals found in Barbados confirmed that they are likely to be the alien species, *Tubastraea coccinea,* although we did not rule out the possibility of a second species *T. tagusensis* being present. A baseline assessment of their distribution revealed that it can now be found on most of the submerged marine artificial structures along the south and west coast of the island. Metal structures (large mooring buoys and shipwrecks) were found to be more likely to host sun corals than other non-metallic artificial structures (piers, reef balls and breakwaters). With regard to management action, we determined that complete removal of sun corals by scraping is an effective strategy to maintain the population at very low levels, at least over three-month periods. However, at sites where sun corals are yet to become dominant, such removal will likely also negatively affect co-occurring native benthic organisms. In summary, although at present sun corals have only colonized artificial structures, they can be found in high abundance on these and have the potential to spread into natural coral reef habitats. As such they should be considered a species of concern and be subject to further monitoring. Because sun corals are attractive and relatively easy to identify (particularly when their tentacles are extended), there is also an opportunity to involve recreational and professional divers in the monitoring to improve early warning detection of a potential expansion into natural reefs. Public awareness and research aimed at assessing their current impact on benthic communities inhabiting artificial structures should also be promoted.

Surveys of all known seagrass areas around Barbados found no evidence of the invasive *Halophila stipulacea*. This seagrass is believed to be spread by vessel anchors uprooting and picking up the plants as they are hauled, and transporting them to the next anchorage where they are deposited back in the sediment as the anchor is deployed. Although Carlisle Bay, which hosts native seagrasses, receives a fairly high number of visiting yachts, the vast majority use Barbados as a first stop after an Atlantic crossing and do not revisit after cruising other Caribbean islands where this invasive seagrass species has become established. As such no management action is currently warranted. However, seagrass areas, particularly in Carlisle Bay, should be surveyed periodically and the spread and potential impacts which may be largely positive given the paucity of seagrasses in Barbados should be monitored in the event that this species arrives here.

Lessons learned include:

(1) The ‘doom-and-gloom’ that was predicted to accompany the arrival of lionfish in Barbados waters has not materialized, primarily due to the current initiatives in place. Both trap and spear fishers have not experienced any significant declines in earnings, but have been able to supplement incomes through the sale of lionfish.
(2) The existing design of traps need not be modified, as numbers of lionfish caught seem to be more influenced by numbers at a site than the design of the trap.
(3) Non-native sun corals are widely distributed in Barbados on a range of artificial structures, but do not appear to have spread to natural reef habitat.
(4) Removal of sun coral colonies by scraping appears to be an effective way of controlling their abundance at least over the short term.
(5) The seagrass, *Halophila stipulacea* is not present on the island.

1 INTRODUCTION

1.1 Project background and objectives

As Party to the Convention on Biological Diversity (CBD), Barbados is required to put in place measures to achieve the objectives outlined in Article 8 of the Convention[1], which states that "Each contracting Party shall, as far as possible and as appropriate, prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species".

To fulfil these obligations, the Government of Barbados through the Ministry of Environment and National Beautification has endorsed the Global Environment Facility (GEF) funded project entitled '**Preventing Costs of Invasive Alien Species (IAS) in Barbados and Countries of the OECS**' with the Centre for Agriculture and Biosciences International (CABI) as the regional executing partner. Under this project, Barbados aims to:

- strengthen existing IAS management frameworks and improve cross sectoral arrangements to reduce IAS threats in terrestrial, marine and coastal ecosystems,
- eradicate or improve the control or management of IAS impacting species of national significance, reducing threats to these key species; and
- strengthen regional biosecurity through providing mechanisms for collaboration at the regional level and for building capacity to allow individual countries to collaborate at the regional level in preventing the introduction of IAS.

These objectives are framed within the existing socio-economic environment of Small Island Developing States (SIDS) of the Eastern Caribbean region where national policies, public awareness of IAS issues and the capacity to effectively address the threats of IAS are below desired levels. This is further exacerbated by the sub-regional characteristic of high trade and movement of people for work and tourism, which makes the issue of IAS prevention and management of paramount importance.

This project aims to address the issue of IAS at the national level through the generation of knowledge pertaining to IAS and their pathways, development of policy and regulatory frameworks, public awareness, capacity building, IAS prevention and management. Through this project, Barbados will also continue to fulfil its commitments under the CBD as it relates to Aichi Biodiversity Target 9[2], which focuses on the prevention, control, and management of Invasive Alien Species.

The '**Assessment and Management of Lionfish and Status of Other Marine Invasive Species of Threat to High Biodiversity-value Reef Ecosystems**' pilot project was undertaken by the Centre for Resource Management and Environmental Studies (CERMES) and the Department of Biological and Chemical Sciences of the University of the West Indies (UWI) in collaboration with the Working Group on Biodiversity (as the Project Steering Committee) and key stakeholders. The project fieldwork was conducted over an 18-month period, with the assessment focusing on the Indo-Pacific lionfish (*Pterois volitans*), as the primary target, and the invasive sun corals

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[1] https://www.cbd.int/doc/legal/cbd-en.pdf

[2] https://www.cbd.int/doc/strategic-plan/targets/T9-quick-guide-en.pdf

(*Tubastraea spp*) and invasive seagrass (*Halophila stipulacea*) as alien species of imminent threat. The assessment was focused on the west and south coasts of Barbados, and had the following objectives as the primary focus:

O1. Re-assess the status of native reef fish (density, size, species composition) and lionfish (density, size) in areas of differing management efforts to control the lionfish.

O2. Re-assess the status of reef fishers' landings (catch composition, catch rates) in areas of differing management efforts to control the lionfish.

O3. Determine the likely impacts of the invasion and the effectiveness of a range of management tools on the native reef fish populations.

O4. Determine the negative and any positive impacts of lionfish on reef fishers.

O5. Test the efficacy of using additional management tools such as specially designed deep water lionfish traps.

O6. Adapt the current national lionfish management strategy to ensure the most effective methods of control are being used to prevent future lionfish population expansion.

The secondary focus had the following objectives:

O7. Collect baseline data on the extent of occurrence of the invasive sun corals and the invasive seagrass species in Barbados.

O8. Pilot the effectiveness of removal treatments on sun corals.

This report presents the completed work of the 'Assessment and Management of Lionfish and Status of Other Marine Invasive Species of Threat to High Biodiversity-value Reef Ecosystems' pilot project.

2 INDO-PACIFIC LIONFISH (*PTEROIS VOLITANS*)

2.1 Background

2.1.1 The lionfish invasion

The lionfish invasion throughout the Wider Caribbean Region has been unprecedented, not only because it was the first case of non-native marine fishes invading and become established in the region, but because of the remarkable speed with which they have been able to spread[3], despite significant natural population connectivity barriers among sub-regional coral reef populations (e.g. Cowen et al. 2006). Reported to have originated from aquarium releases in Florida in the late 1980s and early 1990s (Freshwater et al. 2009), the non-native population comprising two species of lionfish spread up the USA east coast, across to Bermuda and ultimately to the Bahamas by 2004 (Figure 1). Subsequently, one species (*Pterois volitans*) crossed into the Caribbean basin and spread rapidly, invading islands in the northern Caribbean, through Central American reefs and along the Caribbean coast of South America reaching the eastern Caribbean islands from 2010,

[3] Animated map of lionfish spread (https://nas.er.usgs.gov/queries/SpeciesAnimatedMap.aspx?SpeciesID=963)

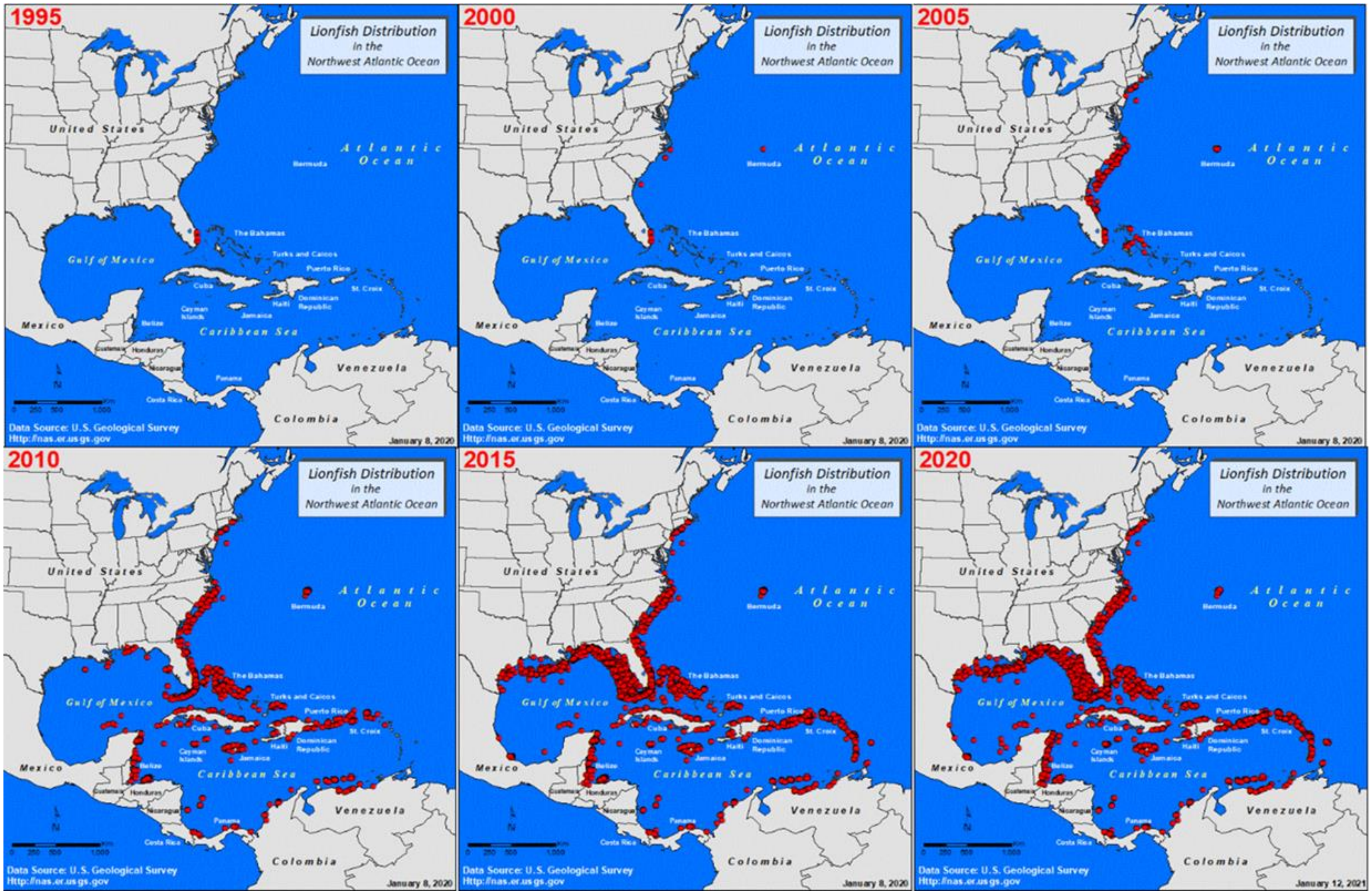

Figure 2.1. Progression of the lionfish invasion in the western Atlantic from 1995-2020 as indicated by lionfish sightings reported to the US Geological Survey Non-indigenous Aquatic Species (USGS-NAS) database[4].

[4] https://www.usgs.gov/centers/wetland-and-aquatic-research-center/science/lionfish-distribution-geographic-spread-biology

including Barbados in late 2011 and Tobago, the last island to be invaded, in 2012 (Figure 2.1).

The remarkable invasion of the western Atlantic has been aided by the fact that *P. volitans* has been exceptionally successful at dispersing across locations and at adapting to alien environments. They have low mortality rates, having few if any native predators or parasites, and have voracious appetites, growing fast and maturing within their first year of life (Morris et al. 2009). Once mature, they produce free-floating gelatinous egg masses containing thousands of eggs every few days, which hatch into planktonic larvae with a free floating 20-35 day pelagic phase (Ahrenholz and Morris 2010). Owing to the aforementioned characteristics, the lionfish is considered a potentially significant ecological threat to the coral-reef ecosystems of the Caribbean through the consumption and displacement of native prey species and potential competitors (Albins and Hixon 2008, Green et al. 2012) and resultant changes in the species composition on reefs. Of particular concern are the parrotfishes and other key herbivore populations, such as surgeonfishes, which provide two main types of services to humans. First, they graze large amounts of algae on the reefs, playing a key role in maintaining healthy and attractive reefs (Bellwood et al. 2004), responsible for coastal protection and supporting high biodiversity and recreational diving, an industry that generates millions of dollars annually in Barbados (Gill 2014). Second, these herbivores are among the most important target families in the nearshore fisheries of Barbados, providing important income to fishers and fish protein to Barbadians, in small scale fisheries that contribute several hundred metric tonnes to the fish landings in Barbados annually (Schuhmann et al. 2011, Gill et al. 2018, 2019).

The invasion was ultimately viewed as a potentially substantial economic threat to the countries whose economies rely heavily on healthy reefs, such as Barbados. The ultimate impact of this scenario would be the loss of vital ecosystem services provided by reefs, including those critical to Barbados (e.g. provision of food and livelihood security to reef fishers and the wider community; aesthetic quality and value to the watersports tourism sector; provision of white sand beaches and coastal protection). In addition, lionfish have venomous spines that can pierce through the human skin and cause envenomation, which constitutes a serious health threat to humans, requiring immediate treatment, and therefore posing an additional social burden, as the invasion has spread across the region. Badillo et al. (2012) reported that in humans, lionfish venom has been found to have many effects ranging from mild reactions, including swelling, dizziness, local numbness and sweatiness to rare, but more serious symptoms including nausea, vomiting, extreme abdominal pain, temporary paralysis of the limbs, loss of consciousness, heart complications and even death.

2.1.2 The Barbados lionfish project (2012)

In response to Barbados' first confirmed sighting in November 2011, the 'Barbados Lionfish Project' was undertaken by CERMES and the Department of Biological and Chemical Sciences of the UWI, in collaboration with partners including the Coastal Zone Management Unit (CZMU) and the Fisheries Division (FD). The project was conceived after the Biodiversity Working Group of the Natural Heritage Department (NHD) of the Government of Barbados drafted a 'Lionfish Response Plan for Barbados' (Brathwaite et al. 2011), in which lionfish were predicted to arrive imminently and begin to establish themselves in Barbados from late 2011. The Response Plan also highlighted the need for a collaborative approach among government institutions, the private sector and the University of the West Indies, with the latter primarily adopting a research role.

The project seized the opportunity to rigorously assess the structure and function of parrotfish and surgeonfish communities in the reefs of Barbados before the establishment of the invasive lionfish. The project was principally designed to collect and archive 'pre-lionfish invasion' baseline data on the reef and reef fishery, thereby providing a valuable opportunity to quantitatively examine the real impact of lionfish in the 'post-lionfish invasion' period. Key project objectives were:

1. To quantify the importance of key herbivore families (parrotfish and surgeonfish) landings in the trap and spear fisheries before the arrival of lionfish.
2. To characterize parrotfish and surgeonfish communities on the reefs of Barbados before the establishment of the invasive lionfish.
3. To document the arrival and spread of lionfish around the reefs of Barbados.
4. To determine the genetic identity (species and source population) of the Barbados lionfish population.

Key findings from the project included (see Oxenford and Valles 2014 for full results):

- The catches of both spear fishers and trap fishers were multispecies and showed minimal seasonal changes regarding availability or choice of target species.
- Parrotfishes and surgeonfishes made up more than half the catch of both spear and trap fishers with parrotfish family being the most important (based on proportions of individuals caught).
- Mean sizes of fish caught by spear fishers and trap fishers were 26.7 cm fork length (FL) and 19.4 cm FL respectively.
- The catch per unit effort (mean catch per fisher per trip) was 10.2 kg for trap fishers and 5.6 kg for spear fishers.
- Relative densities (i.e. total number of individuals per 600 m^2) as well as individual mean size (as FL) of parrotfishes and surgeonfishes differed among reefs located along the west and south coasts of Barbados.
- The rate of lionfish sightings after the first confirmed sighting (November 2011) remained low over the subsequent nine months, but showed a marked increase from August 2012. After one year (November 2012) 54 confirmed lionfish specimens had been collected.
- Genetic analysis of the Barbados lionfish population indicated a single species (*Pterois volitans)* lionfish invasion in Barbados.

2.2 Assessment of native reef fishes and lionfish

2.2.1 Methodology

Underwater surveys, using SCUBA gear, were conducted on the same ten coral reef sites as previously surveyed in the 2012 pre-invasion baseline survey (Figure 2.2) to assess current density, size and species composition of native reef fish and of the lionfish. These reef sites represent replicates of the three reef types representative of Barbados' reef diversity, namely nearshore fringing and patch reefs, and offshore bank reefs, and have received different levels of lionfish management (i.e. targeted removal, commercial fishing, and *ad hoc* culling by recreational SCUBA divers).

The same survey methodology as used in the pre-invasion surveys was utilised in this study to provide results that are directly comparable to the pre-invasion baseline data (see Oxenford and

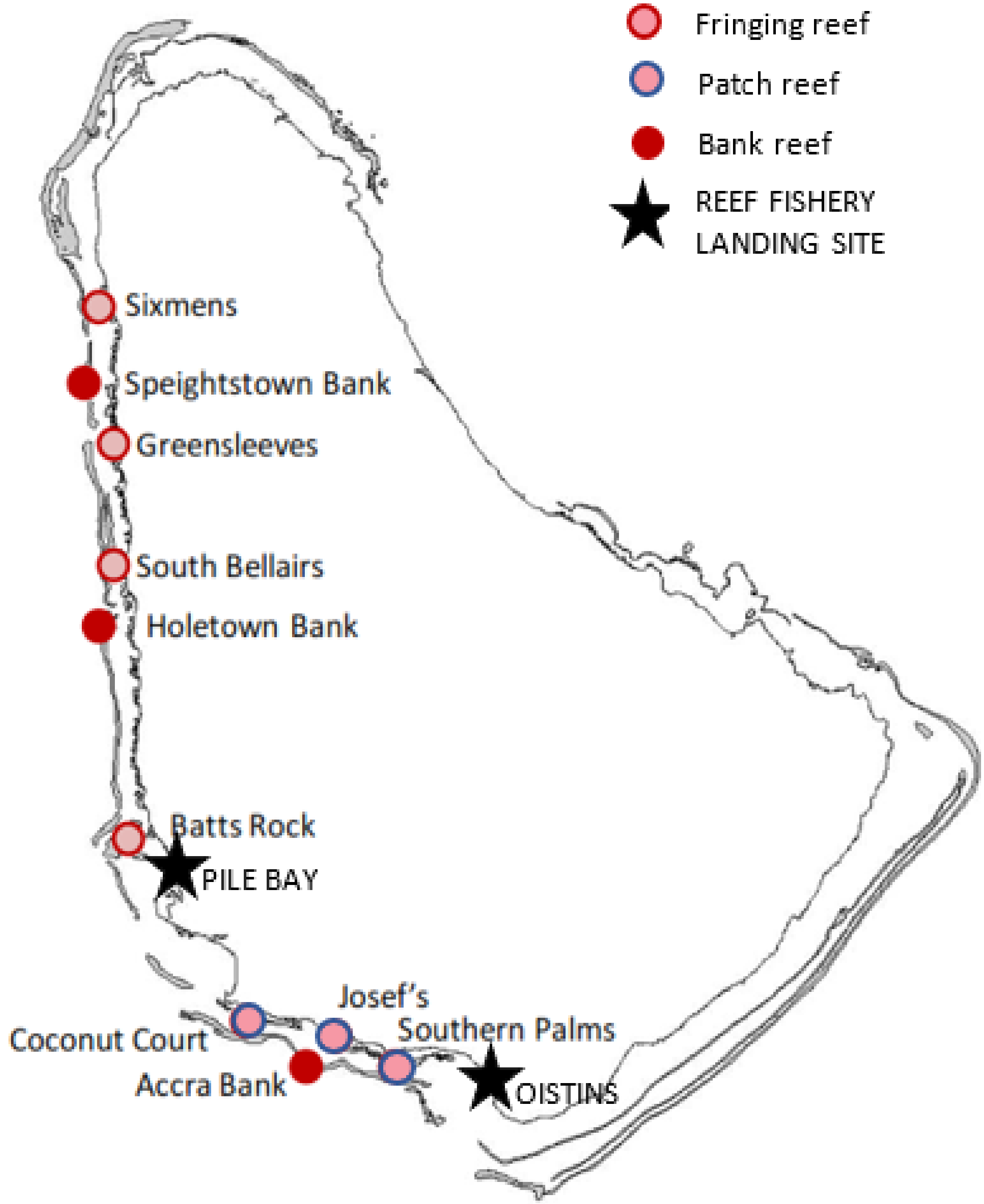

Figure 2.2. Location of the ten coral reef sites on the west and south coast of Barbados used for the repeat surveys of native coral reef fish and lionfish. Black stars indicate the main reef fish landing sites where fisher surveys were conducted (see Section 2.2.2).

Valles [2014] for original survey methodology, summary given in Appendix 1). An additional component comprising roving day and night surveys specifically for lionfish was also added to the previous belt transect survey methodology. These additional lionfish roving surveys were to ensure that we did not miss lionfish, given anecdotal reports that they now tend to hide during the daytime and have become more nocturnal as a response to fishing pressure. These roving surveys were initially conducted at all survey sites during the daytime immediately following the transect surveys and were carried out again at dusk (nighttime). However, guided by the results of the first round of surveys, we discontinued these additional day and night roving surveys on all but the three bank reefs for the second and third round of surveys. The methods are briefly summarized here:

<u>Belt transect surveys:</u> As before, underwater transect surveys were repeated every four months for one year (i.e. June 2021, October 2021 and February 2022), to ensure that any seasonal variation in fish populations were accounted for. A total of ten 30 m x 2 m belt transects were surveyed at each of the ten coral reef sites during each of the three survey efforts, using the same GPS coordinates as in the previous 2012 survey (Appendix 2). Specific species of large mobile fish (LF: parrotfish, surgeonfish, adult yellowtail damselfish and lionfish) and slow-moving fish (SF: bicolor damselfish, other damselfishes except adult yellowtail damselfish, bluehead wrasse, yellowhead wrasse, clown wrasse, slippery dick and other wrasses) were recorded by species and size (to the nearest cm) and other critical grazers, namely *Diadema antillarum* urchins were also counted.

<u>Roving surveys</u>: Newly added, were roving surveys specifically to record lionfish. These were done by a single diver (accompanied by a dive buddy) swimming to and fro over the reef for approximately 20 minutes searching the same overall area as surveyed by the transects and recording the number and size of all lionfish seen. These surveys were done twice on a given survey day and site; they were first conducted immediately after the belt transect surveys, during daylight hours, and they were then repeated right after dusk (which extended into the night) with the help of underwater torches. A key objective of these surveys was to assess whether lionfish abundance estimates differed between the daylight and dusk period, since lionfish are known to become more active during the dusk period (McCallister et al. 2018). Greater lionfish abundance estimates during the dusk period would support this as the best time to hunt lionfish. These daylight vs dusk roving surveys for lionfish were done at the ten sites in the first round of surveys. However, in the following two rounds of surveys, they were only repeated in the three deep bank reefs sites as these were the sites with the consistently highest abundance of lionfish.

2.2.2 Effects of lionfish on reef fish community structure

Visual inspection of fish biomass boxplots, when data were pooled across the ten sites, suggested no overall differences in fish biomass between the pre-invasion and post-invasion periods for all fish groups except lionfish (Figure 2.3). This was evidenced by the substantial overlap between both periods in the boxes around the median values (which show 50% of the range in observed abundance values) for all fish groups examined (Figure 2.3). For lionfish, increases in biomass during the post-invasion period were evident given that no lionfish were recorded during the pre-invasion period, whereas they were recorded in 11.8% of transects during the post-invasion period (Figure 2.3, last panel).

However, when data were broken down by reef type, small differences in fish biomass between the pre- and post-invasion periods became evident in some cases. The extent of these differences

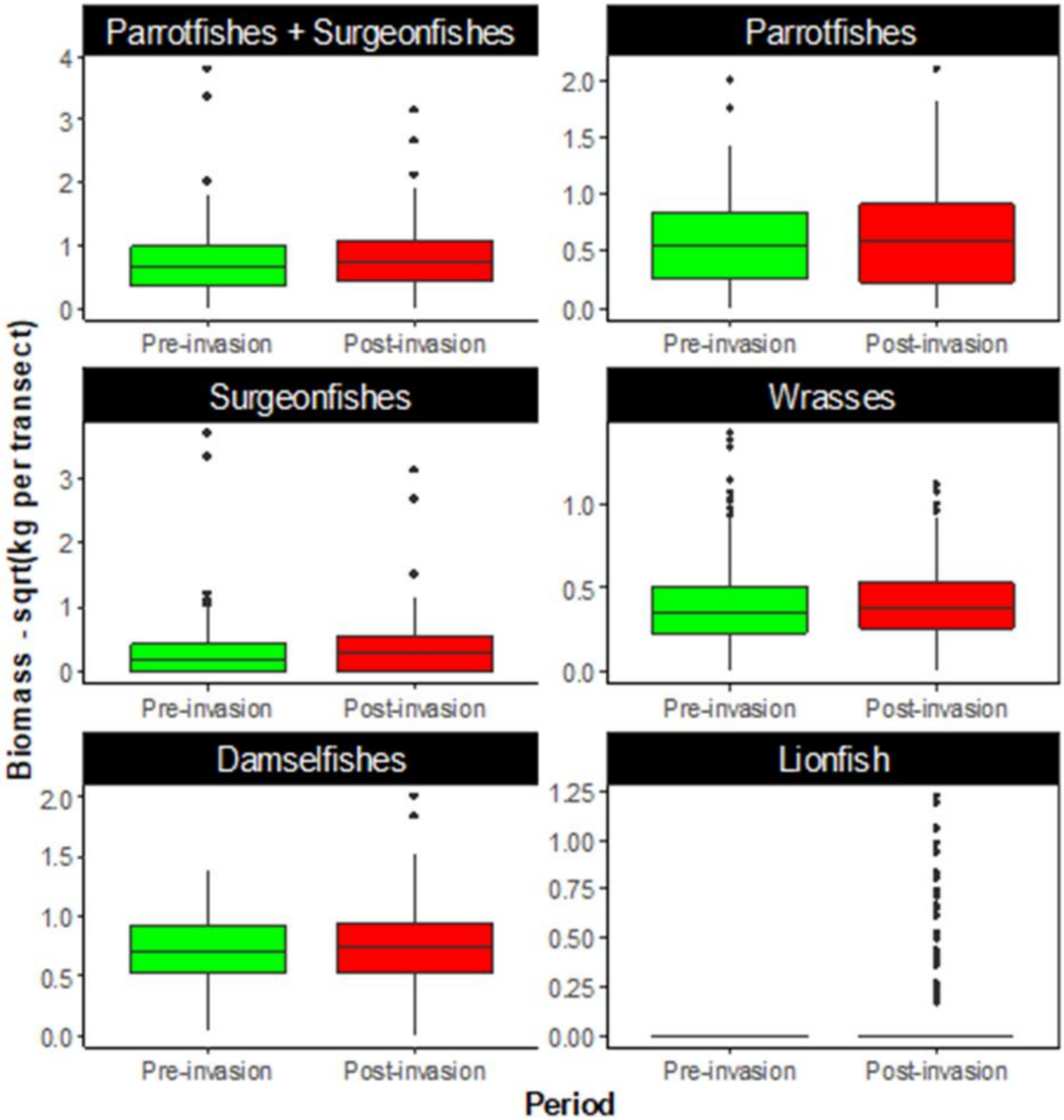

Figure 2.3. Boxplots of (square root-transformed) fish biomass during the pre-and post-invasion periods for key fish groups, i.e. herbivores (parrotfishes + surgeonfishes, parrotfishes, surgeonfishes) and other key lionfish prey species (wrasses, damselfishes), and lionfish. Transect data were pooled across sites and seasons. Horizontal line shows median value, boxes show 1st and 3rd quartiles, whiskers and dots show full range of data points.

depended on reef type and fish group (Figure 2.4). For example, there was an overall trend towards higher damselfish biomass during the post-invasion period on patch reefs, whereas the opposite trend was evident on bank reefs (Figure 2.4). There was also an overall trend towards higher (albeit marginally) parrotfish and wrasse biomass during the post-invasion period on bank reefs, whereas the opposite trend was evident on patch reefs (Figure 2.4). In contrast, there was an overall trend of increase in surgeonfish biomass during the post-invasion period for all three reef types (Figure 2.4). These data also revealed that during the post-invasion period, lionfish were overall more frequently found on bank reefs (32.2% of transects) than on patch reefs (4.4%) and fringing reefs (2.5%) (Figure 2.4).

A more detailed statistical comparison using a multivariate analysis of variance (MANOVA) confirmed that the fish assemblage structure (based on the biomass of surgeonfishes, parrotfishes, damselfishes and wrasses) differed significantly among reef types, among sites, and between the pre- and post-invasion periods (Table 2.1). Moreover, the magnitude of the differences in fish assemblage structure between the two time periods depended on the site, as evidenced by the significant interaction between period and site (Table 2.1). However, and importantly, the MANOVA results also indicated that these differences were not associated with lionfish presence, as evidenced by the lack of statistical significance for this variable (Table 2.1). Thus, overall, this analysis revealed considerable spatiotemporal variability in fish assemblage structure when data were compared among reef types, sites, and between periods; however, such differences were not linked in any way to lionfish presence.

Conclusion: ***Although there was substantial variability in reef fish assemblage structure across sites and between the pre- and post-invasion period, we found no evidence of lionfish effects on these fish assemblages.***

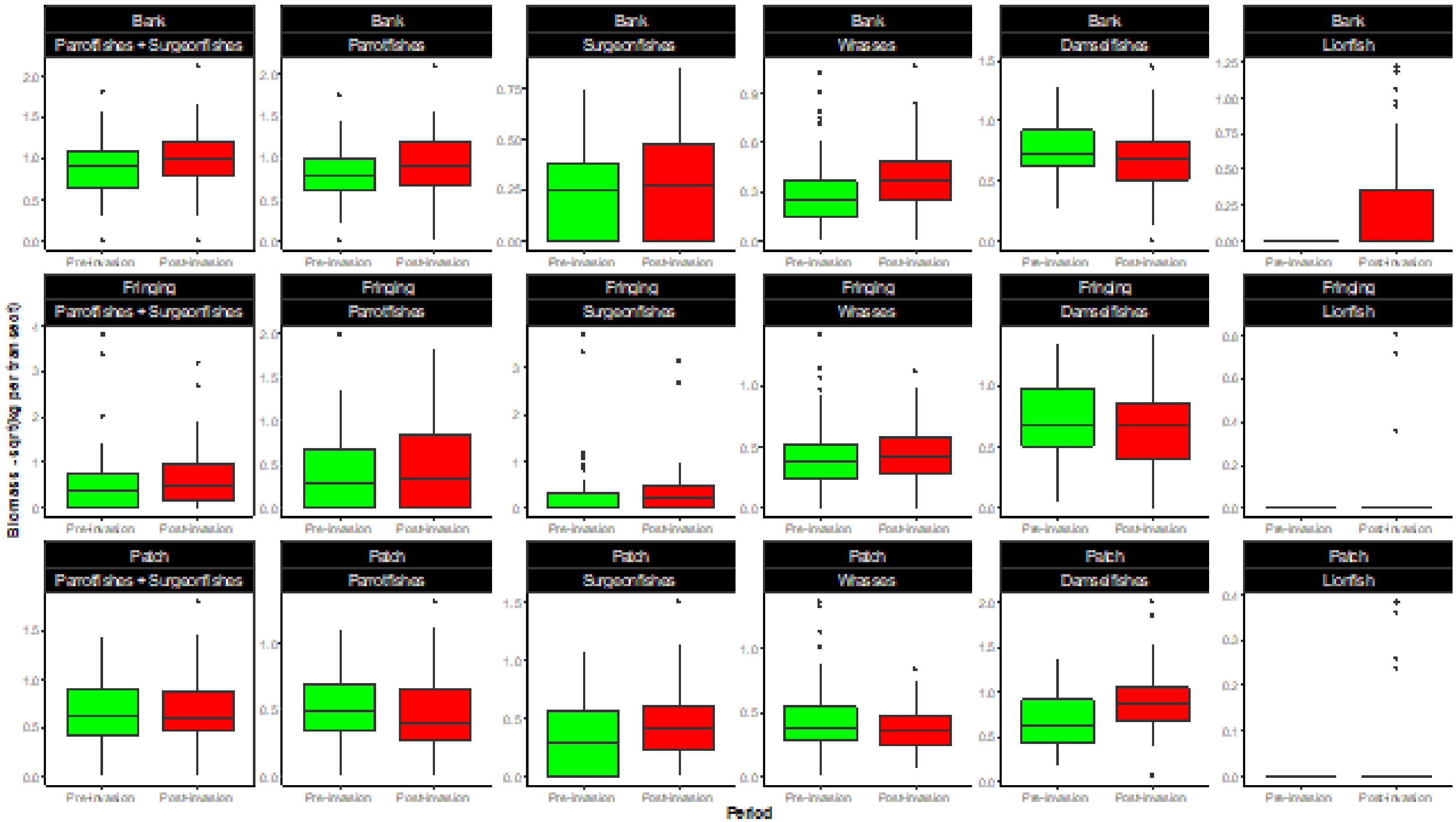

Figure 2.4 Boxplots of (square root-transformed) fish biomass during the pre-and post-invasion periods for key fish groups, i.e. herbivores (parrotfishes + surgeonfishes, parrotfishes, surgeonfishes), other key lionfish prey species (wrasses, damselfishes), and lionfish at bank reefs (top panel), fringing reefs (middle panel) and patch reefs (bottom panel). Transect data were pooled across sites and seasons within each reef type. Horizontal line shows median value, boxes show 1st and 3rd quartiles, whiskers and dots show full range of data points.

Table 2.1. Results of the MANOVA testing for the effect of reef type (bank, fringing, and patch), site, season (winter vs summer vs fall), period (pre-and post-invasion) and lionfish presence on reef fish assemblage structure at the ten study sites. Bold font indicates factors with significant effects ($p<0.05$).

Factor	df	SS	R2	F	p-value
Reef type	2	6.798	0.088	33.34	**0.001**
Site	7	8.62	0.111	12.08	**0.001**
Season	2	0.156	0.002	0.77	0.570
Period	1	0.872	0.011	8.55	**0.001**
Lionfish presence	1	0.037	0.000	0.37	0.771
Site:Season	18	2.194	0.028	1.20	0.176
Site:Period	9	1.652	0.021	1.80	**0.012**
Season:Period	2	0.213	0.003	1.04	0.405
Site:Season:Period	18	2.367	0.031	1.29	0.089
Residual	535	54.54	0.704		
Total	595	77.45	1.000		

2.2.3 Effect of daytime (daylight vs dusk) on lionfish abundance estimates

We examined whether lionfish abundance estimates differed between roving surveys conducted during the daylight versus dusk periods. Here, we focus only on data from the three bank reef sites, namely Holetown Bank, Speightstown Bank and Accra Bank, since these were the reef sites with the highest overall abundance of lionfish (see Section 2.2.2, Figure 2.4).

Overall, we found that whether or not lionfish abundance estimates differed between daylight and dusk periods depended on the site (ANOVA; d.f.= 2,6, $F=9.675$, $p=0.01326$). More specifically, at the site with the lowest lionfish abundance, Holetown Bank, lionfish were recorded in higher numbers at dusk than during the daylight period (Figure 2.5). This was not the case for the other two sites (Figure 2.5), which supported differences among sites in lionfish diel activity.

Conclusion: ***The higher lionfish abundance estimates during the dusk period at Holetown bank indicates that lionfish are more active during this period at this site (and thus more likely to be detected by divers). Holetown bank also had considerably less lionfish than the other two sites, likely because of higher fishing pressure at this shallower site. It is thus likely that higher fishing pressure at Holetown bank might affect lionfish behavior whereby it becomes less active during the day, when spear fisher presence is most likely.***

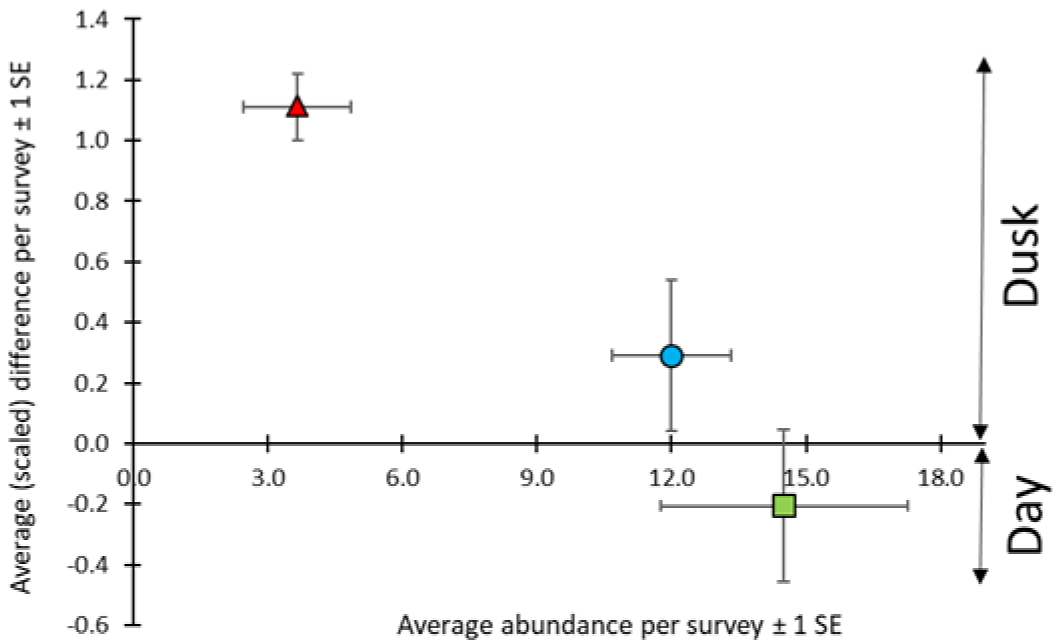

Figure 2.5. Average difference in lionfish abundance between daylight and dusk roving surveys (n=3) at three deep bank reef sites. These differences are expressed relative to the average abundance of lionfish at each site to account for marked differences in lionfish abundance among sites. Values well above the zero line indicate higher abundance during the dusk period; values well below the zero line indicate higher abundance during the daylight period; values near the zero line indicate no difference between periods. Red triangle is Holetown bank; blue circle Accra bank; green square is Speightstown bank.

2.3 Assessment of reef fishers' catch composition

2.3.1 Methodology

The same survey methodology as used in the pre-invasion surveys was utilised to provide results that are directly comparable to the pre-invasion baseline data (see Oxenford and Valles [2014] for original survey methodology). Repeat surveys of reef fisher catches (catch per unit effort, species composition of catches, size of individual fish) were conducted at the same two main reef fish landing sites (Oistins and Pile Bay, Figure 2.2). As before, both spear fishers and trap fishers were targeted, and two surveys were conducted; one within the winter pelagic fishing season (October - December 2021) and the second within the summer off-season for pelagic fishing (April - July 2021). Wherever possible, fishing effort data were also collected from each trap fisher whose catch was recorded and included: total number of fishers on the boat, length of trip, number of traps hauled, location range of traps, trap depth, trap type, mesh size, number of funnels, soak time and total weight of catch. Similarly, wherever possible, fishing effort data were collected from each spear fisher whose catch was recorded and included: total number of fishers on the boat, length of trip, fishing location, fishing depth and total weight of catch (where possible). All fish within the sample obtained from a fisher were identified to the species level and fork length (FL) measured in centimeters using a measuring board. Photos were taken where possible, especially of unknown species to be identified later or of unique species and/or observations. Landed lionfish were also measured (as FL) where possible. To ensure sufficiently large sample sizes, data from the two landing sites and the two seasons were pooled to compare between the pre- and post-invasion periods.

2.3.2 Effects of lionfish on fishing yields and catch composition

No evidence of changes in fishing yields (as weight of fish landed per unit of fishing effort) between the pre- and post-invasion periods was found (Figure 2.6). In fact, pre- and post-invasion estimates were remarkably similar for both spearfishing (Figure 2.6A) and trap fishing (Figure 2.6B) and so did not differ significantly between periods (Welch's t-test: spearfishing: $t = -0.406$, $df = 26$, $p = 0.687$; trap fishing: $t = -0.199$, $df = 48$, $p = 0.842$).

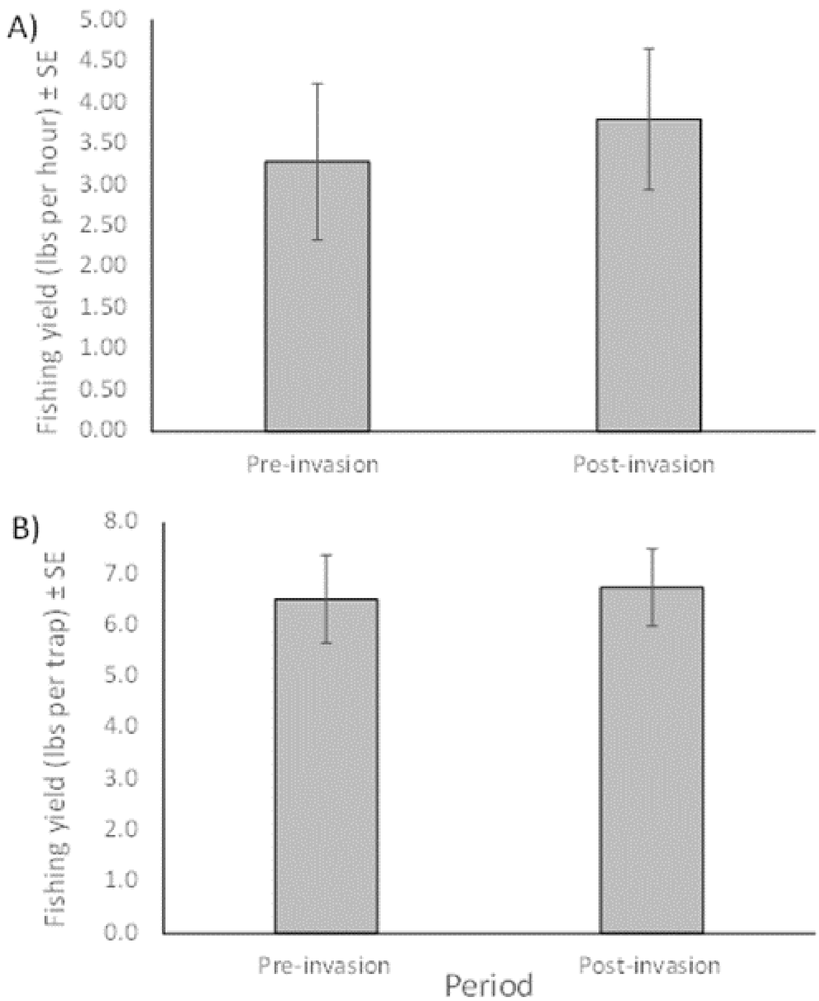

Figure 2.6. Average fishing yields for (A) spearfishing during the pre-invasion period (N=11 fishing trips, average yield 3.3 lb/hr) and post-invasion period (N=27 fishing trips, average yield 3.8 lb/hr) and for (B) trap fishing during the pre-invasion period (N=22 fishing trips, average yield 6.5 lb/trap) and post-invasion period (N=34 fishing trips, average yield 6.7 lb/trap).

For trap fishing, the numerical composition of the catch (by family) was broadly similar between the pre- and post-invasion periods (Figure 2.7). In both periods, the same ten families accounted for 97% or more of the catch (Figure 2.7). Moreover, in both periods parrotfishes (Scaridae) were the most numerically abundant fish group, followed by grunts (Haemulidae), goatfishes (Mullidae), jacks (Carangidae) and surgeonfishes (Acanthuridae), which jointly accounted for approximately three quarters (75%) of the catch (Figure 2.7). A MANOVA indicated that there were no significant differences in catch composition between periods when the 19 families recorded were included in the analysis (MANOVA: d.f.=1, 119, pseudo-F=2.063, p=0.069). This was the case even though some lionfish were recorded in traps during the post-invasion period. However, this was only in a small fraction of fishing trips (7%) and represented only 0.64% of all post-invasion fish recorded in traps.

In contrast, for spear fishing, there was a marked difference in numerical catch composition between the pre- and post-invasion periods (Figure 2.8). Most notably, lionfish (Scorpaenidae) accounted for nearly a quarter of the fish in the catch (23%) during the post-invasion period, whereas it was completely absent in the pre-invasion period (Figure 2.8). Lionfish thus became the second most important fish group caught by spear fishers after parrotfishes (Scaridae). A MANOVA indicated that these differences in catch composition between periods were highly statistically significant when the 17 families recorded were included in the analysis (MANOVA: d.f.=1, 59, pseudo-F=7.60, p<0.001).

Conclusion: ***We found no evidence of differences in fishing yields between the pre- and post-invasion period for neither trap fishing nor spearfishing. We also found no evidence that the composition of the catch in traps differed between the pre- and post-invasion period, even though lionfish were occasionally recorded in the catch. In contrast, we found very strong differences between periods in the catch composition of spear fishers, whereby lionfish had become the second most important fish group, after parrotfishes, in the post-invasion period. Because spearfishing yields were similar between the pre-and post-invasion period, this suggests that the targeting of lionfish has helped relieve some of the spearfishing pressure on other reef fish groups, including parrotfishes.***

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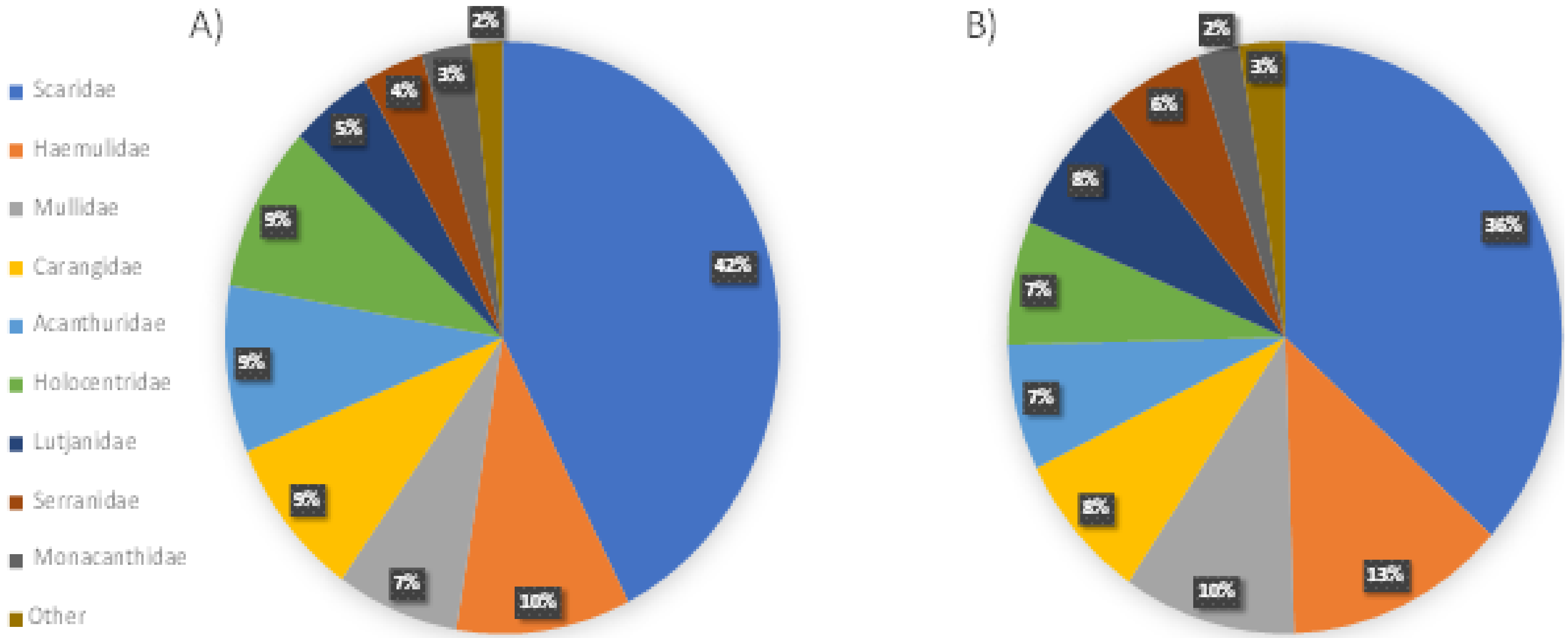

Figure 2.7. Average catch composition (by fish numbers within families) in fish traps during the A) pre-invasion period (N=62 fishing trips) and B) post-invasion period (N=58 fishing trips).

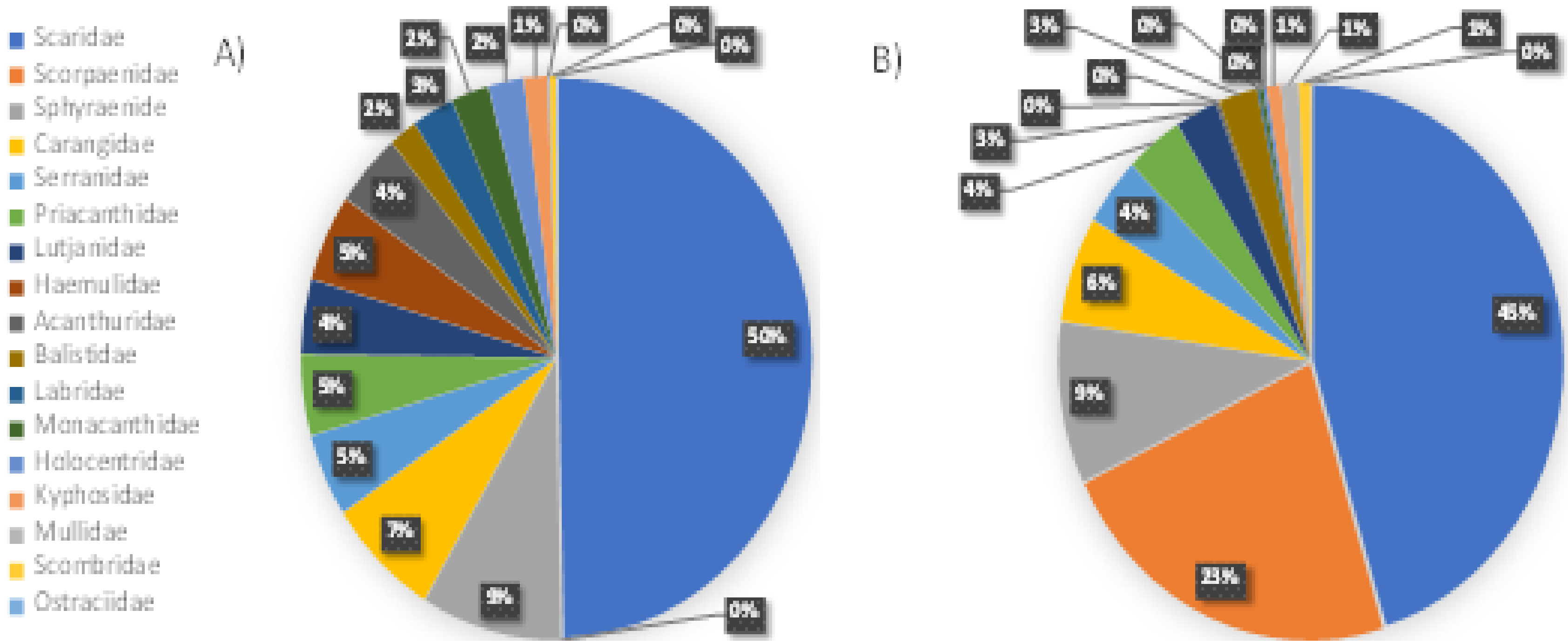

Figure 2.8. Average catch composition (by fish numbers within families) of spearfishing during the A) pre-invasion period (N=32 fishing trips) and B) post-invasion period (N=28 fishing trips).

2.4 Assessment of lionfish management

2.4.1 Methodology

Potential impacts of lionfish on native reef fish populations (especially key grazers such as the parrotfishes and surgeonfishes) were assessed by direct comparison of pre- and post-invasion reef fish surveys across ten coral reef sites (Section 2.2). Likewise, a direct comparison of catch rates (fishing yields) and species composition of commercial reef fishers as determined in pre- and post-invasion surveys was used to assess any changes in catches of native reef species and thus any positive or negative impacts of lionfish on fishers (Section 2.3).

The effectiveness of different management strategies to remove lionfish (commercial fishing, recreational fishing by SCUBA divers and targeted culling) at these sites were examined using data obtained from a number of different sources. Geospatial data (overlay maps) documenting the variation in intensity of commercial reef fishing around the island of Barbados were obtained directly from the authors (see Figure 2.9). Data on intensity of spearfishing of lionfish by recreational SCUBA divers were obtained via interviews with the islands' dive shop staff (Appendix 3).

Assessment of an additional management tool, namely the use of deep traps to help control lionfish numbers below SCUBA diving depth was also trialed with two local trap fishers from Pile Bay with expertise in deep trap fishing, over a 3-month period from August to October 2021. A total of 17 traps were utilised by the two fishers, who retrieved and reset their individual traps (i.e. 7 and 10 traps respectively) once a week (provided the traps were found). Traps were standard square traps (2 funnels) of wire mesh (1.25 inches) on a wooden frame measuring between 4 x 7 ft and 5 x 8 ft, and were set in water depths between 36 – 100 m, offshore from the bank reef (Figure 2.10). The traps were unmarked at the surface and were retrieved using a grapple hook and rope dragged across the area where set.

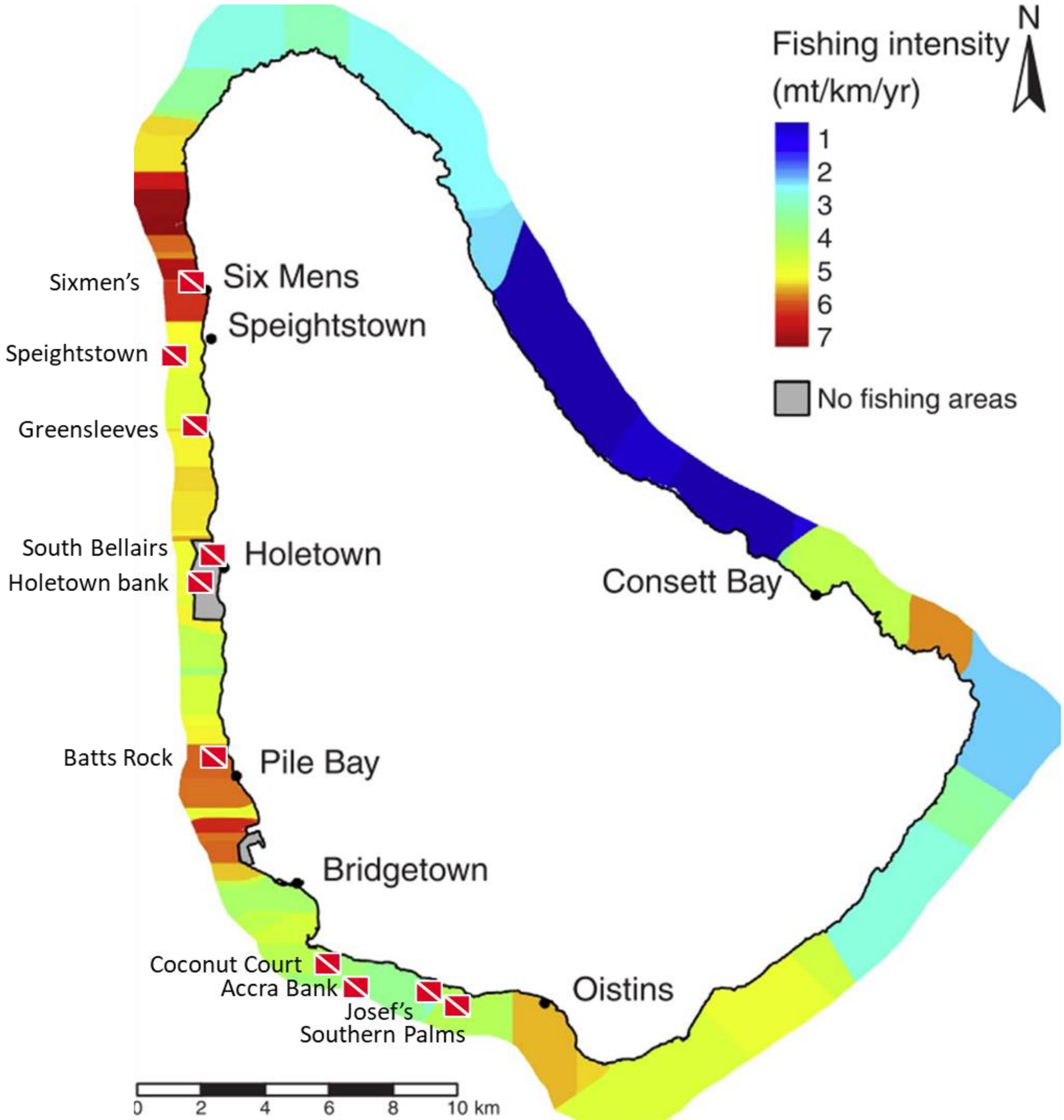

Figure 2.9. Map showing the variation in commercial reef fish fishing pressure around the island (from Gill et al. 2019) and overlay of reef fish survey sites used in this study.

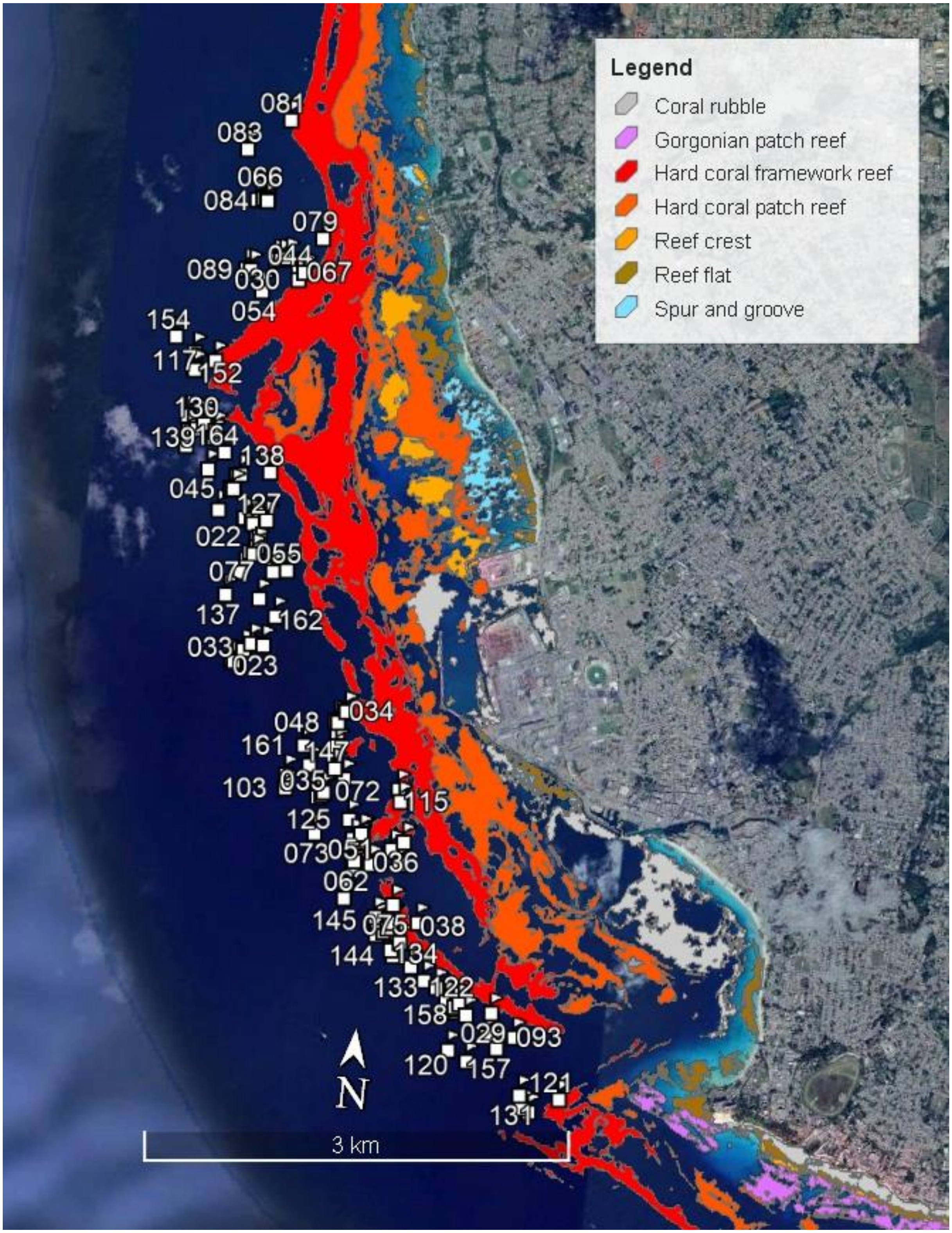

Figure 2.10. Indicative map of deep trap set locations (excerpted from Pile Bay deep trap fisher's GPS log), showing trap positions relative to the coral reef habitat. Base map from Google Earth, habitat layer from 2015 dataset (see Baldwin et al. 2019).

2.4.1 Efficacy of lionfish management tools and strategies

2.4.1.1 Removal through capture on shallow reefs

No relationship was found between observed lionfish abundance at surveyed shallow reef sites (< 30 m deep) and independent estimates of fishing pressure at these same sites (Figure 2.11). These fishing pressure estimates were derived from the fishing pressure map of Gill et al. (2019) (Figure 2.9), which itself was based on fisher surveys and integrated information from all reef fishing gear types before the COVID-19 pandemic.

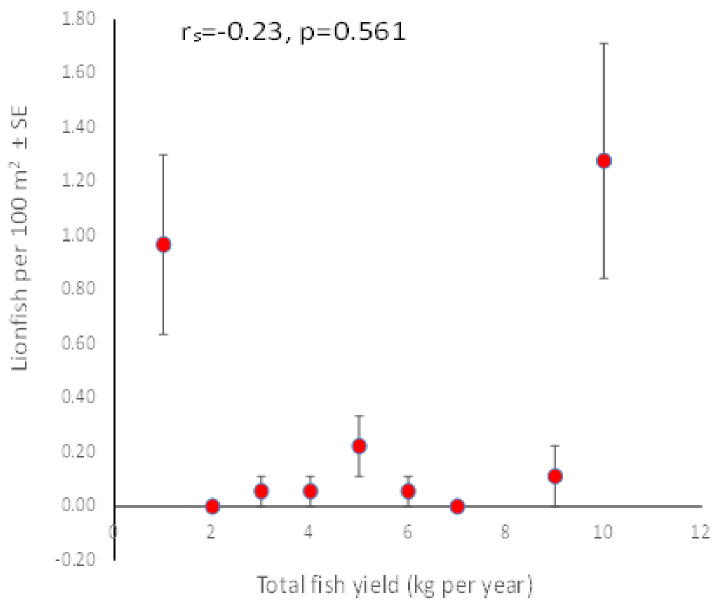

Figure 2.11. Mean lionfish abundance per survey (N=3 surveys), based on the post-invasion reef fish surveys, as a function of the fishing pressure estimates derived from Gill et al. (2019) for nine of the ten coral reef sites. South Bellairs was not included since it is a no-take zone, but lionfish culling does take place, so no comparable fishing effort data are available for that site. The Spearman rank correlation coefficient and corresponding p-value are also shown.

However, we did find a statistically significant negative relationship between lionfish abundance and depth across the ten study site reefs (Figure 2.12). Depth is a well-known proxy for spearfishing pressure (Tyler et al. 2009) and spearfishing is common in Barbados (Simpson et al. 2014, Gill et al. 2019), where lionfish has become an important component of the catch (see fisher data section, Figure 2.8).

An important role of spearfishing was further supported by the results of the dive operator surveys. Key informants from nine dive operators that were active before and/or after the COVID-19 pandemic were asked to identify the dive sites they most frequently visited along Barbados' coastline before and after the pandemic (Figure 2.13) and to provide crude estimates of lionfish abundance at these same sites. Nearly all these dive operators (89%) reported being involved in the spearfishing of lionfish during their dive site visits. Based on these reports, it was possible to

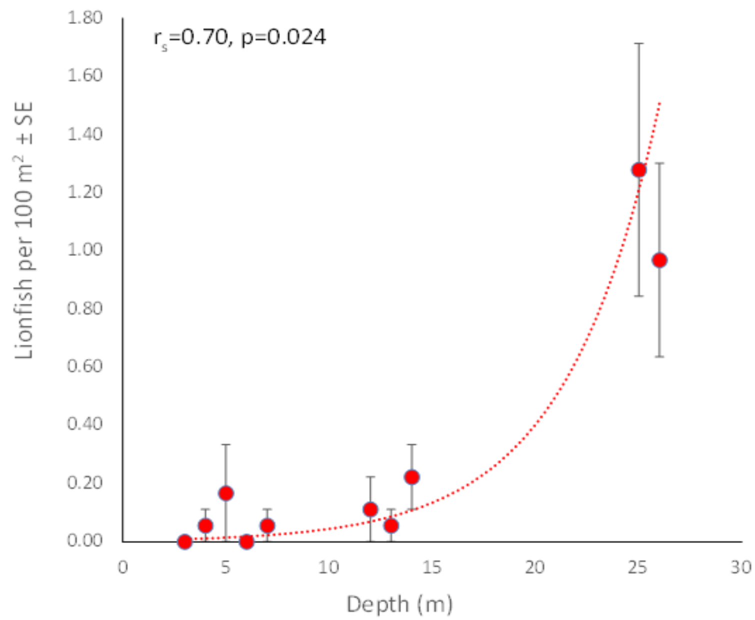

Figure 2.12. Mean lionfish abundance per survey (N=3 surveys) at the ten coral reef sites, based on the post-invasion reef fish surveys, as a function of site depth. Dashed red line indicates the statistically significant increasing trend. The Spearman rank correlation coefficient and corresponding p-value are also shown.

identify a statistically significant negative relationship between the reported total frequency of visits at these dive sites (all dive operators combined; Figure 2.13) and reported estimates of lionfish abundance by the dive operators at these same sites (Figure 2.14; Table 2.2). This relationship remained unchanged before and after the covid-19 pandemic, as evidenced by the lack of statistical significance of the pre-and post-pandemic period term and its interaction with number of dives (Table 2.2).

Moreover, a follow-up survey involving eight of the nine dive operators indicated that they all (100%) believed that spearfishing was helping reduce lionfish numbers at these sites; most (75%) reported that it took several weeks for lionfish to replenish the sites after culling. Most (86%) also reported a change in lionfish behavior over time whereby lionfish had become more evasive in response to divers. Most (67%) also reported an increase in lionfish abundance due to the COVID-19 lockdowns. In contrast, there was less consensus among dive operators in terms of whether site depth influenced lionfish abundance, with half (50%) reporting that lionfish abundance was higher at deeper sites, in line with the findings of the reef fish surveys (Figure 2.12), whereas the other half reported that depth did not play any role.

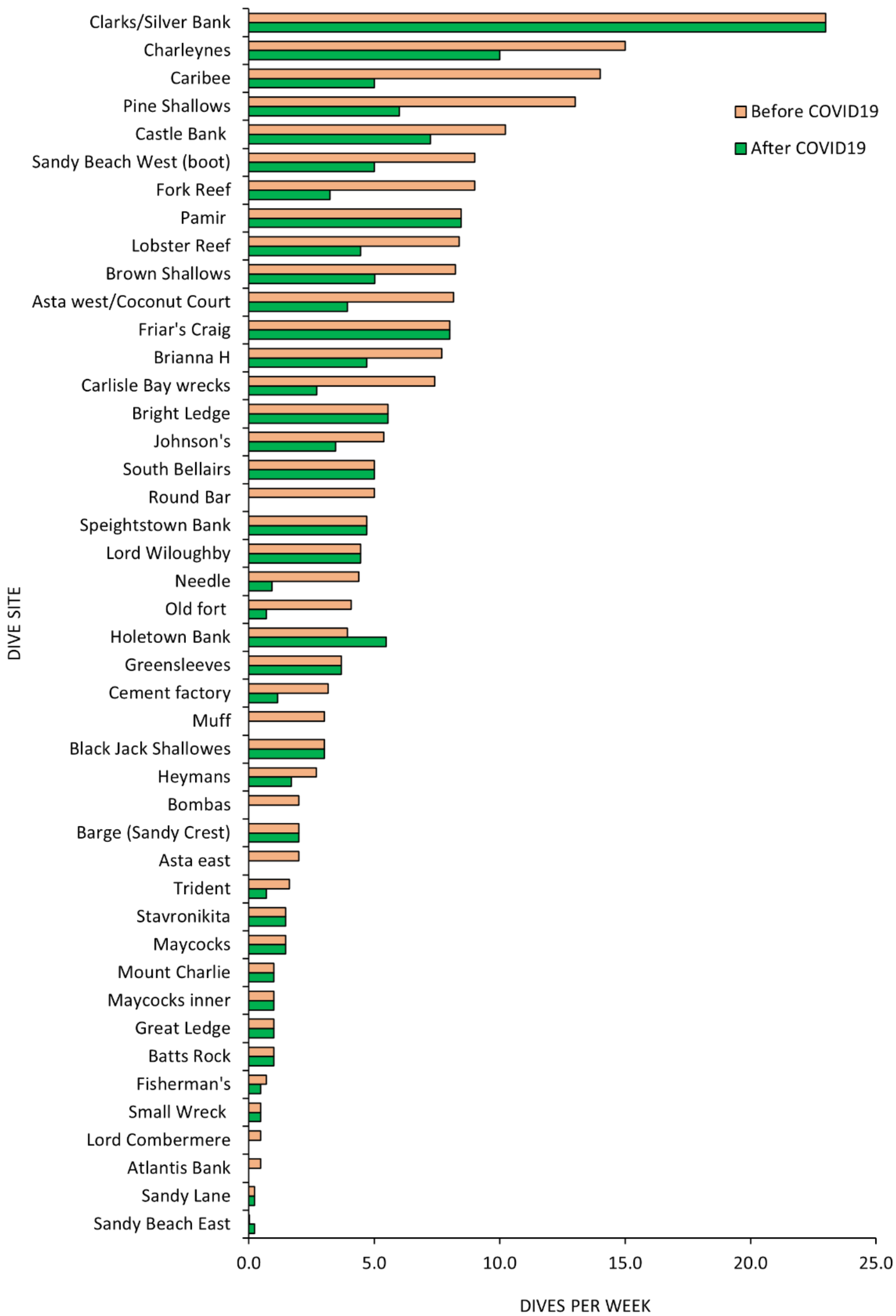

Figure 2.13. Estimated total number of dives per week at the most frequently visited dive sites in Barbados, before and after the COVID-19 pandemic, as reported by the nine most important dive operators on the island, namely, Barbados Blue, Roger’s Scuba Shack, Reefers and Wreckers, Eco Dive, Seahorse Divers, Hightide, Scott Clarke, Gfish and West Side Scuba. Nearly all these dive operators (89%) reported being involved in spearfishing lionfish during site visits.

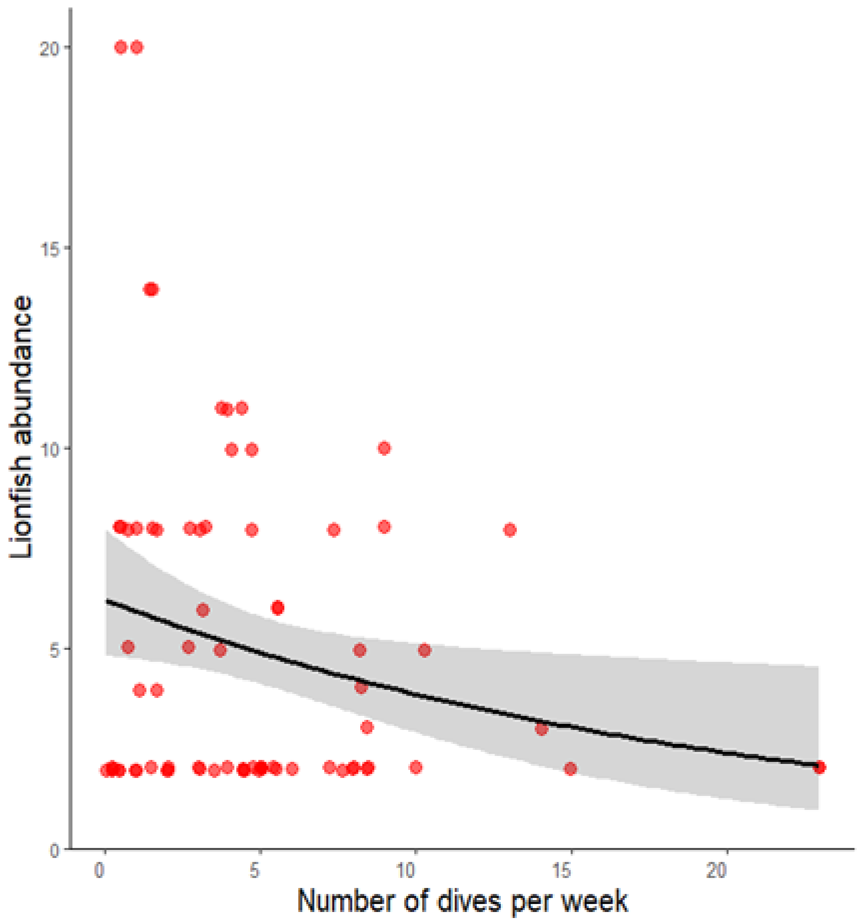

Figure 2.14 Relationship between reported estimates of lionfish abundance at a dive site and reported number of dives per week (involving culling) at the site, based on dive operator interviews. These data were modeled using a negative binomial model; the best fit line and 95% confidence intervals are also shown. See also Table 2.2.

Table 2.2 Results of the Analysis of Deviance for a Negative Binomial model based on the dive operator surveys. The model used reported lionfish abundance at a site as response variable and reported weekly number of dives (all dive operators combined) and period (before vs after the COVID19 pandemic) and their interaction as explanatory variables. Bold font indicates statistical significance ($p<0.05$). D.f – degrees of freedom. See also Figure 2.14.

	D.f	Deviance	Residual D.f.	Residual deviance	p-value
Null	-	-	73	79.624	-
Number of dives	1	5.7017	72	73.922	**0.0170**
Period	1	1.2539	71	72.668	0.2628
Number of dives * Period	1	2.235	70	70.433	0.1349

2.4.1.2 Removal through deep trap fishing

Most demersal and reef fishing (commercial and recreational) in Barbados takes place on shallow (< 30 m deep) reefs. This implies that the deeper reefs will provide a fishing refuge for lionfish, and such refuge will likely help sustain lionfish populations island-wide even under heavy fishing pressure on shallow reefs. We thus evaluated the potential for use of deep traps as a means of targeting lionfish typically beyond the reach of most commercial and recreational trap and spear fishers. This approach involved collecting baseline data on normal deep commercial trap catches between August 7 and October 30, 2021 (N=122 trap hauls) to assess the current contribution of deep trap fishing to lionfish catches. We also modified the current deep trap design with the input of deep trap commercial fishers in an attempt to improve lionfish catches and used these in addition to the normal deep traps between September 25 and October 17, 2021. Figure 2.15 shows the current deep trap design used in Barbados (panel A) and the modified designs utilised by each of the two trap fishers (panels B and C).

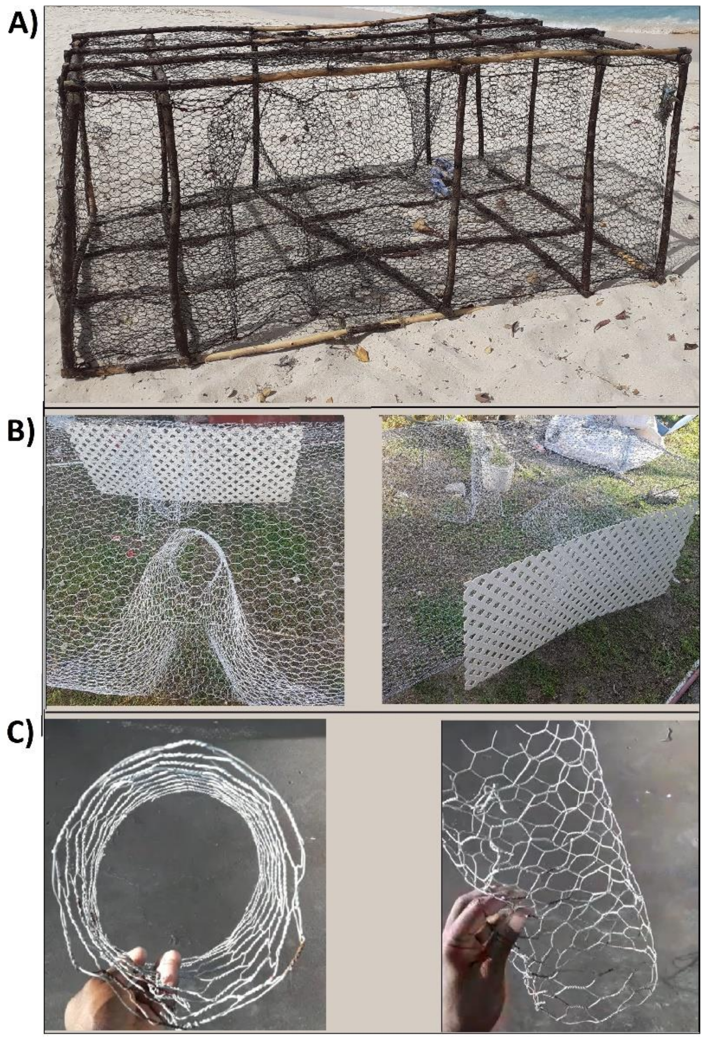

Figure 2.15. Images of deep traps: A) Standard square trap with 2 funnels (horse-neck) and 1.25 inch wire mesh, B) Standard trap modified by fisher to include white lattice opposite the funnel and C) Straight funnel used to replace the horse-neck funnel.

Baseline data derived from the 122 normal deep trap hauls indicated catches of approximately 14.0 fish per trap haul, with only 15.6% of normal deep trap hauls containing lionfish, for an overall average of 0.4 lionfish per haul (Figure 2.16). Against this backdrop, experimental traps performed very poorly as they failed to catch any lionfish and yielded fish catches 3.5 times lower than normal traps, although it is worth pointing out that the total number of experimental trap hauls was small (n=4).

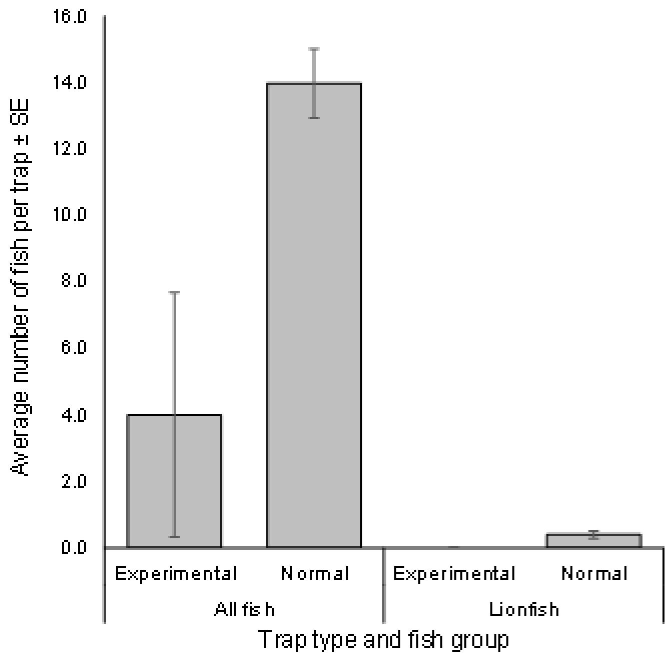

Figure 2.16. Comparison of fish catches per trap haul between normal (n=122) and experimental (n=4) deep trap catches. Catches shown include all reef fish caught as well as only lionfish caught per trap haul.

Conclusion: ***The different lines of evidence presented support a particularly important role of spearfishing by both commercial and recreational divers in helping to control the lionfish population at shallow to moderately deep sites (<30 m deep) to levels where they have no apparent impact on local reef fish communities or reef fisher yields/livelihoods.***

Sites where culling occurs regularly by recreational divers seem to be replenished by lionfish several weeks after culling, likely from lionfish immigration from deeper sites. There is also evidence that lionfish is changing its behavior and becoming more evasive, at least in the very frequently visited sites.

Overall, lionfish catches by commercial deep traps were relatively small, suggesting that deep trap fishing plays a minor role in controlling deep lionfish populations. Moreover, we found no evidence that modifying the trap design improved lionfish catches.

2.4.2 National Lionfish Management Strategy

Although there is no formal Lionfish Management Plan for Barbados, the current management strategy arising from the Lionfish Response Plan (Biodiversity Working Group 2011) for controlling the lionfish population appears to be effective, based on the results of this comprehensive study.

The strategy from the onset of the lionfish invasion was to form a collaborative partnership between the Coastal Zone Management Unit (CZMU), the Fisheries Division (FD) and the University of the West Indies (UWI) to tackle various tasks including:

- Targeted information dissemination (including a 24 hour hotline operated for one full year) to the general public, primary stakeholders (fishers and divers), medical institutions and the media.
- A sustained removal effort through regular culling dives using spears by staff from the Folkestone Marine Park and the CZMU, and by holding competitive Lionfish Derby events.
- Incentivise the development of a commercial fishery through development of a high-priced market for lionfish initially achieved through information sharing and a Lionfish Cook-off event by some of the island’s top chefs.
- Encourage recreational dive clubs and dive shops to remove lionfish as a sustainable activity promoting coral reef health.
- Research and monitoring to increase knowledge and potential impacts of lionfish.

2.4.3 Management Recommendations

Based on the lessons learned in this research and informal discussions with stakeholders we make the following recommendations

2.4.3.1 Overall

- Develop an invasive species policy framework for Barbados which caters generally to the management of invasive species and specifically to the requirements for managing the invasive lionfish.
- Recognise the need for regionally coordinated lionfish management strategies due to the connected nature of our oceans and seas.
- Develop a formal lionfish management plan for Barbados, based on the Lionfish Response Plan (Biodiversity Working Group 2011) and the current study, which feeds into regional strategies.
- Discourage management approaches that have been elsewhere identified as unsuccessful, namely the training of native fish to become lionfish predators and the use of bounty programs to motivate lionfish hunters (Ulman et al. 2022).
- Reboot the development of markets for lionfish meat (which can be sold in high-end restaurants) and for lionfish byproducts such as the skin, which can be used to make leather, as well as the fins which can be used to make jewelry.
- Promote a data-driven and evidence-based approach to management by establishing collaborations between the different actors of Barbados (e.g. Government, UWI, fishers and dive operators).

2.4.3.2 Public awareness and communication

- Re-initiate wider public awareness programmes (meetings, seminars, newspaper articles, pamphlets, social media postings etc.) to inform, update and engage stakeholders.
- Develop communication channels (e.g. WhatsApp group) with key stakeholders, such as fishers and dive shops, to ensure frequent exchange of information and discourse.

2.4.3.3 Harvest and culling

- Ensure that use of the Hawaiian sling to capture lionfish is allowed to continue if reef fishing (especially spearfishing) is to become more strictly controlled.
- Ensure that lionfish culling by authorized persons can continue in the Folkestone Marine Reserve and any other no-take marine parks that may be designated in the future.
- Maintain support for lionfish derby events organized by the private sector and encourage partnerships with the Tourism Sector to promote such events that diversify the tourism product.
- Continue to promote a spear fishery that targets lionfish.
- Avoid modification of current fish trap designs, as there is no evidence that this improves lionfish catches (higher lionfish catch compositions appear to be influenced by abundance at sites and not trap design).

2.4.3.4 Monitoring and research

- Monitor lionfish abundance through improved collection of commercial catch and fishing effort data by the FD to ensure that lionfish are recorded at the species level. In this way, catch per unit of fishing effort (CPUE) can be used as an index of abundance to measure any changes that may require further management intervention.
- Develop a citizen science programme to allow for the collection of in-water data by fishers and divers (location, depth, abundance, behaviour, appearance, activity etc.) as well as the collection of on-land data (e.g. landing sites, ex-vessel prices, restaurants, recipes etc.).

3. SUN CORAL (*TUBASTRAEA* SPP.)

3.1 Background

Other marine invasive species that are reported to be impacting coral reefs and associated ecosystems in other countries within the Southwestern and Western Central Atlantic are also of concern to Barbados, particularly the sun corals, also known as orange cup corals (*Tubastraea* spp.) considered here, and the seagrass (*Halophila stipulacea*) considered in Section 4.

Scleractinian corals in the *Tubastraea* genus (Family: Dendrophylliidae), namely *Tubastraea coccinea* and *T. tagusensis*, hereafter broadly referred to as sun corals, originated in the Indo-Pacific but can now be found across the Caribbean region and Gulf of Mexico, all the way south to Brazil, where they have become a nuisance in many locations (Creed et al 2017, Fenner and Banks 2004, GISD 2023). The most likely primary source of introduction into the Caribbean region is via attachment and subsequent transport on ships' hulls coming from the Indo-Pacific, with further range expansion within the region facilitated by transport of propagules by currents (Creed et al 2017). Sun corals were recently reported in Barbados on the shipwrecks and Radisson pier system of Carlisle Bay.

Unlike most Scleractinian corals, sun corals are azooxanthellate and ahermatypic, meaning that they do not host the photosynthetic zooxanthellae (and therefore do not require sunlight to thrive) and do not contribute significantly to reef building (Creed et al 2017). They generally inhabit shaded vertical surfaces down to great depths and are known to dominate artificial habitats such as shipwrecks, and cement, steel, or granite structures (Creed et al 2017, Vermeij 2006, GISD 2023). Like most stony corals, sun corals are colonial animals and enlarge their colony size by budding new polyps. Sun corals can also reproduce asexually by producing new polyp buds that break away and form a new genetically identical colony. They are also hermaphroditic and can produce planulae (free floating larvae) by both sexual and asexual means. They typically exhibit multiple reproductive periods throughout the year (Capel et al 2017, Creed et al 2017, de Paula et al. 2014). The colonies can start reproducing with as little as two polyps (Creed et al 2017). Their polyps can also abandon the skeleton and produce a new skeleton elsewhere (Capel et al 2014). Sun corals can also regenerate full colonies from small fragments of live tissue (Luz et al. 2018). Sun corals grow fast and can rapidly overgrow hard benthic substrates, competing with other benthic invertebrates for space (Creed et al 2017). In Brazil, they have been shown to negatively affect native stony corals and affect the composition of native benthic communities (Miranda et al. 2016, 2018, Tanasovici et al. 2020).

3.2 Methodology

3.2.1 Confirming the identity of the sun corals

It is difficult to distinguish among sun coral species in the field. However, examination of the taxonomical features of the calices of specimens collected from Carlisle Bay supported that the species found in Barbados is most likely *Tubastraea coccinea*, given the regular arrangement of the septa (Hoeksema et al. 2019), although it cannot be discarded that *T. tagusensis* might also be present in Barbados.

3.2.2 Mapping the distribution of sun coral

A geo-referenced list of large submerged marine artificial structures deployed along the south and west coasts, from the southernmost location, Oistins, to the northernmost location, the Arawak cement factory in St Peter, was compiled. This list contained thirty-eight structures, which included non-metallic structures such as piers, breakwaters, reef balls, and harbour walls, and metallic ones such as large metal mooring buoys and shipwrecks (Table 3.1, Figure 3.1). Between October 2021 and February 2022, a field surveyor visited thirty-five of these structures by snorkeling or diving and assessed whether sun corals were present or absent, in order to map their distribution.

3.2.3 Assessing population recovery rates of sun coral after removal

To assess rates of sun coral population recovery after removal, a removal experiment was conducted at the Trident shipwreck (Figure 3.2) in Carlisle Bay between Nov 2021 and Feb 2022. This location was chosen for three reasons. First, the lateral sides of a shipwreck represent a physically uniform environment with a relatively smooth surface for sun coral colonization and spread, which should help minimize interference from other potentially confounding environmental factors. Second, the Trident is conveniently located a few hundred meters from shore and at a depth of 17 m; this allows for relatively easy access from shore without the need for a boat, yet it is also sufficiently far away from shore (and deep) to minimize visitation rates from recreational divers who could interfere with the experiment. Third, it represents a study system at the early stages of sun coral colonization, when removal measures are likely to be most effective.

Table 3.1. List of submerged marine artificial structures deployed along the south and west coasts of Barbados potentially hosting sun coral populations, showing the type of structure (pier, breakwater, reef ball, mooring, shipwreck), approximate depth, GPS coordinates (Latitude and Longitude), and whether sun corals were ultimately found between Oct 2021 and Feb 2022. NA - data not available. Sites are listed in geographical order from north to south, and their locations are visualized in Figure 3.2.

ID	Site	Coast	Type	Depth (m)	Latitude	Longitude	Presence
1	Cement factory pier	West	Pier	3	N 13º 17' 3.21"	W 059º 39' 9.26"	Yes
2	Port Ferdinand breakwater	West	Breakwater	NA	N 13º 17' 54.12"	W 059º 39' 39.69"	No
3	Port St. Charles breakwater	West	Breakwater	NA	N 13º 15' 49.52"	W 059º 39' 41.46"	No
4	Pamir	West	Wreck	12	N 13º 15' 27.83"	W 059º 38' 48.34"	Yes
5	Pamir submarine	West	Wreck	13	N 13º 15' 27.90"	W 059º 38' 49.98"	Yes
6	Weston breakwater	West	Breakwater	NA	N 13º 13' 00.43"	W 059º 38' 32.13"	No
7	Holetwon Barge	West	Wreck	NA	N 13º 10' 46.22"	W 059º 38' 28.95"	No
8	Sandy Lane breakwater	West	Breakwater	NA	N 13º 10' 33.42"	W 059º 38' 22.29"	No
9	SS Stavronikita	West	Wreck	NA	N 13º 08' 39.13"	W 059º 38' 34.96"	Yes
10	Lord Combermere	West	Wreck	12	N 13º 07' 27.36"	W 059º 38' 46.20"	Yes
11	Lord Wiloughby	West	Wreck	30	N 13º 07' 06.50"	W 059º 38' 25.93"	Yes
12	Pile Bay tanker buoys	West	Mooring	NA	N 13º 07' 30.25"	W 059º 38' 08.94"	NA
13	Coast Guard breakwater	West	Breakwater	NA	N 13º 06' 40.75 "	W 059º 37' 54.40"	NA
14	Shallow Draught	West	Pier	NA	N 13º 06' 31.96"	W 059º 37' 35.55"	No
15	Harbour Wall	South	Pier	NA	N 13º 06' 14.50"	W 059º 37' 57.15"	NA
16	Old Esso Jetty	South	Pier	4	N 13º 05' 58.24"	W 059º 37' 46.01"	Yes
17	Bridgetown Fishing Complex breakwater	South	Breakwater	2	N 13º 05' 45.20"	W 059º 37' 17.55"	Yes
18	Brianna H	South	Wreck	16	N 13º 05' 24.54"	W 059º 37' 27.84"	Yes
19	Boatyard Jetty	South	Pier	NA	N 13º 05' 31.83"	W 059º 36' 47.37"	No
20	Ce-Trec	South	Wreck	NA	N 13º 05' 12.38"	W 059º 36' 42.78"	No
21	Zoom (sailboat wreck)	South	Wreck	16	N 13º 05' 13.35"	W 059º 36' 43.58"	Yes
22	Ellion	South	Wreck	15	N 13º 05' 11.69"	W 059º 36' 43.58"	Yes
23	Cornwallis	South	Wreck	5	N 13º 05' 10.76"	W 059º 36' 42.18"	Yes

24	Bajan Queen	South	Wreck	3	N 13º 05' 10.72"	W 059º 36' 43.49"	Yes
25	Barge	South	Wreck	NA	N 13º 05' 10.11"	W 059º 36' 41.60"	No
26	Berwyn	South	Wreck	5	N 13º 05' 08.93"	W 059º 36' 42.00"	Yes
27	Trident	South	Wreck	17	N 13º 05' 7.59"	W 059º 36' 49.69"	Yes
28	Radisson Pier	South	Pier	2	N 13º 04' 58.88"	W 059º 36' 36.55"	Yes
29	Engineer Pier	South	Pier	NA	N 13º 04' 51.75"	W 059º 36' 45.11"	No
30	Hilton breakwater	South	Breakwater	NA	N 13º 04' 39.28"	W 059º 36' 42.41"	No
31	Friar's Craig	South	Wreck	15	N 13º 04' 20.73"	W 059º 36' 21.37"	Yes
32	Asta reef balls	South	Reef ball	NA	N 13º 04' 22.65"	W 059º 36' 19.83"	No
33	Accra breakwater	South	Breakwater	NA	N 13º 04' 18.02"	W 059º 35' 18.02"	No
34	Sandals breakwater	South	Breakwater	NA	N 13º 04' 49.43"	W 059º 33' 42.26"	No
35	Welches reef balls	South	Reef ball	NA	N 13º 03' 47.08"	W 059º 33' 00.77"	No
36	Oistins Ice Pier	South	Pier	NA	N 13º 03' 45.99"	W 059º 32' 40.43"	No
37	Oistins Jetty	South	Pier	NA	N 13º 03' 44.74"	W 059º 32' 38.36"	No
38	Oistins tanker buoys	South	Mooring	1	N 13º 03' 28.83"	W 059º 32' 35.78"	Yes

Figure 3.1. Map of Barbados showing the location of the submerged marine artificial structures deployed along the south and west coast of Barbados that were included in the list of potential sites hosting sun corals to be surveyed between Oct 2021 and Feb 2022. Orange pins indicate sites where sun corals were present during the survey period; green pins indicate sites where sun corals were absent; dark violet pins indicate sites that were not surveyed. W - shipwreck; B-breakwater; P-pier; B-reef ball.

The experiment involved removing all sun corals (and co-occurring benthic organisms) from small experimental plots located on the shaded side of the Trident shipwreck and then comparing changes in sun coral abundance over time in these removal plots with sun coral abundance on adjacent controls plots (i.e. small plots where nothing had been removed). The shaded side of the Trident was chosen because preliminary observations indicated that this is where most sun corals could be found. To do so, six pairs of small (0.25 x 0.25 m^2) experimental quadrats were demarcated on the vertical hull of the Trident. Each pair represented an experimental block consisting of one control quadrat and one removal quadrat located 0.5 to 1 m apart. These pairs were deployed at regular intervals at the same depth along the side to account for potential depth effects, with approximately 2 m separating the closest pairs of quadrats (Figure 3.4).

The removal treatment consisted in scraping all living material (sun corals and other attached benthic organisms) from inside each quadrat using a scraper, whereas the control treatment involved no modification of quadrats whatsoever (Figure 3.4). Each quadrat was delineated by carefully scraping all living material from its perimeter and maintaining the perimeter by scraping and cleaning each visit.

Data on sun coral abundance was obtained from each quadrat by carefully inspecting each quadrat and counting the number of polyps found. These sun coral abundance surveys were carried out immediately prior to the removal of sun corals (from the designated quadrats) and subsequently every two weeks for a three-month period. The experiment started on Nov 21, 2021 and ended on Feb 13, 2022.

Figure 3.2. Picture of the Trident shipwreck in Carlisle Bay, where the removal experiment was conducted between Nov 2021 and Feb 2022. *Photo credit: R. Bourne*

Figure 3.4. Pictures of a control quadrat (left panel) and a removal quadrat a few weeks after scraping its inside surface clean (right panel). Note the conspicuous sun coral polyps in the control quadrat as well as the delineated white perimeter of each 25 x 25 cm^2 quadrat. *Photo credits: R. Bourne.*

3.3 Distribution of sun coral on artificial structures: South and West coast of Barbados

Thirty-eight marine artificial structures deployed along the south and west coasts of Barbados, from Oistins to the Cement factory pier, were identified as potential habitat for the sun coral. Of these, 35 (92%) were surveyed for sun corals (Table 3.1). Most (51%) of the surveyed structures had sun corals present on them (Table 3.1; Figure 3.1). Sun corals were found on structures deployed on both the west and south coasts (Figure 3.1). Interestingly, overall, there was a significant association between the nature of the structure and the likelihood of finding sun corals on them, whereby metallic structures (wrecks and mooring buoys) were much more likely to host sun corals than non-metallic ones (piers, reef balls and breakwaters) (Chi-square test: df = 1, Chi-square = 12.655, $p<0.001$; Figure 3.5).

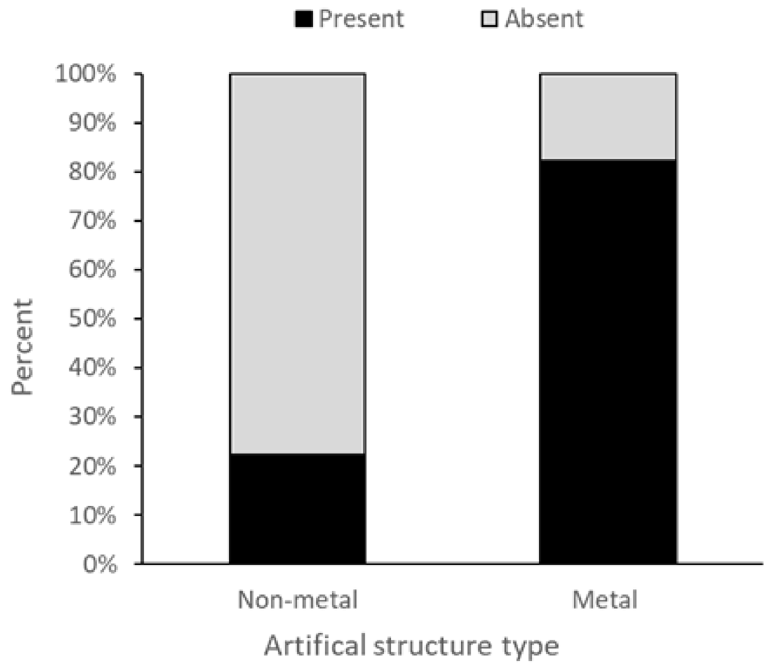

Figure 3.5. Percent of submerged marine artificial structures along the south and west coast of Barbados where sun corals were present between Oct 2021 and Feb 2022, while distinguishing between metal (shipwrecks and moorings) and non-metal (reef balls, breakwaters, and piers) structures.

Conclusion: ***The sun coral can now be found on most of the submerged marine artificial structures along the south and west coast of Barbados. Structures made up of metal (mooring buoys and shipwrecks) were more likely to host sun corals than those not made out of metal (piers, reef balls and breakwaters), pointing to an area for further research.***

3.4 Sun coral removal and population recovery

We compared sun coral abundance between the control quadrats and the removal ones at three key time points: (1) immediately before the complete removal of sun corals and other benthic organisms by scraping in the removal treatment (Nov 21, 2021), (2) two weeks after the removal was conducted (Dec 05, 2021), and (3) at the end of the time series, which corresponded to three months after the removal (Feb 13, 2022). There was no difference in sun coral polyp abundance between the control and removal treatment before the removal was conducted (Nov 21; Mann-Whitney test: $U=18$, $n=12$, $p<0.001$), with an overall average of about 35 polyps per quadrat; this expected lack of difference thus confirmed the appropriateness of the control quadrats. We found a highly significant difference in polyp abundance between the two treatments two weeks after the removal (Dec 05; Mann-Whitney test: $U=36$, $n=12$, $p=0.002$), as evidenced by the zero abundance of sun corals on the removal quadrats (Figure 3.6). There was some evidence of sun coral recovery six weeks after removal (Jan 03; Figure 3.6), with an increase in polyp abundance to an average of 4-5 polyps per plot, about 13% of pre-removal levels. Sun coral abundance remained stable thereafter to the end of the study in Jan 16 (Figure 3.6) and this post-removal abundance remained significantly lower than the control one (Mann-Whitney test: $U=35$ $n=12$, $p=0.004$).

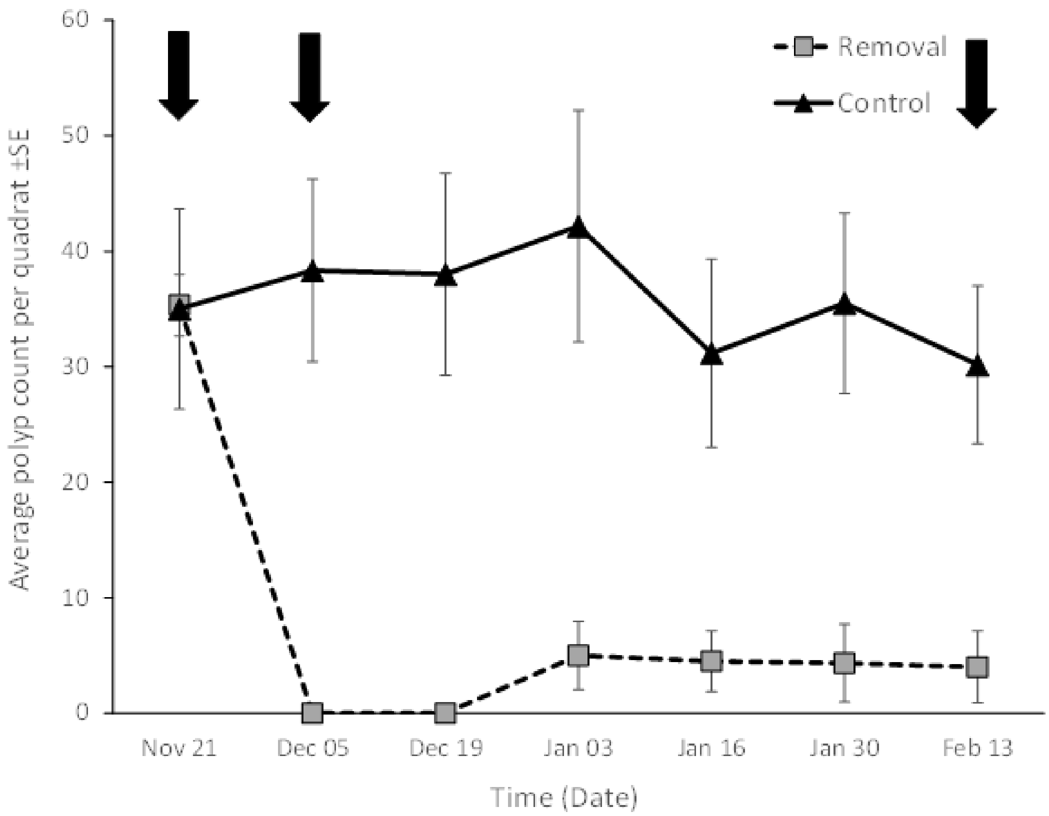

Figure 3.6. Average sun coral polyp abundance for the control and removal treatments at the Trident shipwreck over biweekly surveys between Nov 21, 2021 and February 13, 2022. N=6 (25 x 25 cm^2 quadrats for each treatment). Black arrows represent time periods when control and removal treatments were compared statistically.

Conclusion: ***Complete removal of sun corals by scraping is an effective strategy to maintain the population of sun coral at very low levels, at least over three-month periods and at sites where sun corals are yet to become the dominant benthic organism. However, at sites where sun corals are yet to become dominant, such removal will likely also negatively affect co-occurring native benthic organisms.***

3.5 Sun coral management strategy

This study has shown that sun corals can now be found on most marine artificial structures such as shipwrecks, moorings and piers, along the south and west coasts of Barbados. Although rigorously quantifying sun coral abundance was beyond the scope of this study, it was evident from the site surveys that sun corals had the potential to reach relatively high densities on some of these artificial structures (e.g. Arawak cement factory pier system). However, to date, the distribution of sun corals in Barbados seems to be strongly restricted to artificial structures with no evidence of range expansion into natural reefs. For example, recent coral surveys (in 2022) on 47 permanent natural reef survey sites along the south and west coasts of Barbados, including fringing, patch and bank reefs, did not record any sun coral (BRSP, unpublished data). Overall, this suggests that sun corals in Barbados are currently naturally restricted to artificial structures and marginal reef habitat, and that such structures now represent the main local sources of sun coral propagules. That said, the fact sun corals have quickly colonized most artificial structures and appear to be increasing in abundance on such structures indicates that they should be considered a species of concern and be subject to further monitoring. In particular, it would be important to establish quantitative baselines of their current abundance so as to help rigorously assess their population growth and range expansion. It will also be important to conduct further taxonomic work to confirm whether both *T. coccinea* and *T. tagusensis* are both present in Barbados. It would also be beneficial to document the interactions between sun corals and other benthic organisms on the colonized artificial structures to assess their potential impact on local biodiversity.

The fact that they are currently restricted to artificial structures, which likely account for most their reproductive stock, supports that conducting colony removals by scraping will be a cost-effective means of controlling the total population (Creed et al. 2021). However, careful consideration should be given to the other benthic organisms that would be affected by such removals. Overall, we recommend (1) further monitoring of artificial structures to establish quantitative abundance baselines, and assessing whether both *T. coccinea* and *T. tagusensis* are present; (2) research into how sun corals interact with other benthic organisms (e.g. encrusting sponges) typically found on the same structures; (3) promoting a public awareness campaign to sensitize stakeholders on the possible threat of sun corals; and (4) integrate key stakeholders (e.g. divers) into an island-scale monitoring, particularly in relation to early detection of sun corals potentially expanding their current range into natural reefs (Machado et al 2021).

4. SEAGRASS (*HALOPHILA STIPULACEA*)

4.1 Background

Seagrasses, including the diminutive *Halophila* genus, are among the most productive plant communities in the world. They provide critical ecosystem services, including sediment stabilization, water column filtration, 'blue' carbon burial and refuge, feeding and nursery habitat for a large number of juvenile fish, shrimp and crabs, including important commercial species such as spiny lobster, queen conch and sea eggs. Furthermore, charismatic grazing animals, including the endangered manatees and green sea turtles rely directly on seagrasses for much of their nutrition, and seagrass communities provide the base of food chains supporting many large pelagic species such as sharks and rays, and others of significant commercial importance such as jacks, barracudas and mackerels.

The invasive seagrass, *Halophila stipulacea,* originating from the Indian Ocean and Red Sea and also now present in the Mediterranean Sea, has been spreading rapidly through the eastern Caribbean islands since 2002, taking over native seagrass beds, especially in shallow, nutrient enriched nearshore environments (Willette et al. 2014). It is not yet known to have arrived in Barbados, although this is based only on lack of anecdotal sightings. As such this study aimed to provide a baseline of *H. stipulacea* presence/abundance in Barbados as of 2022.

4.2 Methodology

The invasive *Halophila stipulacea* seagrass is relatively easy to identify and one member of the team has experience with surveys for this species throughout the Grenadine islands. As such we undertook simple roving visual surveys by free-diving or using SCUBA. These reconnaissance surveys were conducted in Barbados during the summer of 2022 in shallow water habitats (down to a depth of 12 m) where native seagrasses are known to occur (Figure 4.1). This included repeated surveys of the extensive (but low density) beds of *Halodule wrightii* and *Halophila decipiens* throughout Carlisle Bay, the site most likely to receive this invasive species since it is believed to be spread by the anchors of visiting yachts (Willette et al. 2014). We also undertook single surveys of the dwindling mixed species beds of *Thalassia testudinum* and *Syringodium filiforme* in Worthing Lagoon and other scattered locations along the southwest coast and several fringing reef flats on the west coast. Single surveys were also undertaken on the sparse *T. testudinum* and *S. filiforme* populations in Consett Bay and Bath on the east coast under conditions of relatively high wave energy and poor visibility as a result of the frequent sargassum brown tide events at these exposed sites.

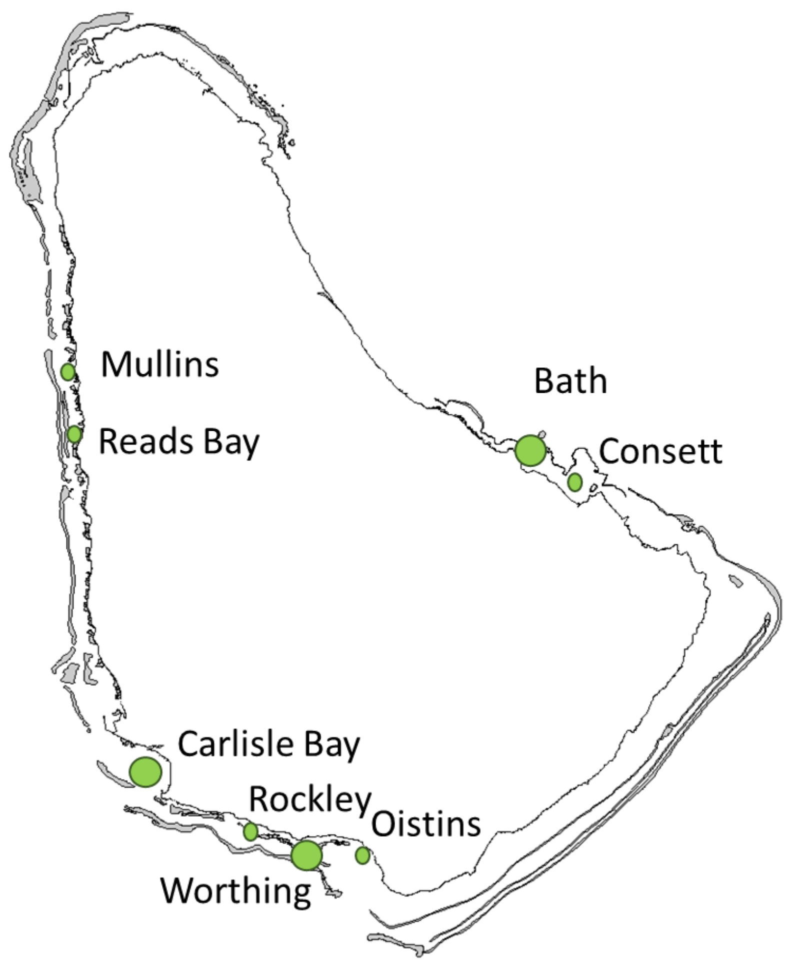

Figure 4.1. Map of Barbados showing seagrass sites surveyed for presence/absence of the invasive *Halophila stipulacea*. Size of green circle indicates the relative extent of native seagrass at site.

4.3 Distribution of *Halophila stipulacea*

No invasive seagrass was found at any of the sites surveyed, confirming anecdotal information that this species has not arrived in Barbados.

Conclusion: ***No evidence of the invasive Halophila stipulacea was found in seagrass sites around Barbados. Although Carlisle Bay, which hosts native seagrasses, receives a fairly high number of visiting yachts during the winter season, the vast majority are anchoring here after a long Atlantic crossing during which time any seagrass plants picked up at their previous anchorage are extremely unlikely to survive. The vast majority of these yachts do not return to Barbados after cruising the eastern Caribbean Islands where this invasive species is now well established.***

4.4 *Halophila stipulacea* Management Strategy

Halophila stipulacea is an invasive seagrass currently spreading through the Caribbean, and since seagrasses are foundation species, this invasion has the potential to be particularly far-reaching through alteration of the associated communities and food webs. Much remains unknown about the impacts of this species, but research to date including long-term monitoring by Muthukrishan et al. (2020) in the US Virgin Islands suggests that *H. stipulacea* is likely to displace some native seagrasses, notably *Syringodium filiforme* and *Halodule wrightii*, but not necessarily some others such as the more robust Caribbean climax species, *T. testudinum*. They also report that the invasive seagrass is less nutritious for consumers than native species, and hosts invertebrate assemblages that are distinct from those in native seagrass beds, although they remain equally diverse and have similar biomass. This has implications for altering food webs especially at the higher trophic levels. However, they noted that the invasive seagrass can also act as a foundation species, providing habitat and trophic support to other species, although it is not entirely equivalent to native species in most of the properties so far quantified (Muthukrishan et al. 2020). The invasive seagrass can also provide valuable supporting ecosystem services such clarifying coastal water, sequestering blue carbon, increasing substrate stability, and helping to protect shorelines from erosion, even if not equivalent to native species, although they remain untested.

In the absence of any known *H. stipulacea* in Barbados, no management action to prevent the spread of the invasive seagrass is necessary at this time. However, it is recommended that divers (dive shops, recreational snorkelers/divers and nearshore fishers) be appraised of the potential arrival of this species through the production of an informative infographic and a reporting mechanism be established (e.g. WhatsApp group, Instagram or Facebook page) monitored by the CZMU. In the event that this species does arrive and become established in Barbados, a long-term monitoring programme is recommended to track spatial expansion, interaction with the existing sparse native seagrass species and provision of ecosystem services.

5. Management of Marine Invasive Species: Lessons Learned

How and when to manage particular species is an essential, yet difficult, question to answer, and can be contentious. There remains debate about the ubiquity of harmful effects of invasive species in general and it can be both scientifically and ethically challenging to determine when an invasive species is similar or beneficial enough to take an accommodative management perspective.

Considering that management of established invasions is likely to involve extensive effort and associated costs, there is need for economic cost-benefit analyses that evaluate the actual costs of management action in relation to the true costs of ecosystem consequences (good and bad) of an alien species invasion (Muthukrishnan et al. 2020). However, this type of information is often not available, especially when dealing with complex systems and foundation species with the inevitable large uncertainties.

Ideally then, management decisions should be based on the specific impacts of invading species in the local context including both their effects on native competitors and how they may or may not play similar roles in broader ecosystem functioning (see Muthukrishnan et al. 2020 and references therein). These authors suggest that the best strategy available is for explicit evaluation of risks and impacts, considering the potential for both positive and negative effects, so that the best available predictions can be made on which stakeholders and managers can base informed decisions (Strayer 2012). In this study, we have assessed the risks and impacts for lionfish in Barbados and have provided baseline data for two other alien species to inform management policy.

In the case of the lionfish invasion, early research presented a 'worst-case-scenario' with far-reaching negative impacts on reef communities (i.e. decreasing biodiversity and resilience) due to increased predation of fish and invertebrate recruits and competition for food with native species (e.g. Albins and Hixon 2008, Green and Côté 2009, Green et al. 2012, Albins 2013). The findings of the current study provided no such evidence of lionfish effects on Barbadian fish assemblages. In fact, the numbers of lionfish are quite low compared to many other locations across the Caribbean. Fishing yields for trap and spear fishers did not differ between the pre- and post-invasion periods. Moreover, lionfish now make up nearly a quarter of the fish caught by spear fishers, thus potentially helping to relieve some of the spearfishing pressure on other reef fish groups, including parrotfishes. Spearfishing, by both commercial and recreational divers, seems to play a critical role in helping to control the lionfish population at shallow to moderately deep sites. The aforementioned fishing pressure, in addition to the consumption of lionfish in Barbados, has been effective at keeping the lionfish population under control on the shallow reefs, therefore minimizing the impact of lionfish on our reef fishes. One potential benefit of the lionfish's arrival is the provision of economic and leisure opportunities to commercial and recreational spear fishers alike.

The recent identification of sun corals as an alien species of concern in Barbados highlights the need for the continued education of the general public on invasives, particularly of the stakeholders more likely to come into contact with marine alien species (divers; fishers). Anecdotal evidence from shipwreck photos indicate that sun corals were already in Barbados way before they were first recognized as non-native in 2018. Their potential threat had thus gone unnoticed until now. Early detection is critical to help effectively control the spread of potential invasives, and this can be better achieved when people can recognize alien species (Machado et al 2021).

In the case of seagrasses which are known to be foundational species, invaders may displace native species but provide similar ecosystem functions. In fact, in Barbados, where native seagrasses have undergone significant decline in the last few decades, the establishment of invasive seagrasses may replace valuable ecosystem functions that have been lost. As such we recommend only that stakeholder be appraised of the potential arrival of *H. stipulacea*, and that expansion and impacts are monitored should an invasion occur.

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7. APPENDICES

Appendix 1 Survey methodology utilised for the assessment of native reef fish and lionfish at each of the 10 coral reef sites

- A minimum of 10 transects were surveyed on each reef site.
- Where the reef was large enough all transects were set 30 m apart in all directions.
- On west coast fringing reefs transects were set on the reef near the seaward edge and oriented to run across the spur and grooves (NOT along a spur), and set in a line end to end, at least 30 m apart.
- For small fringing reefs transects were laid parallel or perpendicular to one another to fit onto the reef to maximise the surface area while obtaining the maximum possible distance between transects.
- On narrow bank reefs, the transects ran parallel to one another and perpendicular to shore, separated by 10-15 m. On these banks, the transects were along the top of the reef crest.
- Each transect was 30 m x 2 m for large mobile fish (LF) and 30 m x 1 m for slow moving, attached species, territorial slow-moving fish (SF).
- Each transect was surveyed by making two passes, the first to record slow moving/attached species (SF) and the second to record mobile species (LF).
- Mobile fish (LF) included parrotfish, surgeonfish, adult yellowtail damselfish and lionfish.
- Slow moving fish (SF) included bicolor damselfish, other damselfishes (EXCEPT adult yellowtail damselfish), bluehead wrasse, yellowhead wrasse, clown wrasse, slippery dick and other wrasses and diadema urchins. All other ‘cryptic’ species were ignored.
- Fish were allowed to settle after rolling the tape.
- On pass 1, mobile fish (LF) were recorded - each diver counted, identified and sized (to the nearest cm) all parrotfish, surgeonfish, adult yellowtail damselfish and lionfish within a belt of 30m x 2 m. Parrotfish in the terminal phase were indicated using TP.
- Fish were once again allowed to settle after pass 1.
- On pass 2, slow moving fish (SF) and diadema urchins were recorded - each diver counted, identified and sized (using three size classes - S1: <5 cm, S2: 5-10 cm, S3: >10 cm) wrasses (bluehead wrasse, yellowhead wrasse, clown wrasse, slippery dick and other wrasses) and damselfishes (bicolor damselfish and other damselfishes EXCEPT adult yellowtail damselfish) within a belt of 30 m x 1 m.
- The tape was rolled up after the data were collected and the diver navigated to the start of the second transect. Each diver collected data from a minimum of two transects.
- Data were collected onto pre-marked underwater slates, which were photocopied and archived on return to shore. Data were then entered into a standard template in an excel spreadsheet.

Appendix 2. GPS coordinates for fish survey sites

Site	Transect	Co-ordinates		Site	Transect	Co-ordinates	
Accra Bank	1	N13 04 02.8	W59 35 40.0	Holetown Bank	1	N13 11 00.3	W59 38 45.7
Accra Bank	2	N13 04 02.8	W59 35 41.5	Holetown Bank	2	N13 10 59.8	W59 38 45.7
Accra Bank	3	N13 04 03.2	W59 35 40.0	Holetown Bank	3	N13 11 00.3	W59 38 45.2
Accra Bank	4	N13 04 03.2	W59 35 41.5	Holetown Bank	4	N13 10 59.8	W59 38 45.3
Accra Bank	5	N13 04 03.5	W59 35 40.0	Holetown Bank	5	N13 11 00.2	W59 38 44.8
Accra Bank	6	N13 04 03.5	W59 35 41.5	Holetown Bank	6	N13 10 59.8	W59 38 44.8
Accra Bank	7	N13 04 03.8	W59 35 40.0	Holetown Bank	7	N13 11 00.2	W59 38 44.4
Accra Bank	8	N13 04 03.8	W59 35 41.5	Holetown Bank	8	N13 10 59.8	W59 38 44.4
Accra Bank	9	N13 04 04.2	W59 35 40.0	Holetown Bank	9	N13 11 00.2	W59 38 44.0
Accra Bank	10	N13 04 04.1	W59 35 41.5	Holetown Bank	10	N13 10 59.8	W59 38 44.0
Southern Palms	1	N13 03 51.2	W59 34 13.7	South Bellairs	1	N13 11 25.9	W59 38 33.6
Southern Palms	2	N13 03 51.4	W59 34 12.4	South Bellairs	2	N13 11 25.9	W59 38 33.6
Southern Palms	3	N13 03 50.4	W59 34 13.7	South Bellairs	3	N13 11 24.6	W59 38 32.8
Southern Palms	4	N13 03 49.1	W59 34 13.1	South Bellairs	4	N13 11 24.6	W59 38 32.6
Southern Palms	5	N13 03 48.7	W59 34 11.9	South Bellairs	5	N13 11 27.2	W59 38 33.9
Southern Palms	6	N13 03 49.1	W59 34 10.8	South Bellairs	6	N13 11 27.2	W59 38 32.9
Southern Palms	7	N13 03 48.2	W59 34 10.0	South Bellairs	7	N13 11 28.3	W59 38 33.4
Southern Palms	8	N13 03 47.8	W59 34 09.0	South Bellairs	8	N13 11 28.3	W59 38 32.4
Southern Palms	9	N13 03 47.3	W59 34 08.1	South Bellairs	9	N13 11 28.9	W59 38 32.3
Southern Palms	10	N13 03 46.7	W59 34 07.1	South Bellairs	10	N13 11 28.9	W59 38 31.3
Josef's	1	N13 03 51.9	W59 34 38.1	Greensleeves	1	N13 13 37.0	W59 38 42.7
Josef's	2	N13 03 51.6	W59 34 36.9	Greensleeves	2	N13 13 36.0	W59 38 42.0
Josef's	3	N13 03 51.5	W59 34 35.9	Greensleeves	3	N13 13 36.7	W59 38 40.7
Josef's	4	N13 03 51.5	W59 34 34.8	Greensleeves	4	N13 13 36.7	W59 38 40.7
Josef's	5	N13 03 51.5	W59 34 34.0	Greensleeves	5	N13 13 36.2	W59 38 39.1
Josef's	6	N13 03 51.7	W59 34 32.9	Greensleeves	6	N13 13 36.6	W59 38 38.0
Josef's	7	N13 03 51.4	W59 34 31.8	Greensleeves	7	N13 13 34.6	W59 38 40.4
Josef's	8	N13 03 52.0	W59 34 30.5	Greensleeves	8	N13 13 35.1	W59 38 39.3

Josef's	9	N13 03 53.1	W59 34 35.6	Greensleeves	9	N13 13 35.8	W59 38 41.7
Josef's	10	N13 03 53.0	W59 34 33.7	Greensleeves	10	N13 13 34.5	W59 38 40.9
Coconut Court	1	N13 04 24.0	W59 36 17.6	Speightstown Bank	1	N13 14 53.6	W59 38 08.4
Coconut Court	2	N13 04 24.0	W59 36 16.6	Speightstown Bank	2	N13 14 53.1	W59 38 08.4
Coconut Court	3	N13 04 24.0	W59 36 15.6	Speightstown Bank	3	N13 14 53.6	W59 39 08.1
Coconut Court	4	N13 04 24.0	W59 36 14.6	Speightstown Bank	4	N13 14 53.1	W59 39 08.1
Coconut Court	5	N13 04 24.0	W59 36 13.6	Speightstown Bank	5	N13 14 53.6	W59 39 07.8
Coconut Court	6	N13 04 24.1	W59 36 12.7	Speightstown Bank	6	N13 14 53.1	W59 39 07.8
Coconut Court	7	N13 04 24.2	W59 36 11.6	Speightstown Bank	7	N13 14 53.6	W59 39 07.5
Coconut Court	8	N13 04 24.2	W59 36 10.6	Speightstown Bank	8	N13 14 53.1	W59 39 07.4
Coconut Court	9	N13 04 24.3	W59 36 09.6	Speightstown Bank	9	N13 14 53.6	W59 39 07.1
Coconut Court	10	N13 04 24.4	W59 36 08.6	Speightstown Bank	10	N13 14 53.1	W59 39 07.1
Batts Rock	1	N13 08 08.2	W59 38 19.8	Six Men's	1	N13 16 20.1	W59 38 51.3
Batts Rock	2	N13 08 08.5	W59 38 18.8	Six Men's	2	N13 16 18.7	W59 38 51.3
Batts Rock	3	N13 08 09.2	W59 38 17.6	Six Men's	3	N13 16 20.4	W59 38 50.7
Batts Rock	4	N13 08 10.2	W59 38 17.7	Six Men's	4	N13 16 19.9	W59 38 50.7
Batts Rock	5	N13 08 09.9	W59 38 16.6	Six Men's	5	N13 16 19.4	W59 38 50.6
Batts Rock	6	N13 08 08.9	W59 38 15.7	Six Men's	6	N13 16 18.9	W59 38 50.6
Batts Rock	7	N13 08 07.4	W59 38 17.7	Six Men's	7	N13 16 18.3	W59 38 50.8
Batts Rock	8	N13 08 06.7	W59 38 17.0	Six Men's	8	N13 16 17.8	W59 38 50.8`
Batts Rock	9	N13 08 08.6	W59 38 18.3	Six Men's	9	N13 16 20.1	W59 38 49.3
Batts Rock	10	N13 08 07.2	W59 38 14.9	Six Men's	10	N13 16 18.6	W59 38 49.3

Appendix 3. Survey instrument used for key informant recreational diver interviews

Name of key informant(s): **DATE:**

Contact info:

Name of diveshop:

Diveshop location:

Is your diveshop involved in any way in lionfish culling? Yes No

If yes, explain how (e.g. taking guests to hunt lionfish, staff hunting, derby participation, etc)

Has your diveshop or any of its staff been involved in research on lionfish? Yes No

If yes, please explain how and, if possible, provide contact info of researchers involved.

Diveshop sites (Visitation rates and lionfish abundance – before COVID)

		Visitation rates					Lionfish abundance			
Site #	Site name	Daily to several times a week	Once a week to a couple of times a month	From once a month to a few times a year	A couple of times a year to never	Number of "culling" divers per dive	Relatively high (most divers catch or see several lionfish)	Medium (only a few divers catch or see several lionfish)	Relatively low (most divers do not catch or see any lionfish)	Don't know
1										
2										
3										
4										
5										
6										
7										
8										
9										
10										
11										
12										
13										
14										
15										

Diveshop sites (Visitation rates and lionfish abundance – before COVID)

Site #	Site name	Visitation rates				Number of "culling" divers per dive	Lionfish abundance			
		Daily to several times a week	Once a week to a couple of times a month	From once a month to a few times a year	A couple of times a year to never		Relatively high (most divers catch or see several lionfish)	Medium (only a few divers catch or see several lionfish)	Relatively low (most divers do not catch or see any lionfish)	Don't know

Project target sites (before COVID)

Site #	Site name	Visitation rates				Number of "culling" divers per dive	Lionfish abundance			
		Daily to several times a week	Once a week to a couple of times a month	From once a month to a few times a year	A couple of times a year to never		Relatively high (most divers catch or see several lionfish)	Medium (only a few divers catch or see several lionfish)	Relatively low (most divers do not catch or see any lionfish)	Don't know
1	South Bellairs									
2	Sixmens reef									
3	Holetown Bank									
4	Speightstown Bank									
5	Southern Palms									
6	Coconut Court									
7	Accra Bank									
8	Josef's reef									
9	Batts Rock									
10	Greensleeves									

Project target sites (After COVID)

Site #	Site name	Visitation rates: Daily to several times a week	Visitation rates: Once a week to a couple of times a month	Visitation rates: From once a month to a few times a year	Visitation rates: A couple of times a year to never	Number of "culling" divers per dive	Lionfish abundance: Relatively high (most divers catch or see several lionfish)	Lionfish abundance: Medium (only a few divers catch or see several lionfish)	Lionfish abundance: Relatively low (most divers do not catch or see any lionfish)	Lionfish abundance: Don't know
1	South Bellairs									
2	Sixmens reef									
3	Holetown Bank									
4	Speightstown Bank									
5	Southern Palms									
6	Coconut Court									
7	Accra Bank									
8	Josef's reef									
9	Batts Rock									
10	Greensleeves									

Map of the project target sites

APPENDIX - Site list and map

#	Site	Type	Latitude	Longitude
1	Maycocks	Reef	N 13^{0} 17' 32.85"	W 059^{0} 39' 47.53"
2	Maycocks Inner	Reef	N 13^{0} 17' 21.90"	W 059^{0} 39' 27.66"
3	Bright Ledge	Reef	N 13^{0} 15' 54.91"	W 059^{0} 39' 09.47"
4	Pamir	Wreck	N 13^{0} 15' 27.90"	W 059^{0} 38' 49.98"
5	Great Ledge	Reef	N 13^{0} 14' 45.31"	W 059^{0} 39' 09.20"
6	Heymans	Reef	N 13^{0} 14' 13.25"	W 059^{0} 39' 05.22"
7	Spawnee	Reef	N 13^{0} 13' 38.39"	W 059^{0} 39' 08.46"
8	Weston	Reef	N 13^{0} 12' 53.19"	W 059^{0} 38' 53.72"
9	Boom	Reef	N 13^{0} 12' 24.18"	W 059^{0} 38' 53.59"
10	Fisherman's	Reef	N 13^{0} 11' 00.97"	W 059^{0} 38' 46.75"
11	Dottins	Reef	N 13^{0} 10' 32.85"	W 059^{0} 38' 51.53"
12	Sandy Lane	Reef	N 13^{0} 10' 32.85"	W 059^{0} 38' 47.55"
13	Small Wreck	Reef	N 13^{0} 09' 52.88"	W 059^{0} 38' 22.63"
14	Johnson's	Reef	N 13^{0} 09' 42.60"	W 059^{0} 38' 38.34"
15	Victors	Reef	N 13^{0} 08' 54.60"	W 059^{0} 38' 39.22"
16	Stavronikita	Wreck	N 13^{0} 08' 39.13"	W 059^{0} 38' 34.96"
17	Lord Combermere	Wreck	N 13^{0} 08' 06.50"	W 059^{0} 38' 25.93"
18	Atlantis Bank	Reef	N 13^{0} 07' 30.76"	W 059^{0} 38' 45.90"
19	Clarks/Silver Bank	Reef	N 13^{0} 07' 17.22"	W 059^{0} 38' 40.43"
20	C'Bay North Deep		N 13^{0} 05' 15.85"	W 059^{0} 36' 49.20"
21	C'Bay North Shallow		N 13^{0} 05' 13.27"	W 059^{0} 36' 39.35"
22	C'Bay South Deep		N 13^{0} 05' 06.83"	W 059^{0} 36' 47.47"
23	C'Bay South Shallow		N 13^{0} 05' 07.19"	W 059^{0} 36' 38.83"
24	Old fort	Reef	N 13^{0} 04' 39.32"	W 059^{0} 36' 55.16"
25	Friar's Craig	Wreck	N 13^{0} 04' 20.73"	W 059^{0} 36' 21.37"
26	Asta west	Reef	N 13^{0} 04' 23.79"	W 059^{0} 36' 13.55"
27	Asta east	Reef	N 13^{0} 04' 24.08"	W 059^{0} 36' 08.47"
28	Pieces of Eight	Reef	N 13^{0} 04' 18.38"	W 059^{0} 35 47.46"
29	Accra	Reef	N 13^{0} 04' 17.10"	W 059^{0} 35 29.00"
30	Castle Bank	Reef	N 13^{0} 04' 10.19"	W 059^{0} 36' 35.55"
31	Fork Reef	Reef	N 13^{0} 04' 08.74"	W 059^{0} 36' 18.35"
32	Charleynes	Reef	N 13^{0} 04' 09.02"	W 059^{0} 36' 08.70"
33	Caribee	Reef	N 13^{0} 04' 07.92"	W 059^{0} 36' 04.38"
34	Brown Shallows	Reef	N 13^{0} 03' 58.62"	W 059^{0} 35' 32.34"
35	Moo	Reef	N 13^{0} 03' 54.22"	W 059^{0} 34' 52.01"
36	Sandy Beach West (boot)	Reef	N 13^{0} 03' 53.88"	W 059^{0} 34' 48.47"
37	Sandy Beach East	Reef	N 13^{0} 03' 52.98"	W 059^{0} 34' 43.18"
38	Close Encounters	Reef	N 13^{0} 03' 50.40"	W 059^{0} 34' 15.20"
39	Muff	Reef	N 13^{0} 03' 37.03"	W 059^{0} 35' 04.10"
40	Needle	Reef	N 13^{0} 03' 31.92"	W 059^{0} 34' 12.00"
41	Mount Charlie	Reef	N 13^{0} 03' 19.37"	W 059^{0} 34' 10.28"
42	Black Jack Shallowes	Reef	N 13^{0} 03' 28.74"	W 059^{0} 34' 04.32"
43	Welcome Inn	Reef	N 13^{0} 03' 35.52"	W 059^{0} 33' 25.74"
44	lord Wilouby			
45	Brianna H	wreck	N 13^{0}05.409'	W 059^{0}37.464'
46	Trident			
	Shark Hole			
	Lighthouse drop off			
	Jetty North			

#	Site	Type	Latitude	Longitude
	Jetty South			
	Cement Plant			
	barge Sandy crest marine park			
	Baracuda Junction			
	Jordon shallows			
	Lord Wiloughby			
	Brianna H			
	Carlisle Bay			
	Trident			
	The Dredge /Butt crack			
	Bell Bouy			
	Old fort deep			
	Lobster reef			
	Round Bar			
	Fairy Dust			
	Pine Shallows West (the deep)			
	Pine Shallows			
	Pine Shallows East(Collymore rock)			
	Atlantic shores			
	Silver Sands			
	Long beach			
	Ocean city			
	foul bay			
	crane			
	sam lords			
	harrismith			
	well pits			

ST. LUCY
ST. PETER
ST. ANDREW
ST. JAMES
ST. JOSEPH
ST. THOMAS
ST. JOHN
ST. GEORGE
ST. PHILIP
ST. MICHAEL
CHRIST CHURCH
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2.5
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Kilometers
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