PHYSIS

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CIEE Research Station

PHYSIS φύσις

Physis, first used by Homer to describe the growth of nature, embodies the essence of conservation. The term is simply defined as the process of allowing nature to heal itself. *Physis* exemplifies nature's ability to rebuild and maintain a steady state, where all intricate processes work in synchronicity.

Despite being in a time of ingenuity and progress, there is a mistaken belief that humans can recreate nature, something that is so beautifully complex. Growth defines human culture. Unsustainable growth, however, characterizes the human race as the most destructive species on the planet. As Chief Seattle, a Native American proponent of ecological responsibility stated:

"Humankind has not woven the web of life. We are but one thread within it. Whatever we do to the web, we do to ourselves. All things are bound together. All things connect."

As we blindly manipulate land and sea, we are slowly ensuring our demise. While some evidence of this destruction is seen above the surface, the damage below the surface remains hidden. The ocean has been treated carelessly, as if it were a vast basin with infinite resources. Through the processes of overfishing, eutrophication, pollution, habitat destruction, disease, and climate change, the ocean is severely crippled, a mere shadow of the robust entity that it once was.

The disappearance of species thought to be indestructible and the irreparable habitat degradation has enlightened some people to make a difference. It is important to remember one detail: Nature has always been, and will always be, the best possible restorative agent. It is vital to understand our place in nature, and that we are just as vulnerable to destruction as any other species.

Upon our arrival to Bonaire, we approached the sea from the surface, which seemed so complete and unbreakable. We embarked on our mission and plunged in. We discovered a world we never could have imagined. We were also exposed to the scars of its destruction.

The studies in this journal are to help others gain an understanding of *Physis*. During our time here we have had many remarkable experiences that will never be forgotten. We have compiled our research as a way to give thanks to Bonaire for teaching us valuable lessons about conservation and the value of nature.

We present *Physis*.

FOREWORD

The Council on International Educational Exchange (CIEE) is an American non-profit organization with over 100 study abroad programs in 41 countries around the world. Since 1947, CIEE has been guided by its mission:

> "*To help people gain understanding, acquire knowledge, and develop skills for living in a globally interdependent and culturally diverse world*."

The Tropical Marine Ecology and Conservation program in Bonaire is a one-of-a-kind program that is designed for upper level undergraduates majoring in Biology. The goal of the CIEE Research Station Bonaire is to provide a world-class learning experience in Marine Ecology and Conservation. The field-based science program is designed to prepare students for graduate programs in Marine Science or for jobs in Natural Resource Management and Conservation. Student participants enroll in six courses: Coral Reef Ecology, Marine Ecology Field Research Methods, Advanced Scuba, Tropical Marine Conservation Biology, Independent Research and Cultural & Environmental History of Bonaire. In addition to a full program of study, this program provides dive training that prepares students for certification with the American Academy of Underwater Scientists, a leader in the scientific dive industry.

The student research was conducted within the Bonaire National Marine Park with permission from the park and the Department of Environment and Nature, Bonaire, Dutch Caribbean. The research this semester was conducted on the leeward side of Bonaire where most of the population of Bonaire is concentrated. Students presented their findings in a public forum on the 23rd of December 2011 at the station for the general public.

The proceedings of this journal are the result of each student's Independent Research project. The advisors for the projects published in this journal were Rita B.J. Peachey, PhD and John A.B. Claydon, PhD. In addition to faculty advisors, each student had CIEE Interns that were directly involved in logistics, weekly meetings and editing student papers.

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Dr. John Claydon is the Tropical Marine Conservation Faculty. He received a B.S. in Marine and Environmental Biology from St. Andrews University in Scotland and a M.S. and Ph. D. degree in Tropical Marine and Fisheries Ecology from James Cook University in Australia. His research interests include spawning aggregations of coral reef fishes, spatial ecology of groupers, nursery habitat enhancement for groupers and spiny lobsters, the red lionfish invasion, migration of reef fishes and reef fish fisheries. John teaches Tropical Marine Conservation Biology and Independent Research.

Professor Caren Eckrich is the Coral Reef Ecology Faculty and the Dive Safety Officer for CIEE. She holds a B.S. in Wildlife and Fisheries Management from Texas A&M University and a M.S. in Biological Oceanography from the University of Puerto Rico in Mayaguez. Caren is also the instructor for Marine Ecology Field Research Methods and Advanced SCUBA. Caren's research interests include fish behavior, seagrass and algal ecology, sea turtle ecology, and coral disease and she is currently working on a study of algae that over grow corals in Lac Bay.

STAFF

Amy Wilde is the Administrative Assistant for CIEE. She holds a Bachelor of Science in Business Administration as well as a Masters of Science in Management Administrative Sciences in Organizational Behavior from the University of Texas at Dallas. She has worked in call center management for the insurance industry and accounting for long term care while living in Texas. Amy currently provides accounting and administrative support for staff and students at CIEE.

Anouschka van de Ven is the Assistant Resident Director for CIEE. She is a PADI dive instructor and underwater videographer. She assists with Advanced SCUBA and Cultural and Environmental History of Bonaire courses. She has a BA First Class Honors degree in communications studies from the London Metropolitan University and worked in television and advertising in Amsterdam before moving to Bonaire. Anouschka provides administrative support for the research station and is responsible for the website and public relations. She is also a volunteer operator at the hyperbaric chamber.

INTERNS

Jason Flower assisted Dr. Claydon with Independent Research and Prof. Eckrich with Advanced SCUBA and Marine Ecology Field Research Methods. He recently finished his masters degree in Tropical Coastal Management at Newcastle University in England during which he carried out research into Tobagan fishers' livelihood security and attitudes to management. He has worked as a PADI Dive Instructor for the past 6 years in several locations including Honduras, Grand Cayman, Greece and Tobago where he ran the dive operations for a reef conservation organization. His experience also includes two years teaching English in Latvia.

Lisa Young is the intern for Tropical Marine Conservation Biology, Cultural and Environmental History of Bonaire, and Independent Research. She is a native Floridian with an A.A. in Business from Valencia College, and a B.S. and M.S. in Marine Biology from Florida Institute of Technology. Lisa's research interests involve coral reef fish ecology, and she hopes to continue teaching and research.

Christina Wickman is the Coral Reef Ecology Intern. She recently received her undergraduate degree in Marine Biology from the University of Oregon. In the fall of 2008 she was a student at CIEE Bonaire, where she looked at the possibilities of predicting coral bleaching around the island. Her research interests include coral reef ecology, coral reef preservation and public education of tropical reef ecosystems.

TABLE OF CONTENTS

Effects of differences in hole size of artificial reef units on coral reef fish abundance and species richness
Tessa J. Code……1–10

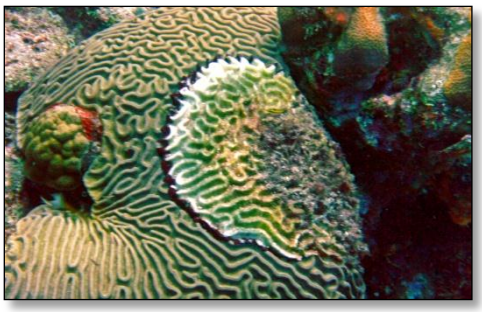
Potential for transmission of yellow band disease between colonies of *Montastrea annularis* through common research techniques
Dana L. Hergert……...……11–15

Effects of a small harbour on nutrient levels and coral reef health in Bonaire, Dutch Caribbean
Kendall G. Miller……...……16–20

Fish density, species richness, and diversity between stands of natural and artificial *Acropora cervicornis*
Stephen A. Nelson…… ……..……..……21–27

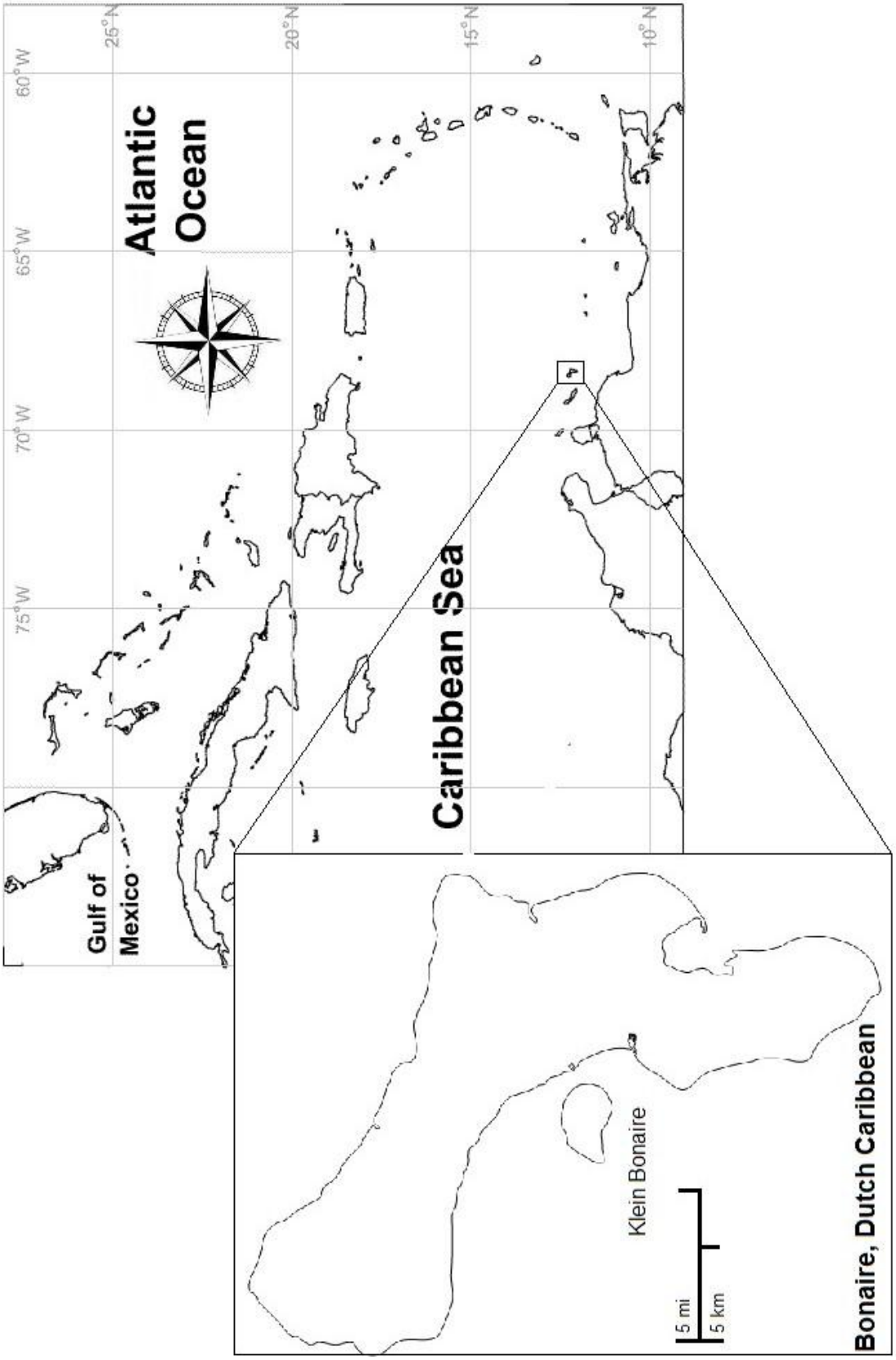
Response of three predatory reef fish species to fishing protection in Bonaire, DC
Jack C. Olson…….…….……28–34

Sponge abundance and the prevalence of disease on scleractinian coral in Bonaire, DC
Luke N. Powell……...……...……35–39



Assessing the correlation between coral disease prevalence and fish species richness from 2007-2011 in Bonaire, Dutch Caribbean
Ian M. Ster……...…..…40–46

Use of trophic structure as an indicator of reef fish assemblages in areas open and closed to fishing
Benjamin E. VanDine……...……..47–52


Atlantic Ocean
Caribbean Sea
Gulf of Mexico
60°W
65°W
70°W
75°W
25°N
20°N
15°N
10°N
Klein Bonaire
5 mi
5 km
Bonaire, Dutch Caribbean

Effects of differences in hole size of artificial reef units on coral reef fish abundance and species richness

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Abstract

The biodiversity of coral reefs will decline as the global degradation of reefs continues, pushing the science of restoration techniques such as the utilization of artificial reefs, to the forefront of coral reef ecology. Artificial reefs provide shelter and habitat through design features such as topographic complexity, substrate diversity, refuge hole size, vertical relief, and percentage live coral cover. These elements have been manipulated to study the relationship between design and fish community response. This study examines the relationship between hole size in artificial reef units (ARUs) and species richness and abundance of coral reef fish at a study site on Bonaire, Dutch Caribbean. Four types of ARUs were constructed including units with no holes, small holes, large holes and a combination of small and large holes. Various fish and invertebrate species utilized ARUs for grazing, benthic egg laying, protection and hunting. There was no difference in fish density or species richness among the four ARU types. However, differences in species composition among the four ARU types were found. The smaller, benthic fish were more prevalent at the small hole ARU, while the larger, territorial fish were found utilizing the large hole and no hole ARUs more often. Mixed hole ARUs exhibited a compilation of both small and large fish. Since there are differences in community composition, specialized artificial reef designs could possibly be used to enhance particular species that are important to maintain community structure of degraded reefs.

Introduction

Over the past 40 years, coral reefs have been degraded by overfishing, sedimentation, disease, storms, coral bleaching, and eutrophication caused by increasing anthropogenic pressure on reefs through poor land use practices, including coastal development and agriculture; unsustainable fishing practices; lack of sewage treatment and global climate change (Rilov and Benayahu 2002; Bries et al. 2004; Alvarez-Filip et al. 2009). Coral degradation reduces the topography and structural complexity of coral reefs. Although coral reefs only make up 0.09% of the ocean, roughly 25% of all fish species inhabit this ecosystem (Wilson et al. 2006). Studies ranging from the late 1970s to the current day indicate that coral reefs with extensive habitat complexity yield higher species richness and an increased abundance in fish (Luckhurst et al. 1978; Roberts and Ormond 1987; Ferreira et al. 2001; Gratwicke and Speight 2005a, 2005b). The complexity of reefs is attributed to the diversity of coral species and the integration of different sizes, shapes, and refuges, providing habitats for specialist species (Gratwicke and Speight 2005a).

In a single area, individuals can use distinct spaces for shelter from predators, breeding, camouflage (Ferreira et al 2001; Rilov and Benayahu 2002; Gratwicke and Speight 2005a) and refuge from environmental stressors (Alvarez-Filip et al. 2009). Caribbean structural complexity has deteriorated significantly due to the white band disease epidemic that greatly reduced *Acropora cervicornis* and *Acropora palmata* in the early 1980s. Additionally, mass bleaching events in 1998 jeopardized coral growth, reproduction, and increased the susceptibility to coral disease (Steneck 2009). Bleaching, in congruence with elevated disease prevalence, collectively reduced the proportion of structurally

complex reefs in the Caribbean from 45% to only 2% of based on relative rugosity measurements (Alvarez-Filip et al. 2009). On the islands of Curacao and Bonaire, located in the Dutch Caribbean, the regional deleterious factors plus Hurricane Lenny in 1999, decreased coral cover by 25-70% on the leeward sides of the islands (Bries et al. 2004).

The combination of species richness and abundance govern an ecosystem's ability to resist environmental stressors and provide resilience (Wilson et al. 2006). High biodiversity is a type of ecosystem insurance because repetitive functionality provides ecological replacement when coral reef ecosystems experience climate change, disease, overfishing, and invasive species (Chapin et al. 2000). Areas with higher biodiversity are more stable, show slower rates of degradation and habitat collapse in addition to decreased extinction rates of commercially favored fish and invertebrate species (Worm et al. 2006).

Although there has been a large decrease in coral cover throughout the Caribbean due to anthropogenic stressors, previous observations have shown this does not immediately equate to a substantial loss of local marine biodiversity because the dead coral is able to temporarily persist as a structural framework (Wilson et al. 2006; Alvarez-Filip et al. 2009). In fact, Luckhurst and Luckhurst (1978) found no correlation between live coral species richness and fish species richness. However, with frequent storms, strong wave action, and parrotfish biting, the coral skeleton framework will erode leading to an architecturally homogenous environment (Bries et al. 2004; Alvarez-Filip et al. 2009). In order to preserve future biodiversity in the Caribbean, the survival of structurally complex coral reefs is essential to sustain the maximum level of marine diversity.

Artificial reefs have been utilized since the 1700s to replenish fisheries and to improve fish stocks (Bohnsack and Sutherland 1985). Since 1985, France has deployed 40,000 m^3 of artificial reefs in order to increase fish assemblages (Charbonnel et al. 2002). In the United States, over 500 reefs had been deployed in the coastal waters by 1997, costing approximately 1 million dollars annually (Grossman et al 1997). Primarily used by Japan and the United States, artificial reefs are constructed of varying shapes, sizes and components. Materials for artificial reefs vary and most reefs constructed in the United States are comprised of scrap materials and discarded construction items like tires, pipes, rock, concrete blocks and ships (Bohnsack and Sutherland 1985). In contrast, the Japanese National Government has only approved "steel reinforced or pre-stressed concrete, rubber, polyethylene concrete and fiberglass reinforced plastic" as acceptable materials for artificial reef construction (Bohnsack and Sutherland 1985).

Fish use artificial reefs for protection, feeding, orientation, and spawning (Bohnsack and Sutherland 1985; Grossman et al. 1997; Rilov and Benayahu 2002). The response in fish abundance and species richness to an artificial reef differs due to design and geographic location. Presently, the factors influencing the success of artificial reefs remain poorly understood. Artificial reefs have been observed to provide habitat for solitary, roaming individuals, some separated from large local populations, and can become a secondary epicenter of biomass (Alfieri 1975). The subsequent decline in fish abundance after the collapse of a natural reef could be mitigated by a nearby artificial reef. If a structure is within a short migratory distance, then it may act as an alternative habitable location by providing a permanent area to settle or as a temporary launching point for species to migrate to distant natural reefs (Grossman et al 2007; Wilson et al. 2006).

Studies found that artificial reefs had many times more fish biomass than natural reefs (Alfieri 1975; Lim et al. 1976; Bohnsack and Sutherland 1985). A previous study attributed greater fish biomass on

artificial reefs than the natural habitat to the increased complexity and superior protection from predators in the artificial structure (Smith et al. 1979). Although this may not be the case in all artificial reef projects; artificial reefs are used as an important fishery tool that can supplement fisheries (Bohnsack and Sutherland 1985; Grossman et al 2007).

Due to the usefulness of artificial reefs for fish attraction, stock replenishment, and as a replacement for the degraded natural reefs, a number of studies have been published regarding the ideal reef construction to maximize biodiversity under specific circumstances (Lim et al. 1975; Smith et al. 1979; Bohnsack and Sutherland 1985; Roberts and Ormond 1987; Ferreira et al. 2001; Charbonnel et al. 2002; Gratwicke and Speight 2005a, 205b). Over the decades, numerous designs have been utilized for a variety of purposes and no design has been widely accepted even for coral reefs. The most common studies about artificial reef construction compare fish species richness and abundance with variables like rugosity of the substrate (Roberts and Ormond 1987; Charbonnel et al. 2001; Gratwicke and Speight 2005b), habitat complexity (Roberts and Ormond 1987; Charbonnel et al. 2001; Ferreira et al 2001; Gratwicke and Speight 2005b) vertical relief (Ferreira et al. 2001; Rilov and Benayahu 2002; Gratwicke and Speight 2005b), growth forms (Gratwicke and Speight 2005b), variety of hole sizes (Luckhurst and Luckhurst 1978; Hixon and Beets 1989; Ferreira et al. 2001; Gratwicke and Speight 2005a, 2005b), percent hard substrate (Gratwicke and Speight 2005b), percent live coral (Roberts and Ormond 1987), substratum diversity (Roberts and Ormond 1987; Ferreira et al. 2001) and spatial arrangement (Roberts and Ormond 1987; Ferreira et al. 2001; Charbonnel et al. 2002; Gratwicke and Speight 2005a, 2005b). In fact, according to a comprehensive study comparing habitat assessment scores by Gratwicke and Speight (2005a) the aforementioned artificial reef characteristics resulted in as much as 71% of the species richness in near-shore coastal habitats.

The present study compares the effects of difference holes sizes on coral reef fish species richness and abundance in Bonaire. On fringing reefs in the Red Sea, Roberts and Ormond (1987) found that the largest number of holes in natural reefs were located between the depths of 1.5 m- 6 m and that 77% of the variability in fish abundance was explained by the total number of holes of 5 size classes on the reef. However, there was no correlation between number of holes and species richness. Hixon and Beets (1989) compared hole quantity in artificial reefs and found a direct relationship between abundance of large holes and number of large fish: large fish were more abundant on the artificial reef with large holes and small fish were more abundant on the artificial reef with small holes. Gratwicke and Speight (2005b) compared artificial reef unites (ARUs) with small hole sizes, large hole sizes and multiple hole sizes to test this hypothesis that an increase in the variety of hole sizes would increase species richness and abundance of fish. ARUs with different hole sizes did not increase the species richness or abundance of fish resulting in the conclusion that the number of holes needed to be constant in order to test the hypothesis (Gratwicke and Speight 2005b). The present research maintains a constant number of hole sizes to address the unintentional effects of different numbers of holes in the previous study by Gratwicke and Speight (2005a, 2005b). This study investigates the relationship between species richness and varying hole size in ARUs, maintaining a constant number of holes in each unit and testing the following hypothesis:

H_1: ARUs with multiple hole sizes will support a greater fish abundance and species richness than ARUs with one hole size or no holes.

By standardizing the number of holes, this study will shed light on the difficulties in determining the relationship between fish

abundance and species richness due to hole size.

Materials and Methods

Study Site

The ARUs were deployed on the leeward side of Bonaire, Dutch Caribbean where reef flattening is similar to reefs in the rest of the Caribbean (Bries et al 2004, Alvarez-Filip et al. 2009). The fringing reef surrounding Bonaire is a marine park, protected by the National Parks Foundation, STINAPA. The ARUs were placed at the Yellow Sub dive site (Fig. 1) where mooring blocks that were placed by the marine park near the reef crest have recruited live corals and attracted a variety of fish species showing that artificial structures have been successful at the study site in the past.

Initial Construction

Four types of ARUs (40 cm x 60 cm x 30 cm) were constructed using concrete blocks and mortar. Four replicates of each type were constructed resulting in total of 16 ARUs. The types included an ARU with no holes, constructed with three large and three small solid blocks (Fig. 2A). An ARU with 6 large holes was constructed using six concrete blocks with 15 cm x 15 cm holes and three solid blocks (Fig. 2B) and an ARU with 6 small holes was constructed using two concrete blocks with 6 cm x 6 cm holes, two large solid blocks and three small solid blocks (Fig. 2C). The fourth ARU type consisted of a combination of small and large hole sizes and included two blocks with 15 cm x 15 cm holes, two blocks with 6 x6 cm holes, one large solid block and three small solid blocks (Fig. 2D).

To standardize the number of holes in each ARU, excess holes were filled with mortar. After construction, the blocks were moved by hand truck to the Yellow Sub dock, placed into the water, and then distributed to sites along the reef line using lift bags and scuba. The ARUs were positioned 8 m from the reef crest and 10 m from each other at approximately a 6 m depth.

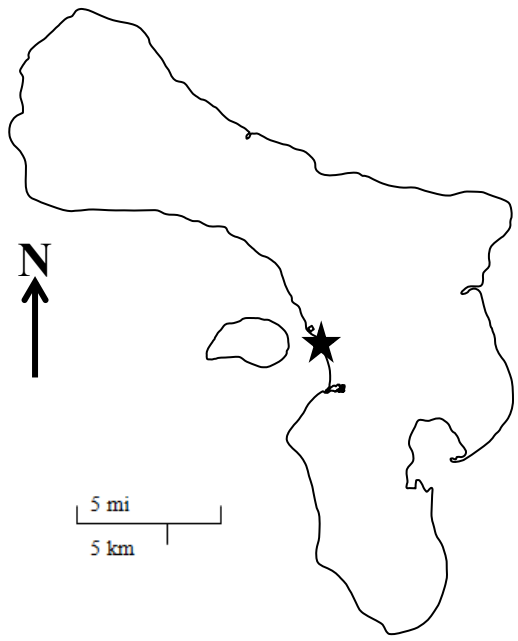

Fig. 1 Map of the island of Bonaire, Dutch Caribbean, showing the artificial reef study site Yellow Sub indicated by a *black star* (12° 09'36.3" N, 68°16'55.2" W)

Data Collection

A five min observation of each block was conducted bi-weekly between the hours of 11:00 and 14:00 using SCUBA. The first two minutes of the observation were made at a distance of approximately 4 m from the block. All species within 20 cm of the blocks were identified and counted. The next observation of fish abundance and species, lasting two minutes, was taken within 1 m from the ARU and lastly, one minute was spent counting and identifying small benthic species and inspecting inside the holes. Life stage, if discernible, was recorded for fish.

Data Analysis

In order to assess fish density around the ARUs, counts were divided by the area around the unit that was included in observations. Fish from the first observation period were added to the second and final observation period resulting in a measure of

the density of fish (individuals m^{-2}). These values were summed by unit type after every observation period. Species richness was calculated by summing the presence or absence of each species for every observation period. Since species richness passed parametric assumptions, a one-way ANOVA using (StatView V^3). Fish density did not satisfy parametric assumptions and therefore the Kruskal-Wallis test was utilized.

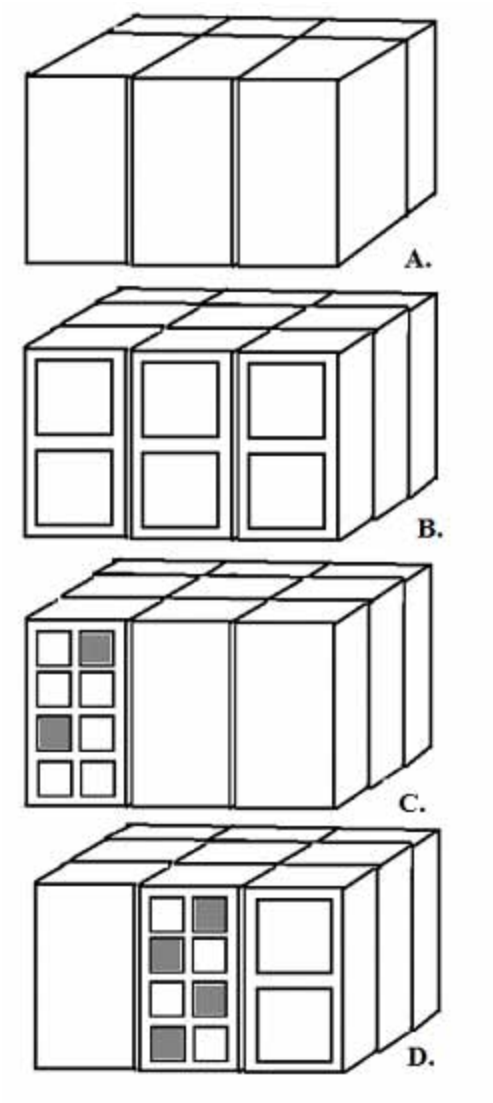

Fig. 2 Diagram of the 4 types of artificial reef units (ARUs) used in this study (40 cm (h) x 60 cm (w) x 30 cm (d)) constructed with concrete blocks and mortar including ARUs with (A) no holes, (B) large holes, (C) small holes, and (D) a combination of small and large holes. Holes in the concrete blocks with *solid grey* were filled with mortar to standardize the number of holes among blocks

Results

Fish communities were surveyed eight times over a period of three weeks. A total of 42 species and 5244 total fish were observed. Excluding gobies, a total of 382 fish utilized ARUs with no holes, 653 fish used ARUs with small holes, 292 fish used ARUs with large holes and 622 fish were observed on ARUs with a combination of large and small holes.

There was no difference in fish density (individuals m^{-2}) among ARUs (χ^2 = 6.261, n = 8, p = 1.10). Fish density ranged from 77.9 individuals m^{-2} to 211.6 individuals m^{-2} during the 3 week sampling period. The mean fish density was lower for ARUs with no holes and ARUs with large holes (Fig. 3). ARUs with small holes and mixed holes had means that were around 20% higher than the other ARU types. Large hole and no holed ARUs had the smallest average fish density. Small hole blocks had the largest density value of approximately 138 individual m^{-2}.

There was no difference in fish species richness among the 4 ARU types (F = 0.438, df = 3, p = 0.728). The mean number of species was similar among the 4 types of ARUs varying by approximately 0.5 species (Fig. 4).

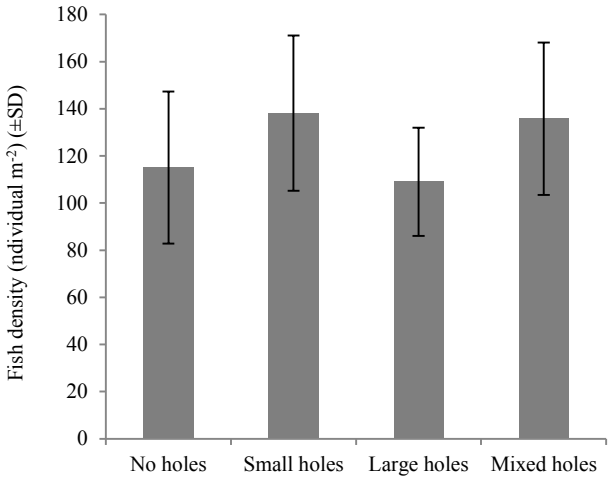

Fig. 3 Comparison of mean fish density (± SD) among four artificial reef treatments (n = 8)

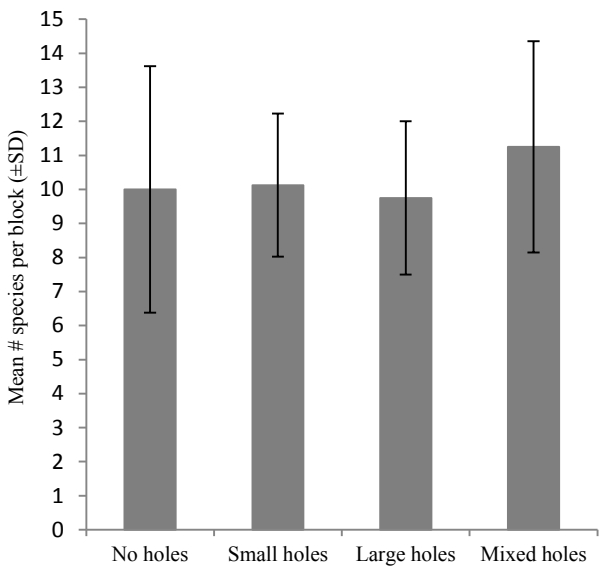

Fig. 4 Comparison of mean species richness (±SD) among 4 artificial reef units with differing holes sizes (n = 8)

As a whole, the relative species composition was very similar between ARUs. The most common species found around all blocks were the bicolor damselfish (*Stegastes partitus*), slippery dick wrasse (*Halichoeres bivittatus*) and sergeant major (*Abudefduf saxatilis)*. Sergeant majors were more abundant on large hole and no holed blocks but seen to a much lesser degree on small holed ARUs. Sergeant majors were observed to use the inside of large holed ARUs as a protective area for nests. The sergeant majors that colonized the no holed blocks were observed to attack more frequently, possibly due to the exposed egg location. Sergeant majors also attacked smaller, benthic species and grazing herbivores. The blocks, with sergeant majors had a comparably lower amount of smaller species like bicolor damselfish and juvenile wrasse (Fig. 5) and had thicker alga growth than the other block types due to attacks on other herbivores. Although there were proportionally less Acanthurid species on the large and no holed blocks, there was a greater diversity of fish like butter hamlets (*Hypoplectrus unicolor)*, four eyed butterflyfish (*Chaetodon capistratus)* and yellow goatfish (*Mulloidichthys martinicus)* at these blocks (Fig. 5). Two common octopus (*Octopus vulgaris*) also colonized the large holes of an ARU for a period of time.

Sergeant majors used the no hole ARUs frequently, sometimes colonizing every side, for egg laying. The ARUs with no holes had the highest abundance of predators including trumpetfish (*Aulostomus maculatus)*, graysby (*Cephalopholis cruentata)*, schoolmaster snapper (*Lutjanus apodus)* and bar jack (*Carangoides ruber)* (Fig. 5). These predators were observed using the blocks as a mechanism for hunting. Bar jacks and trumpetfish were seen using the blocks to sneak up on prey. In addition, many of the larger herbivorous fish were seen feeding on algae growing on the blocks; most common was the ocean surgeonfish (*Acanthurus bahianus*), four-eyed butterflyfish and various parrotfish species (Fig. 5).

The highest amount of small fish and juveniles were observed on the small holed ARUs. Most of the species composition can be attributed to the bicolor damselfish and species of wrasse (Fig. 5). Smaller fish like the harlequin bass (*Serranus tigrinus*) and sharpnose puffers (*Canthigaster rostrata*) were also seen near these blocks. In addition, high abundances of juveniles of slippery dick, bluehead wrasse and ocean surgeonfish were observed using these blocks as protection. Density of sergeant majors were low on small holed ARUs (Fig. 5).

The mixed hole ARUs had a more equal distribution of small fish and larger species. There was a high proportion of bicolor damselfish, longfin damselfish, wrasse, and juveniles observed in the mixed holed ARUs as well as sergeant major and Acanthuridae species (Fig. 5). Multiple banded coral shrimp (*Stenopus hispidus*) were commonly observed using the small holes within these blocks.

Discussion

The results of this study did not support the initial hypothesis that differences in species

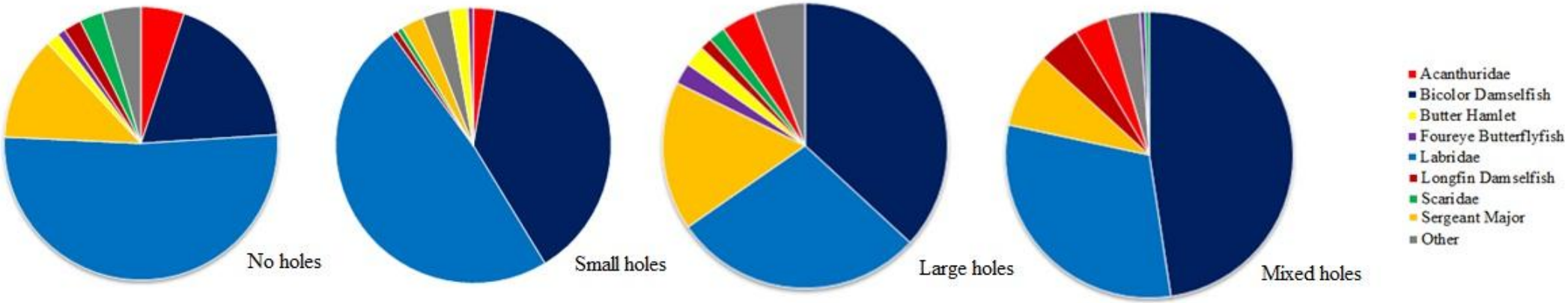

Fig. 5 Pie chart showing the differences in fish species composition among the 4 artificial reef units utilized (n = 8). The *other* category includes species that three individuals or less were seen and harlequin bass, yellow goatfish, and yellowfin mojarra that were seen infrequently on one ARU. Labridae species combines bluehead wrasse (*Thalassoma bifasciatum)*, slippery dick (*Thalassoma bifasciatum)*, and yellowhead wrasse (*Halichoeres garnoti*) values. Scaridae species includes all parrotfish species observed. Acanthuridae species includes all surgeonfishes that were seen. Gobiidae species were excluded from the analysis because they were consistently high on all block types

diversity and fish density would increase at ARUs with mixed hole sizes. There was no significant difference in mean densities among the four different ARU types.

Sergeant majors utilized the no hole and large hole ARUs for egg laying. The defensive habits of this species to protect their nests could contribute to the lower average fish density seen in the no hole and large hole ARUs.

Mean species richness was equivalent across all block types. This could be explained by the proximity of the ARUs to each other, allowing easy migration of species to multiple units. Although there was no difference in species richness, the differences in species composition show that there were specialized uses for different types of blocks by fish. For example, sergeant majors were observed to use the large holes for egg laying. These areas are more easily protected because they are only approachable from one direction.

The ARUs in this study provided an added complexity to the habitat that provided shelter and food, were utilized by fish for hunting. The excess algae seen on the blocks heavily defended by sergeant majors and absence of algae on blocks that did not facilitate sergeant major recruitment implies that ARUs were grazed and provided food. In addition, fish were observed residing close to blocks, indicating that the presence of the blocks acted like satellite reefs. The blocks, separate from the reef crest, provided increased habitat and allowed individuals to venture further into the area. Predators also utilized the blocks as a method of camouflage from prey. The diversity of these mechanisms supports the idea that artificial reefs are useful in providing a secondary habitat for species in declining complexity areas.

ARUs are usually colonized quickly and species may change seasonally or illustrate a successional pattern (Lim et al. 1976; Bohnsack and Sutherland 1985). However, equilibrium community structure may take several months and maintaining a consistent community structure can take multiple years (Bohnsack and Sutherland 1985). This could explain the similarity between all artificial units deployed for the present study. Bicolor damselfish, sergeant majors and wrasses were abundant among all block types (Fig. 5). This could be due to high concentrations of these species throughout the reef and the high level of competition for space on the reef (Hixon and Beets 1989).

Some suggestions for future studies include a longer observation period and varied daily observation times. Because fish partake in different activities throughout the day, it would have been helpful to observe the ARUs at various times, especially in the late evening. The original hypothesis could possibly be supported if fish utilized the holes in the evening, when predation is at its highest. In addition, conducting the study during different weather conditions would present a more complete picture on the use of the blocks. One observation occurred when it was raining with low visibility underwater. This period resulted in the highest fish count on the blocks and in the open areas away from the protection of the reef. The distance between the reefs may have also been a confounding factor because the proximity allowed a significant amount of migration between areas and less pressure on selection and colonizing. If the reefs were further apart, perhaps the populations of individual blocks would have been more consistent and a greater difference in overall species composition among types would have occurred.

The holes in the artificial reefs were not utilized as expected. According to Hixon and Beets (1989), shelter of the appropriate size is a limiting factor for reef fishes and these fishes particularly choose shelter close to their body size. The absence of this relationship in this study may indicate that the other hole sizes are important. For example, small benthic fish like gobies and bicolored damselfish were only seen hiding in crevices underneath the blocks. Perhaps even the small holes were too large and open for some of the smaller fish vulnerable to

predators. In fact, the most consistent use of holes for protection was by the banded coral shrimp. There was an absence of bigger fish hiding in the larger holes; however, this may be due to the reduced presence of apex predators and overall predation on the reef.

Hole sizes in reefs are important because they are utilized by fish of various sizes as protection from predators (Luckhurst and Luckhurst 1978) and the design of ARUs will benefit from the study of the influence of hole size and number on coral reef fish communities. There is still a lack of clarity about the relationship between factors such as fish density, fish species richness and fish diversity and the abundance and size of holes in coral reefs. Many studies have found that various types of complexity are related to species richness and biodiversity (Luckhurst and Luckhurst 1978; Roberts and Ormond 1987; Charbonnel et al. 2001; Ferreira et al. 2001; Gratwicke and Speight 2005b) and it will be important for the development of ARUs to understand the importance of hole size an abundance to fish living on coral reefs if ARUs are going to be utilized in coral reef habitats to enhance coral reef fish populations.

Coral reefs support ecologically and economically robust fish communities, a driving force for tourism and diving. The loss of coral reefs would have a significant impact on fish biodiversity, recruitment, and survival. Artificial reefs can be used as a tool to support fish populations in the face of reef degradation. Ultimately, artificial reefs may help nature endure through changing environmental conditions and continue to be productive.

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Potential for transmission of yellow band disease between colonies of *Montastrea annularis* through common research techniques

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Abstract
This study investigates the relationship between yellow band disease (YBD) infected *Montastrea annularis* coral colonies and its potential to spread by contact with transect tapes, as commonly used in research. *M. annularis* plays an important role in maintaining reef complexity and diversity in Bonaire as it is a structural reef-building coral, yet the recent spread of YBD has created an degrading the reefs (Bruckner 2006). The study has two primary goals, to investigate whether the use of transect lines had the potential to transfer YBD from one coral to another, and whether a simple cleaning protocol can reduce this transfer. Transects placed on the YBD infected colonies of *M. annularis* had the most percent bacterial growth, though not statistically significant. It also showed that even though specific species of bacteria were unable to be identified, the transect lines are indeed capable of carrying bacteria. Although the difference was not significant in this study, cleansing treatments may have an effect on lessening the growth of bacteria.

Introduction

It has been estimated that coastal and oceanic ecosystems account for approximately two-thirds of the world's natural ecosystem services (Craig 2007). Coral reefs provide many benefits to the success of tropical marine environments; they promote ecosystem biodiversity, provide coastal protection against storm systems and erosion, and support economic value through fisheries and tourism (Craig 2007). However, there are currently many factors that are threatening the health of corals, diseases being a prominent example. Coral diseases have become increasingly prevalent in the Caribbean and are predicted to continue to spread at an increasing rate (Goreau et al. 1998). Diseases can affect many life forms, and can cause extreme harm to the stability of an ecosystem. For example, in the 1980s, the loss of bird habitat through Dutch elm disease in England and Wales led to a loss of bird diversity. (Osborne 1985). The same process is occurring on the coral reef ecosystems around the world, particularly in the Caribbean (Goreau et al. 1998). It is believed that the health of a reef can be determined solely by the diversity of fish found, yet diseases can negatively modify the ability of a reef to support such a diverse population (Harvell et al. 2008). Yellow band disease (YBD) is a bacterial infection that commonly attacks colonies of reef-building corals, particularly *Montastrea annularis.* It kills the symbiotic algae of a coral, causing bleaching and eventually killing the coral. YBD has a particular presence in the Caribbean, specifically in the Lesser Antilles and Bonaire, where its existence has reached a status of an epidemic among the reefs (Dona et at. 2008). YBD directly targets colonies of *Montastrea annularis,* which is often argued to be one of the most important corals in Atlantic reefs due to its extensive distribution and abundance (Weil and Knowlton 1994).

The composition of the bacteria causing YBD has yet to be determined, but studies have shown that outbreaks have occurred after a significant rise in sea temperature (Harvell et al. 2008). An example of this was a major increase in YBD abundance among the coral reefs of

the Caribbean after an irregularly warm season in 2005 (Miller et al. 2006). Other causes of disease outbreaks are overfishing and increased nutrient levels (Harvell et al. 1999). One possible cause of disease transfer could be through direct contact by humans.

The overall purpose of this study was to determine whether the use of transect tapes in scientific research has the potential to act as a vector transmitting YBD between colonies of *M. annularis.* In addition, simple procedures to limit transfer of bacteria were tested. Little is known about the transmission of coral reef disease through human activities. However, it is important to address this potential vector as an increase of coral disease may lead to an increase of studies and thereby a heightened transmission of YBD.

The hypotheses that were tested within this project are as follows:

H_1: Transect tapes used as research equipment can carry and transmit YBD among coral colonies.

H_2: Washing and sterilizing research equipment between dives and research locations can reduce the ability for the equipment to act as a vector for YBD.

The results from this study may help better understand disease transmission and to prevent the spread of YBD among colonies of *M. annularis* on the reefs of Bonaire.

Materials and Methods

Observations and data collection on the effects of transect tapes and the transmission of YBD was completed at Yellow Sub dive site in Bonaire, Dutch Caribbean (N 12°09'36.3", W 68°16'55.2") (Fig. 1). This site was chosen for data collection because of high prevalence of YBD found in pilot research; it also presents itself as a site greatest at risk for transmission due to the fact that it is a highly researched location.

Study Site

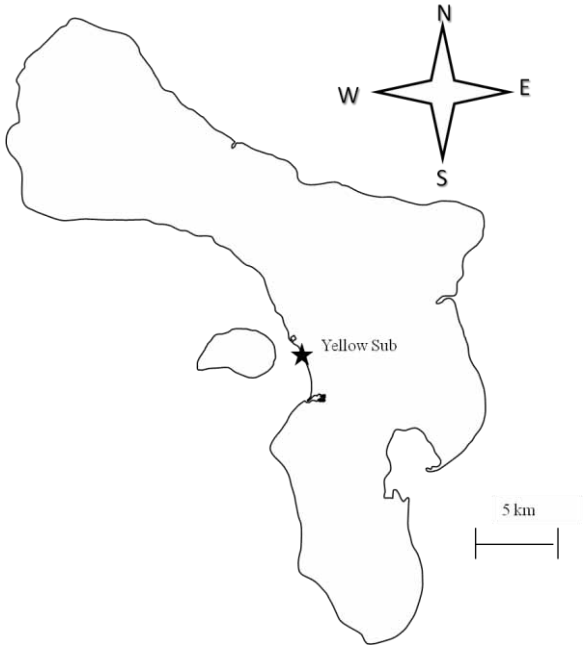

Fig. 1 Map of Bonaire, Dutch Caribbean indicating the location of the sample site, Yellow Sub

Sampling

The data collection took place at a depth of 5 m to 10 m because the greatest density of *M. annularis* infected with YBD are found within these depths (Dona et al. 2008).

Transect tapes are regularly used to study the coral reef benthos (Lang et al. 2010). Therefore, we attempted to simulate how bacteria could be transferred to transect tapes during Atlantic Gulf Rapid Reef Assessment (AGRRA) surveys (Lang et al. 2010).

A sterilized transect tape was cut into 50 cm long strips and fastened with 28 g fishing weights on either end to keep in place. A set of 30 control strips were placed upon colonies of *M. annularis* with no visible sign of disease, and an additional 30 strips placed on colonies infected with YBD. The individual strips of transect tape were set in location on the different coral colonies for a mean total of 20 minutes, which is how long transect tape would lie over the benthos during an AGRRA survey (Lang et al. 2010). Once collected, tapes were subjected

to one of three treatments: (1) rinsing with fresh water, (2) rinsing with a bleach solution (10% bleach concentration), and (3) a control treatment involving no rinse. Thus 20 strips of transect tape were subjected to each treatment, 10 that were deployed on healthy corals and 10 from YBD infected corals.

Strips were then swabbed and material was transferred to agar plates with a sterile loop to assess bacterial growth (as per Robertson et al. 1951). The agar plate, or streak plate method, was used because it can isolate individual bacterial strains and multiply into separate colonies that are clear to evaluate (Perry 2002). The number of scrapes per plate was set at 15. Plates were incubated at a constant temperature of 35.5 ºC. The area of bacterial growth on the agar plate was then measured from digital photographs using ImageJ (Kohler and Gill 2006). Percent cover, different types of bacterial growth, and rate of spread were recorded.

Data analysis

Differences in percent bacterial cover were analyzed using a 2-factor ANOVA with YBD diseased versus healthy corals, and bleach rinse versus freshwater rinse versus no rinse as levels.

Results

In the trials where no rinse were applied, the overall mean percent growth cover of bacteria was observably greater on the transects that were placed on YBD infected *M. annularis* than those on non-infected colonies (Fig. 2). This relationship was also found to be true with the transects that were treated with a freshwater rinse though the results were not significantly different (Table 1). The transects that were treated with a bleach solution rinse displayed varying results. The tapes placed on colonies of YBD infected corals that were then rinsed with the bleach solution showed much lower percentages of overall mean bacterial growth in comparison to the other treatments. Tapes placed on the coral colonies that were not infected with YBD had higher percentage of overall mean bacterial growth than other treatments.

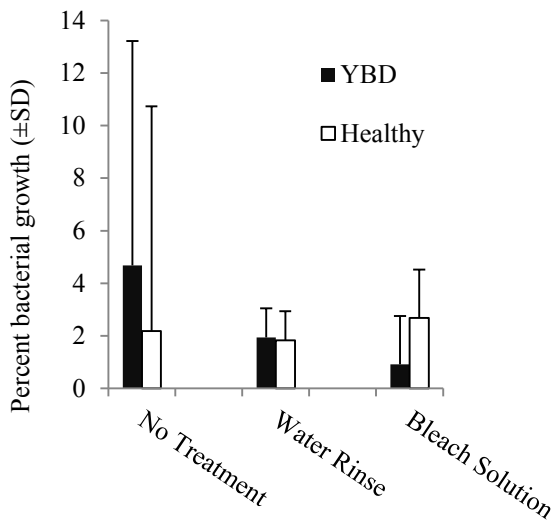

Fig. 2 Mean bacterial growth (% cover of agar plate, ±SD). Bacteria cultured from transect tapes laid on both healthy and yellow band diseased *Montastrea annularis* with three treatments: no rinse, freshwater rinse and bleach solution rinse

There was no significant difference between bacterial growth and any of the treatments ($p = 0.451$; Table 1). The ANOVA test also showed no significant difference between the colonies infected with YBD and those that were not infected ($p = 0.813$), and it looked at an overall comparison of all the levels of the experiment, for which there was no statistical significance for either ($p = 0.342$).

Table 1 Results of two-way ANOVA of bacterial growth from YBD colonies versus healthy colonies and different rinse treatments (no rinse, freshwater rinse, bleach solution rinse)

Source of Variation	*SS*	*df*	*MS*	*F*	*P-value*
YBD vs. non-diseased	0.003	2	0.001	0.811	0.451
Rinse Treatment	9.2E-05	1	9.3E-05	0.0568	0.813
Between	0.004	2	0.002	1.102	0.342
Within	0.069	42	0.002		
Total	0.076	47			

Discussion

The results showed that the trials treated with a bleach solution rinse and placed on non-infected colonies of *M. annularis* had a higher overall mean percent growth cover of bacteria than did the trials placed on YBD infected colonies. This result did not support the original hypotheses that bacterial growth cover would be least from colonies with no YBD and treated with bleach. One possible explanation for this heightened percent growth in these trials is the characteristics of the bacterial growths; for example, abnormal shape, size, and color of the growths. Also in contrast to the original hypotheses, transects treated with freshwater resulted in a higher overall mean percent growth cover of bacteria than those that were given no treatment. Aside from the possible outside contaminant in the bleach, lower bacterial growth rates were observed in these trials which is in support of the original hypotheses.

Large variation among the data (Fig. 2) may be attributed to many other factors, one of which being bacterial growth type. The majority of the bacteria which grew on the agar plates were uniform in color and shape (small round spots with a deep yellow coloring). However, a few abnormal growths appeared on several plates. This was the case with several of the non-infected colony tapes treated with a bleach solution rinse, which had a growth that was light in color, abnormal in shape and size, and nearly transparent, covering a significant portion of the plate. If this bacterial growth was in fact different from the majority of other bacteria measured, it may have been an outside contaminant adding error to the data. It can be assumed that this may have been caused by outside contaminants that lead to inconsistent bacterial growths.

Removing these outliers, the overall trend of the collected data suggests that there is potential for YBD to be transferred through scientific research methods, though further studies on the topic are necessary. This study has shown there is potential for disease transfer, it is important to further the study by specifically identifying the species of bacterial growth through isolation of the YBD bacteria. Current common research methods may hold the potential to spread bacteria, and there may be ways to help prevent that transfer. The data collected showed that the trials with the least bacterial growth were the tapes set on the coral colonies infected with YBD and treated with a bleach solution rinse. If underwater research materials, such as transect tapes, were rinsed in a bleach solution post-diving, it could help slow the rapid spread of disease throughout the Caribbean.

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Effects of a small harbour on nutrient levels and coral reef health in Bonaire, Dutch Caribbean

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Abstract
The creation of harbours through dredging combined with boat maintenance and waste is known to increase nutrients and sedimentation within coastal waters causing the degradation of nearby coral reefs. Winds and currents play a large role in the transportation of nutrients and sediments. This study investigated the distribution of nutrients in water surrounding Harbour Village Marina in Bonaire, Dutch Caribbean, as well as the extent to which the harbour affects adjacent coral reefs. Water samples were collected at increasing distances from the entrance to the harbour and were tested for nutrients. Percent coral cover was estimated using 10 m line-intercept transects. By noting the wind direction and currents when collecting data, it was determined that both the surface and bottom currents flow predominantly north. Results show that current plays a large role in nutrient distribution, but the effects on coral cover are unknown. Coral cover was also much lower north of the harbour in the direction of current flow compared to the south. This study increases the knowledge on nutrient distribution and coral cover as well as the impact harbours have on surrounding reefs. The results can be used for management efforts to help maintain coral reef health, and to keep nutrient levels low during the creation and upkeep of small harbours.

Introduction

Harbours and marinas are usually artificial inlets used to provide protection from intense wave action (Iannuzzi et al 1996). These inlets are often created through processes such as dredging or shoreline stabilization, both of which can lead to a reduction in habitat complexity and microhabitat availability (Iannuzzi et al 1996). Dredging can destroy the reefs (Richmond 1993) and has been known to increase the amount of light reaching the benthic substrate, leading to a possible increase in production of macro algae (Iannuzzi et al 1996).

The creation and upkeep of harbours through dredging is known to inhibit coral growth (Bak 1978). Harbours and boats have been shown to increase concentration of bacteria, nutrients, trace elements, industrial organic micro-pollutants, petrochemicals within the water (Chapman 1992). Boats have been known to leak biocides such as Irgarol 1051 into marinas and surrounding waters (Carbery et al. 2006). Nutrients, sediment, and toxic pollutants are known stressors that degrade coral reefs (Smith et al. 2008), and high nutrient levels can affect coral nutrition, metabolism, and calcification (Richmond 1993). Coral stressors can lead to both increased coral mortality and increased susceptibility to disease (Smith et al. 2008).

Coral reefs throughout the Caribbean are being degraded due to a number of anthropogenic and natural causes such as sedimentation, dredging, shore stabilization, storm damage and ship groundings (Szmant 2002, Iannuzzi 1996, Chapman 1992). Pollutants, nutrients, and sediment are distributed throughout the marine environment by wind and wave action (Haynes et al. 2007). Possible sources of nutrients within harbours include accidental release of chemicals, discharge from untreated waste and terrestrial run-off of inorganic nitrate fertilizers (Chapman 1992).

Although no previous research has been conducted, it is possible that Harbour Village Marina on Bonaire, Dutch Caribbean, is increasing the level of nutrients and sediment in the surrounding waters and potentially degrading the coral reefs. The purpose of this study is to (1) determine the distribution of nutrients around the harbour entrance, (2) observe the effects proximity to a harbour has on coral reef health, and (3) determine if the concentration of nutrients and sediments is greater in the direction of current flow. Hypotheses that will be tested throughout this study are:

- H_1: Nutrient levels will be higher at sites north of the harbour due to the currents and wind.
- H_2: Nutrient levels will be higher within the harbour and will decrease with increased distance from the harbour.
- H_3: Coral reefs will exhibit fewer diseases and increased coral cover at sites farther away from the entrance to the harbour.

This study will create a better understanding of the effect harbours and marinas have on the coral reef ecosystem.

Materials and Methods

Study Site

All research was conducted during September and October in the waters surrounding Harbour Village on the leeward side of Bonaire, Dutch Caribbean (Fig. 1). Harbour Village (12° 09'46.31" N, 68° 17'09.43" W) is located 1700 m north of the capital Kralendijk. The harbour is 273 m long and 170 m wide. The site is also in close proximity to a number of popular dive sites and the fringing reef that runs along the leeward side of Bonaire. This site was chosen because it was more accessible than the other harbours on Bonaire. Data were collected along the coast from 300 m north of the harbor to 300 m south.

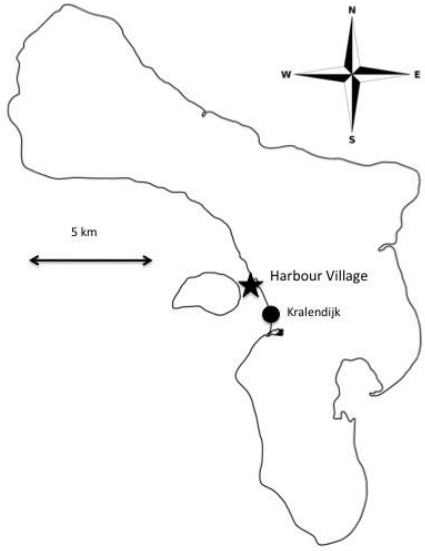

Fig. 1 Map of Bonaire, Dutch Caribbean. *Black star* indicates Harbour Village, research site. *Black circle* represents the capital, Kralendijk

Nutrients

In order to determine the distribution of nutrients from the harbour water samples were collected at 25 m intervals starting at the center of the harbour extending 300 m north and south. All water samples were collected in 50 ml bottles at a depth of 10 m. Water samples were kept cool until tests for the nutrients nitrate (NO_3^-) and nitrite (NO_2^-) could be conducted. Nitrate and nitrite levels were analyzed using the LaMotte Salt Water Aquaculture test kit.

Coral cover

Benthic cover was recorded with a series of 10 m line-intercept transects at 25 m distance increments from Harbour Village; two transects were laid at each distance. The first transect was laid at a depth of 10-12 m and the remaining transect was laid 2 meters deeper on the reef slope. Health of the reef was assessed using a modified AGRRA benthic method (Kramer et al. 2005). The species and size of living coral directly underneath the transect were recorded in centimeters. The size of macro algae directly under each transect was also

measured and presence of coral diseases recorded. The amount of live coral under each transect was used to determine the percent coral cover at each site.

Wind direction, surface current, bottom current, weather, and other environmental factors were noted at every data collection site and every time data were collected.

Data Analysis

Sites within close proximity to the harbour entrance were compared to sites that were farther away to determine the effects the harbour has on nutrient levels and coral cover. Regression analysis was used to test the association between (1) distance from the harbour and mean percent coral cover (2) distance from the harbour and nitrate levels (3) distance from the harbour and nitrite levels and (4) nutrient levels at each distance and mean percent coral cover. Means of percent coral cover were used within the analysis and standard deviations were calculated for all means.

Results

Nutrients

Preliminary current studies showed that both the surface and bottom currents primarily flow north. Results show that nitrite levels are consistently higher north of the harbour (Fig. 2). Nitrite levels from both the north and south ($F = 1.10$, $df = 1$, $p = 0.324$) had no significant relationship with distance from the harbour.

There was a significant increase in nitrate levels north of the harbour ($F = 9.38$, $df = 1$, $p = 0.015$) compared to south. Nitrate levels to the south were fairly constant (Fig. 3) and there was no significant decrease in levels at increasing distance from the harbour ($F = 0.86$, $df = 1$, $p = 0.381$).

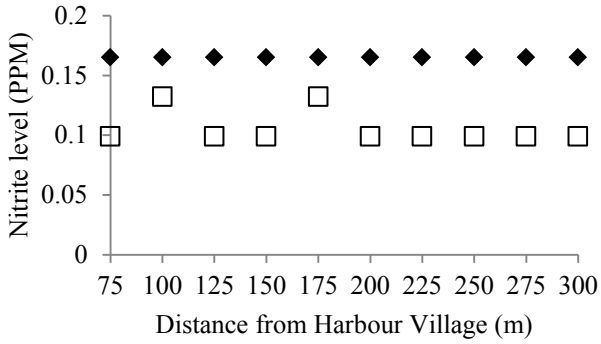

Fig. 2 Nitrite levels (ppm) at increasing distances from Harbour Village. *Black diamonds* indicates north of the harbour and *white squares* indicates south

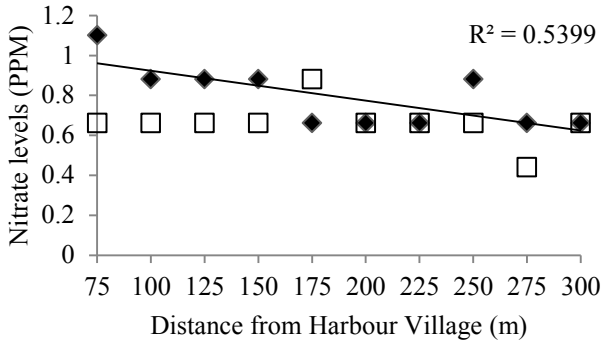

Fig. 3 Nitrate levels (ppm) at increasing distances from Harbour Village. *Black diamonds* indicates north of the harbour and *white squares* the south. *Black line* shows significant negative regression ($p < 0.05$)

Coral cover

The only disease that was present within the transects was yellow band disease, thought this was only recorded at two sites: 100m south of the harbour and 150 m south of the harbour.

Rubble and dead coral dominated the benthic substrate north of Harbour Village. A substantial amount of cyanobacteria was growing on the rubble and dead coral within 150 m of the harbour and turf algae growth from 150m to 300 m north of the harbour. There were insufficient data to determine the macro algae cover.

Percent coral coverage was substantially higher south of the harbour in comparison to the north (Fig. 4). Although percent coral coverage shows increasing trends with distance from the harbour the regression was never significant in either

the north (F = 2.57, df = 1, p = 0.148) or the south (F = 3.43, df = 1, p = 0.101).

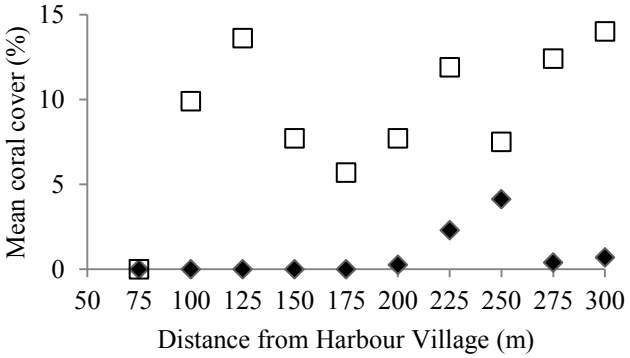

Fig. 4 Mean percent coral cover at increasing distances from Harbour Village. Means calculated from 2 replicate transects at each distance. *Black diamonds* indicates data from north of the harbour and *white squares* indicates south

Discussion

The hypothesis that there would be higher levels of nitrite and nitrate north of the harbour was supported. Although there was no significant decrease in nitrite levels at increasing distances from the harbour, nitrite levels were greater north of the harbour. Since there is no negative regression in relation to distance it is unclear if nitrite is sourcing from the harbour and distributed by currents. Nitrate levels did show a significant decrease with distance north of the harbour. These results were expected since preliminary research showed that current flows predominantly north. From this we can conclude that the current did distribute nitrate north of the harbour. The lower and primarily constant nitrate levels in the south are also to be expected since the current is not flowing in that direction and therefore it would not carry nutrients sourcing from the harbour. Although nutrient levels have been known to contribute to coral reef degradation (Szmant 2002) there was no significant relationship between nutrient levels and coral cover in the north and south due to the fact that coral cover did not show a regression with distance from the harbour. Although there was a significant increase in nutrient levels north of the harbour, there might be insufficient coral growth for the effects of increased nutrient levels to be detected.

It is likely that the very low percent coral cover north of the harbour is due to various other factors rather than increased nutrient levels from the harbour. Initial damage the reef likely occurred through dredging to create the harbour and possibly surrounding beaches and resorts and the corals have yet to recover. There is also speculation that hurricane Lenny played a role in the degradation of the reefs. Hurricane Lenny in 1999 is known to have affected some of the reefs north of the study area, but it is unlikely that it could be responsible for the lack of coral in the area north of the harbour (C. Eckrich, CIEE Research Station, Bonaire, pers. Comm.). It is also possible that the area has had historically low coral cover. Although it cannot be concluded from the results that nutrient levels played a role in determining coral cover, it is possible that in the increased nutrient levels in the north are a contributing factor to the low coral cover. A combination of these anthropogenic and natural causes is likely to explain the low coral cover north of Harbour Village that is seen today.

The only coral disease found in the transects, yellow band disease, was most probably not detected north of the harbour because there was insufficient coral coverage. None of the live corals under the transects were *Montastrea annularis*, the species of coral yellow band disease is commonly found on in Bonaire (Cervino et. al 2001). Since the disease was only observed under the transect on three occasions, there were not enough data to test for a correlation between disease and nutrients levels, or distance from the harbour. As percent coral cover does not show a significant increase with increasing distance, and nitrate levels do, it cannot be concluded that nitrate levels influence

coral cover. Although it can be concluded that currents play a major role in the distribution of nutrients, the same conclusion cannot be determined for percent coral cover. Nutrient levels were higher in the direction of current flow, and nitrate levels significantly decreased at increasing distances from the harbour. Results and observations show that the creation of the harbour itself most likely had the greatest impact on coral cover in the north. It is likely that disturbances following the initial creation of the harbour are relatively limited. Harbour Village appears to have few negative effects on the overall health of reefs in surrounding waters when considering nutrients.

The results from this research are beneficial to understanding distribution of nutrients from harbours and the impacts harbours have on reefs. Future studies might look into sedimentation rates and the prevalence of coral disease in relation to small harbours as well as the presence of cyanobacteria in dredged areas and its effects on coral recruit settlement. This research will help in efforts to restore reefs that have been dredged to create harbours and beaches. Continued research might also help limit the damage caused by dredging and other processes to create harbours.

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Fish density, species richness and diversity between stands of natural and artificial *Acropora cervicornis*

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Abstract
Coral reefs around the world are experiencing high levels of degradation due to temperature changes, increased nutrients and destructive fishing techniques. For example, where there were once large thickets of the branching coral *Acropora cervicornis* along the coasts of Bonaire, Dutch Caribbean, there is now sand and dead coral. Loss of an entire highly complex habitat has likely altered the local reef fish community. Artificial reefs have been used in the past to test hypotheses about structural complexity and its effects on reef fish communities. However, no studies have sought to discover if artificial reef structures modeled after *A. cervicornis* would support reef fishes found in the natural branching coral colonies by mimicking the structural complexity provided by the coral. To answer this question, four patches of artificial *A. cervicornis* were constructed and placed near the reef crest on the leeward side of Bonaire, Dutch Caribbean. Natural thickets of *A. cervicornis* were used to compare differences in fish species richness, diversity and density. Artificial reef structures were found to support higher diversity but lower abundance of fish. There was no significant difference in species richness between the natural and artificial reef stands. Overall, the artificial reef structures were able to provide some shelter to certain fish species, but were not able to support the fish community that is supported by natural stands of *A. cervicornis*.

Introduction

Due to climate change, eutrophication and overfishing, coral reefs around the world are experiencing declines and need intervention in order to survive (Pandolfi et al. 2003). Changes in environmental policy will take time to implement, and until then overfishing and ocean pollution will continue to occur. One part of the solution may be to temporarily supplement the reefs in order to reduce reef stress; for example, artificial reefs that simulate coral reefs may provide structure and habitat for recruitment of benthic invertebrates and reef fish (Ambrose and Swarbrick 1989).

Artificial reefs have been used to supplement some fisheries, and to provide a mechanism to assess the condition of these fisheries (Yoshida et al. 2010). A variety of materials have been used in the fabrication of artificial reefs, including glass bottles, concrete blocks, tires and tree branches (Gratwicke and Speight 2005; Fitzhardinge and Bailey-Brock 1989; Ambrose and Swarbrick 1989, Stone et al. 1979).

Several studies have determined that artificial reefs support a higher diversity and number of species than natural reefs (Ambrose and Swarbrick 1989; Walton 1982; Turner et al. 1969). Ambrose and Swarbrick (1989) sampled 10 artificial and 16 natural reefs in southern California and found that the artificial reefs had higher species diversity than the natural reefs, mostly due to the diversity of benthic fishes that were found near the artificial structures. Stone et al. (1979) found that artificial reefs and natural reefs in the Florida Keys supported similar species diversity and abundance, although the artificial reefs were in close proximity to the natural reefs. In contrast, other studies have found that artificial reefs support lower species richness than natural reefs (Fast and Pagan 1974; Carr and Hixon 1997). Carr and Hixon (1997) compared fish abundance and species richness of 16 natural and 16

artificial reefs in the Bahamas and found that artificial reefs were less diverse with fewer fish. Although results differ among studies, all show that artificial reefs can support fish species.

The island of Bonaire in the southern Caribbean has experienced losses in overall coral cover since 1988 (Stokes 2010). It is clear that in many areas the degradation of coral is severe, but perhaps the most obvious example is found in the near-shore areas, previously occupied by large thickets of *Acropora cervicornis*. Van Duyl (1985) mapped the types and location of corals around Bonaire. In the past 30 years *A. cervicornis* has been drastically reduced due, in large part, to white band disease (Bruno et al. 2007) and hurricanes (C. Eckrich pers comm). Habitats that were formerly composed of highly complex *A. cervicornis* are now mostly sand, which provides minimal structure to the reef fish that would normally use the complex branches of *A. cervicornis* as shelter. An additional problem associated with the loss of the fast-growing *A. cervicornis* is the decrease in the framework it provides for future reefs (Aronson and Precht 2001).

The purpose of this study is to determine if artificial reef structures are an effective way to support the fish populations that once depended on the shallow thickets of *A. cervicornis* by comparing the density, species richness and diversity of fish observed around artificial and natural reefs. Artificial reefs may prove to be an effective temporary solution to habitat loss by providing habitat for fish in the place of degraded natural reefs. The usefulness of implementing artificial reefs in Bonaire as substitutes for natural *A. cervicornis* were evaluated in this study by testing the following hypotheses:

- H_1: The artificial substitute for *A. cervicornis* will support higher densities of fish when compared to natural *A. cervicornis*.
- H_2: The artificial substitute for *A. cervicornis* will support higher species richness of fish when compared to natural *A. cervicornis*.
- H_3: The artificial substitute for *A. cervicornis* will support a higher diversity of species when compared to natural *A. cervicornis*.

While many studies have been done on the effectiveness of artificial reefs, none have specifically targeted Bonaire, or thickets of *A. cervicornis*. *A. cervicornis* is an endangered species with little in the way of research on its role as a habitat. A better understanding of this role may help in limiting the impact from the loss of *A. cervicornis*.

Materials and Methods

Study site

Bonaire is an island in the southern Caribbean (Fig. 1). The island's reefs are classified as fringing reefs and most dive sites are easily accessed by shore entries. Many of the reefs are dominated by *Montastraea* and *Diploria* species. Most of the near shore areas are composed of sand with a few corals before the reef crest begins at a depth of 4 to 5 m.

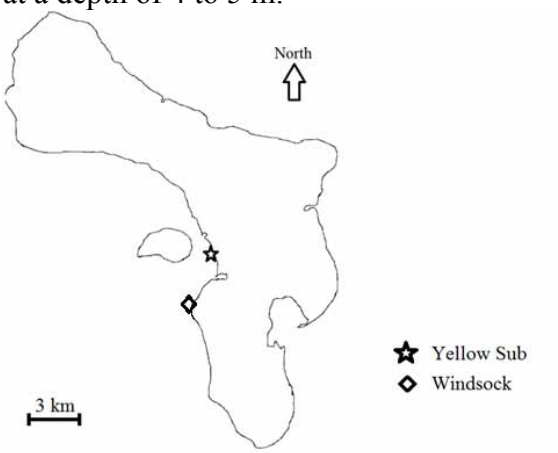


Fig. 1 Map of study sites Yellow Sub and Windsock on Bonaire, Dutch Caribbean

Several thickets at the dive site Windsock were used as the study area for the natural coral. The site is characterized by a sandy bottom that is followed by reef at about 6 m of depth. The natural *A. cervicornis* consists of several patchy

thickets at a distance of 1 m to 3 m between each thicket.

The artificial reefs were placed at the Yellow Sub dive site. This area has a few corals and mooring blocks sparsely scattered until it reaches approximately 4 m deep, where the fore-reef begins. To the east of the reef, most of the substrate is sand. This area is ideal as it was previously covered in thickets of *A. cervicornis* (van Duyl 1985).

Construction

To create a habitat similar to an *A. cervicornis* thicket, 30 concrete blocks (0.4 m x 0.2 m) with eight holes (7 cm x 7 cm) were used as a base to insert tree branches. A 15 cm PVC pipe tube was placed in opposite corners to anchor the blocks to the substrate (Fig. 2). Each cement block contained one tree branch per hole. Branches were held in place by rocks and mortar and allowed to dry. The blocks were placed in 6 m to 7 m of water in front of the Yellow Sub dive site, about 3 m away from the natural reef. The artificial reef thickets were arranged to mimic patches of the natural *A. cervicornis* at the Windsock dive site (Fig. 3). Nine blocks were placed in a 3 x 3 arrangement in an area of 1.7 m x 1.4 m to create three patches. The remaining three blocks were placed to create three patches that were each 1.7 m x 1.5 m. Rebar was used to anchor the block in place.

Observation

Ten survey dives (five on artificial and five on natural *A. cervicornis* beds) were completed over a period of two and a half weeks. In order to determine the abundance and types of fish species found in natural *A. cervicornis,* three patches of *A. cervicornis* (dimensions: 1.9 m x 1.4 m, 1.5 m x 1.4 m, 1.8 m x 1.7 m) and an additional area (1.6 m x 1.5 m) with a more patchy distribution of *A. cervicornis* were surveyed (Fig. 3). Each of these areas was assigned a number and the order in which the patches were surveyed was determined randomly before every dive. At the beginning of the survey period the observers stayed 4 to 5 m away from the study area and a 2 min observational period took place where all species observed within 0.5 m of any of the coral patches and the abundance of each fish species was recorded. Then, two divers, positioned on either side of an *A. cervicornis* patch, recorded the species and abundances that were observed for 3 min. A 1 min survey of the base of the *A. cervicornis* patch was then completed to record any small, benthic fish species. The survey process was repeated for the remaining natural thickets, and the four artificial blocks.

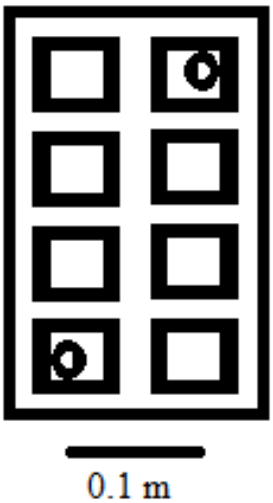


Fig. 2 Diagram of a cement block (0.4 m x 0.2 m) before branches were added. *Circles* represent 15 cm PVC pipes that were used to anchor the artificial reefs to the substrate with rebar. *Squares* represent the holes (7 cm x 7 cm) that branches were placed into using mortar

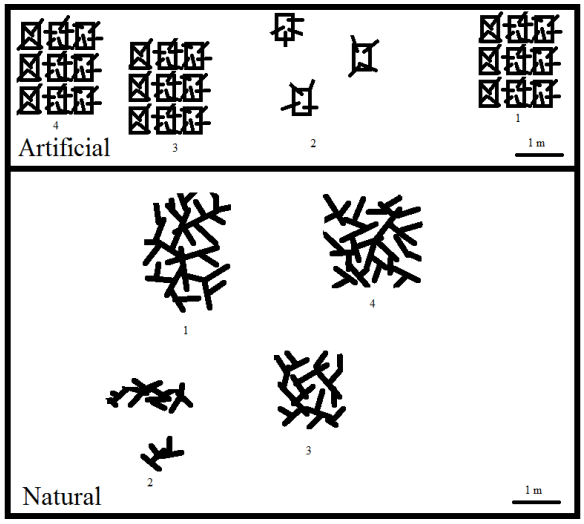


Fig. 3 Diagram of the distribution of the natural and artificial *Acropora cervicornis* stands. The *numbers* beneath each thicket signify the number assigned to that thicket to determine the random order of surveying. The *squares with lines* represent the artificial blocks and the *large lines* represent the natural *A. cervicornis*

Data Analysis

Fish density, species richness and species diversity were calculated to compare the natural *A. cervicornis* bed at Windsock to the artificial bed at Yellow Sub. Species density was calculated by dividing the total number of all fish seen during a survey by the total area of all sections of the habitat surveyed (10.22 m^2 artificial, 9.69 m^2 natural). These numbers were compared using a Mann-Whitney U test. The mean species richness, the number of species observed during each survey, from the natural and artificial groups was also compared using a Mann-Whitney U test. Species diversity of each group was calculated using the Shannon-Weaver Index, $H = -\sum_{i=1}^{s}(p_i \, ln \, p_i)$, where p_i was the frequency of a species. The closer to 1 the resulting number is, the higher the diversity of the fish population. The results from the Shannon-Weaver Index were compared using a Mann-Whitney U test. The Mann-Whitney U tests were run because the normality assumption was not met for a parametric test.

Results

A total of 3315 fish from 46 species were observed between the two habitat types during 10 hours of observation. The relative abundances of certain species varied between the natural and artificial reefs (Fig. 4). For example, 83 pomacentrids and 260 scarids were observed at the artificial site while 1395 pomacentrids and 46 scarids were observed at the natural site. The number of gobiids by the artificial structures (504) was notably higher than at the natural site (157). The natural site also supported a large school of juvenile *Thalassoma bifasciatum*, which led to an inconsistent count between some the surveys if the school was not by the thicket at the time of the survey. School fish were not seen at the artificial reef except for in the case of juvenile scarids. Juvenile and initial phase scarids in groups as large as 14 individuals were commonly observed feeding on the artificial structures as algae began to grow on the branches. Pomacentrids and acanthurids were also observed feeding on the branches.

Fish Density

The natural structures had a significantly higher mean fish density (42.36 ± 1.27 ind m^{-2}) than the artificial structures (23.74 ± 4.08 ind m^{-2}; U = 0.00, df = 9, $p < 0.01$; Fig. 5a). The fish density of the natural *A cervicornis* structures was nearly double that of the artificial reef structures.

Species Richness

There was no significant difference between the mean species richness of the natural structures (23.80 ± 2.58) and the artificial structures (23.20 ± 2.68; U = 10.00, df = 9, p = 0.59; Fig. 5b).

Species Diversity

The mean Shannon-Weaver Index was significantly lower (0.62 ± 0.01) for the

natural reef than the artificial *A. cervicornis* (0.71 ± 0.04; U = 0.00, df = 9, p < 0.01; Fig. 5c).

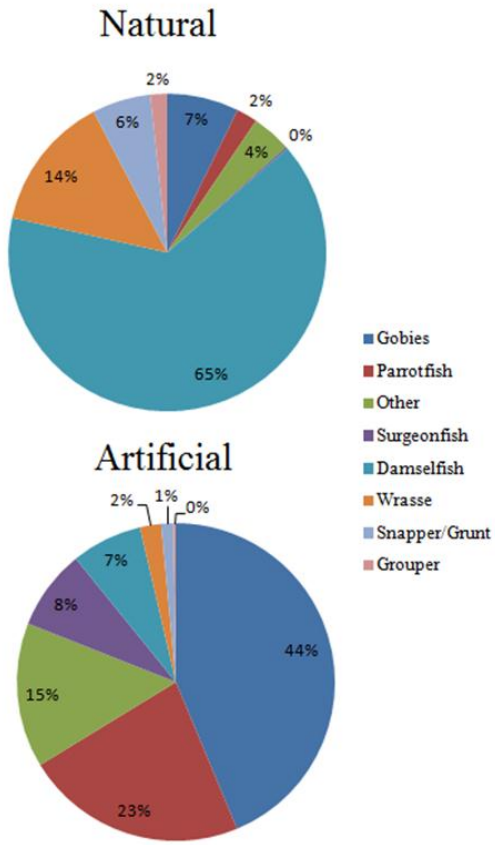
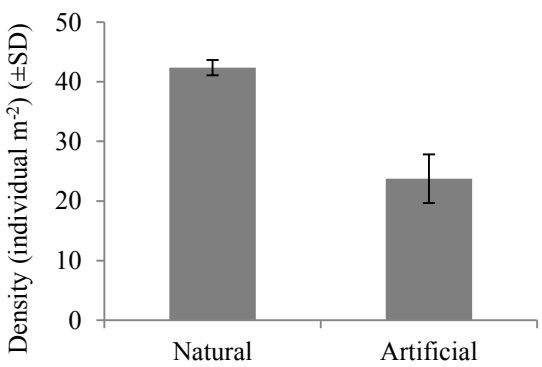
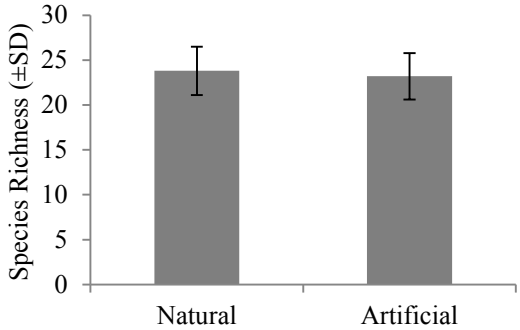


Fig. 4 Relative percentages of the common groups of fish seen at the artificial and natural sites for all observations

a.



b.



c.

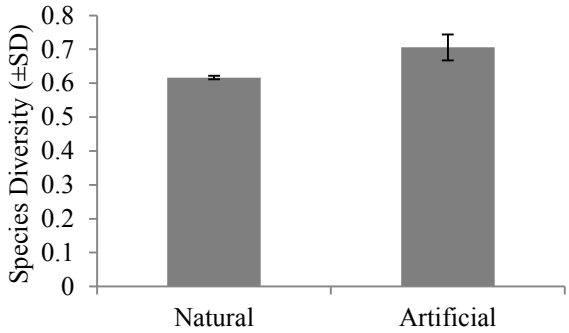


Fig. 5 Comparison of **a**. mean fish densities (±SD), **b**. mean species richness (±SD) and **c**. mean species diversity (using the Shannon-Weaver Index; ±SD) between a natural and an artificial *Acropora cervicornis* thicket (n = 5)

Discussion

The first hypothesis, that artificial reefs would support higher densities of fish, was not supported because the natural structures had a higher density of fish species than the artificial reef structures. While this result was not initially expected, it became clear after observing the natural reef that it was more complex than the artificial structures used for the experiment. Complexity has been shown to increase species richness, abundance, density and biomass in the Mediterranean and in Florida, USA (Charbonnel et al. 2002; Sherman et al. 2002), which may explain the increased density of fish on natural *A. cervicornis* reefs.

There was no difference in species richness between the two reef types as was predicted in the second hypothesis. This

result may be explained by the concept of ecological limits (Rabosky 2009). This theory suggests that the number of species that occur in an area or habitat is restricted by the resources in that area. With this in mind, it is likely that a limited number of species utilize the habitat provided by branching corals (such as *A. cervicornis*) and those species are similar between the artificial and natural reef structures due to the proximity of the two sites.

The third hypothesis predicted that artificial reefs would have higher species diversity than the natural reef. There was a higher mean diversity at the artificial site than the natural site. The natural reef had been in place longer than the artificial reef. Placing the artificial reef created a disturbance, but after it was set up, new fish began to aggregate at the site. Over the relatively short two and a half week period, it is unlikely that the dominant competitors would be able to outcompete the other fish that utilized the structures. In contrast, the dominant fish at the natural reef had years to outcompete other species; leading to a lower diversity. Artificial reefs in the Red Sea were quickly colonized, and the increase did not level off until seven months later (Golani and Diamant 1999). The artificial structures in this study may have needed more time to get a more accurate picture of the fish assemblage there since the surveys only took place over two and a half weeks. Golani and Diamont (1999) also found that the low structural complexity of an artificial reef contributed a low settlement of small fish species. This may explain the lack of pomacentrids on the artificial reef.

The results of this study conflict with those of previous artificial reef studies (Ambrose and Swarbrick 1989; Walton 1982; Turner et al. 1969). One of the contributing factors to this difference could have been the territorial actions of the pomacentrids that were abundant at the natural reef and were less abundant on the artificial structures. Complexity may have played a role in recruiting the large numbers of pomacentrids to the natural reef, but the interactions between the pomacentrids and the other fish species may have played a role in the lower species diversity. Many pomacentrids are known to exclude other fish from their territory in order to protect algae beds used for food (Itzkowitz 1977). Attacks by pomacentrids were observed several times during every survey and the large number of pomacentrids may have influenced the species richness and diversity because of this aggression. Gobiids may have been one of the more affected groups as the number of gobiids decreased from 504 individuals at the artificial site to 157 at the natural site. Attacks by pomacentrids on gobies were seen frequently with the gobiids moving away every time. However, Ambrose and Swarbrick (1989) noticed that artificial reefs attracted more benthic fish species, which could mean that the gobiids were attracted to the artificial reef more than the natural reef structure. This difference was thought to occur because the artificial structures were set on a sandy substrate, which caused a large number of fish to seek out the limited shelter that became available. This may be due to the lower density of other fish at the artificial site, or due to reasons that could not be determined in this study.

In order to address temporal limitations, future studies should observe artificial structures for at least seven months to allow species richness and abundance to stop increasing (Golani and Diamant 1999). Placing the artificial reef at the same site as the natural reef would also reduce variability in the two sites. If a higher number of natural sites could be located, additional comparisons would be beneficial as well. Larger artificial structures that have extra complexity, especially around the base of the structure, would also replicate the natural reef more closely. This could be accomplished by adding more branches to a structure similar to the one used in this study. Other materials could also be tested to see if a more suitable substitute for *A. cervicornis* could be found.

Overall, this study found that artificial *A. cervicornis* supported higher fish diversity and lower fish density than natural *A. cervicornis*. There was no significant difference in species richness between the two habitats. While an artificial reef may provide some habitat and some of the ecological functions of *A. cervicornis*, it is difficult to create a structure that represents the natural coral due to its high complexity. However, it is unknown whether populations of *A. cervicornis* will recover in the near future, and artificial reefs may play a part in helping some marine ecosystems endure. More work must be done to ensure that the artificial reefs are as effective as possible at replicating natural reefs.

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Response of three predatory reef fish species to fishing protection in Bonaire, Dutch Caribbean

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Abstract

Fish protected areas are increasingly utilized as a means of bolstering existing fish stocks while laying the foundation for healthy future populations. Such areas can promote species abundance, density, total biomass, and fecundity. This project set out to study the effects of FPAs on three reef predators, the graysby (*Cephalopholis cruentata*), the coney (*Cephalopholis fulva)* and the great barracuda (*Sphyraena barracuda)*. *S. barracuda* has a substantially greater range (range of median distance traveled between 0.5 and 8 km) than either of *C. cruentata* or *C. fulva,* and has complex range tendencies that include site fidelity along with spawning migrations. Since home range size and response to fish fishing protection are correlated, the effects of FPAs on *S. barracuda* populations may differ in relation to the effects felt by *C. fulva* and *C. cruentata* populations. Using visual sampling methods inside and outside of Bonaire's FPAs, sightings per unit effort were recorded for all three species to compare how the predators might respond to the establishment of FPAs. The results of this study indicate that *S. barracuda* has a greater response to protection than either *C. fulva* or *C. cruentata.* All species showed increasing total length trends going from un-protected to protected areas. Yet, neither species showed a significant difference in abundance between the two areas. This may be an indication that, at present, Bonaire's FPAs are not effectively protecting *S. barracuda, C. fulva,* or *C. cruentata*.

Introduction

As human populations have increased globally, fish populations have been pushed towards ecological extinction. Marine reserves and no-fishing zones are increasingly used as a means to boost fish stocks (Halpern and Warner 2002), and have been shown to increase population density, mean size, and species diversity within protected areas (Roberts et al. 2001; Palumbi 2004). No take reserves may protect important spawning sites, and in doing so enhance a fishery by protecting the critical reproductive habitat of the target population (Ashworth and Ormond 2005; Roberts et. al 2001). In the decades to come, trends in human consumption will undoubtedly necessitate the formation of more fish protected areas (FPAs).

On the island of Bonaire, Dutch Caribbean, two fish protected zones were recently implemented (STINAPA 2007). These two areas, enforced since 2008, restrict all reef fishing in water to 60 m (STINAPA 2007). Part of the impetus to form these two FPAs came from a report by Couture et al. (2004), which noted that local fishermen and divers had witnessed a decline in size and abundance of predatory reef fish. The study advocated the institution of protected areas as a socio-economic and ecological remedy for this problem. In the years since the reserves were put in place, piscivorious fish populations have yet to be surveyed intensively.

Predatory fish, especially those found on reefs, are known to show the greatest benefit from no-take reserves (Palumbi 2004). This group is often the most heavily targeted by fishing pressure and therefore respond the strongest when pressure ceases (Chiappone et. al 2000; Palumbi 2004). These findings may be of particular relevance to managers of fisheries containing great barracuda (*Sphyraena barracuda)*. In areas of the Caribbean where

Ciguatera toxicity is not prevalent, *S. barracuda* is a valuable food source (Kadison et. al 2010). Ecologically, *S. barracuda* is a top predator; a trophic position positively correlated with reef health and increased total reef-fish biomass (Stevenson et. al 2006; Kadison et. al. 2010). With these qualities in mind, protecting *S. barracuda* from overexploitation is in the interest of all stakeholders in coral reef systems.

Like *S. barracuda,* grouper are predatory fish prized by fisherman in the Caribbean and beyond (Chiappone et. al 2000; Trott 2006). Within the grouper family Serranidae there is variability in size at maturity ranging from 45 cm to over 90 cm (Chiappone et. al 2000). These size differences may account for discrepancies in fishing pressure and, consequently, abundance. In numerous Caribbean localities, larger grouper have been fished to extirpation (Trott 2006). Smaller grouper such as the graysby (*Cephalopholis cruentata)* and coney (*Cephalopholis fulva)* may fill the niche created by the absence of larger Serranids and may come to represent the majority of grouper biomass (Chiappone et. al 2000; Trott 1996). As an outcome, *Cephalopholis spp* grouper may receive fishing pressure as fishermen target smaller and smaller species.

Palumbi (2004) found that marine species' home range size, and their response to protection within FPAs are positively correlated. Coastal fish species with home range lengths > 100m benefitted the most from protection, reflecting increased fishing pressure on large and hence more mobile species. Although *S. barracuda* is a species known to exhibit site fidelity within several kilometer study sites (Wilson et al. 2006; O'Toole et al. 2011), including long-term site fidelity for *S. barracuda* >70 cm; evidence was also found indicating that the species may embark upon long seasonal spawning migrations. Taking into account the lack of research that exists on the range behavior of *S. barracuda*, it remains to be seen whether *S. barracuda* can benefit from relatively small (1-2 km), localized FPAs such as those that have recently been implemented in Bonaire.

In contrast to the barracuda, grouper in the genus *Cephalopholis* have been found to have small average home range lengths (46.03 m) (Popple and Hunte 2005). Based upon their small home range and status as less targeted species, *C. cruentata* and *C. fulva* are ideal candidates to compare to *S. barracuda*, which may have a larger home range and receive considerably more fishing pressure. To date, little data exists on the predatory fish populations surrounding Bonaire. However, *S. barracuda* are known to be an important food source on the island, and *C. cruentata and C. fulva* are regularly caught by fishermen (R. Peachey, CIEE Research Station, Bonaire, pers comm; pers obs 2011). This study will provide valuable information on how *S. barracuda, C. cruentata,* and *C. fulva* populations are affected by protection within Bonaire's two FPAs. The aim of this study is to test the following hypotheses:

- H_1: *S. barracuda, C. cruentata, and C. fulva* abundances will be greater inside the two FPAs as opposed to the two fished areas (FAs) as a result of reduced fishing pressure.
- H_2: All three species will show a greater percentage of large size classes inside the two FPAs because individuals have been able to grow larger in the absence of fishing pressure.
- H_3: *S. barracuda* will exhibit a more positive response to FPA protection than *C. cruentata* or *C. fulva* due to higher commercial importance and thus greater fishing pressure on *S. barracuda* within the FAs.

Materials and Methods

Study Site

The study site was comprised of two marine reserves west of Kralendijk, Bonaire, and two corresponding FA zones adjacent to

each reserve (Fig. 1). The northern area, FPA 1 (start: 12°10'47.78" N ,68°17'37.99" W) extended south 2.134 km to the mouth of the Harbor Village marina. The northern FA began at the mouth of the marina and also extended 2.134 km south. The second FPA, adjacent to the town pier (start: 12°8'45.26" N , 68°16'36.47"W) extends 1.19 km south to the mouth of the Plaza resort harbor. The corresponding fished area was also 1.19 km and was situated immediately south of the southern FPA. Both FPAs provide protection to a depth of 60 m. For the purposes of this study, FPA 1 excluded the northernmost 400 m of the STINAPA-enforced marine reserve. This decision was made to ensure that the each reserve and correlated reference zone had the same dimensions. All study sites were situated in areas containing fore-reef and shallow sand habitat.

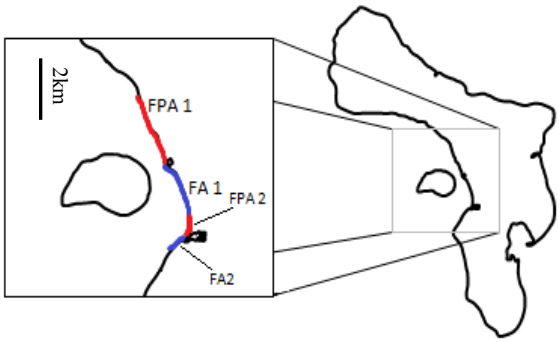

Fig. 1 Map of the four study sites (FPA1, FA1, FPA2, FA2) on the leeward coast of Bonaire, DC

Data Collection

Several methods of visual survey were utilized both underwater and from the surface. During all surveys, barracuda, graysby, and coney size estimates were recorded. Prior to the visual surveys, observers in this study were trained to estimate *S. barracuda, C. cruentata,* and *C. fulva* length underwater as this has been shown to increase sampling accuracy (Ashworth and Ormond 2005). All observed *S. barracuda* were grouped into categories based on length: small (≤ 30 cm), medium (30-60 cm) and large (> 60 cm). *C. cruentata* and *C. fulva* size categories were small (≤ 10 cm), medium (10 - 20 cm), and large (> 20 cm).

Snorkeling was chosen because it provided a quick survey method that would not compromise sampling accuracy (Ashworth and Ormond 2005). Snorkeling transects were 100 m long and conducted on the reef flat between 1 and 8 m deep, with depth assigned randomly prior to sampling. Snorkeling transects were conducted at randomly chosen locations within the FPAs and FAs. During each snorkel, a reel with 100 m of weighted line was used to standardize survey length. Any *S. barracuda* within sight were recorded and classified into one of the three length categories. Any *C. fulva* observed in the first 100 m pass were counted and grouped into length categories. At the end of each surface survey, total time was noted for the compilation of sightings per unit effort (SPUE) data.

Several fish abundance studies have indicated stationary point-counts and belt transects as equally accurate underwater methods for visually surveying coral reef predators (Micheli et al. 2004; Carlos and Samoilys 2000). A combination of both methods was utilized for this study. As in the surface surveys, a reel and weight with 100 m of line was used to standardize the underwater transects while using SCUBA. Care was taken to swim at a consistent rate of 15 min transect^{-1}. Point count survey time was 10 min based on a study by Ashworth and Ormond (2005), which found 10 min intervals to be the optimal time for reef-predator surveys. SCUBA surveys began with a 10 min *S. barracuda* point count at a randomly chosen depth between 18 m and 8 m followed by a *S. barracuda* and a *C. cruentata* survey that followed a contour at the same depth. As with the surface swims, direction alternated between northerly and southerly headings from the randomly assigned survey sites. A total of eight underwater surveys and 32 surface surveys were conducted for this study.

Data Analysis

The mean *S. barracuda*, *C. cruentata and C. fulva* SPUE within FPAs and FAs was compared using a t-test (α =0.05). As a means of comparing the species' response to protection within FPAs, FPA sighting frequency to FA sighting frequency ratios were calculated for *S. barracuda* and *C. cruentata* and *C. fulva* combined. Higher ratios indicated a proportionally greater response to protection. Size estimate data for the three species was reviewed using a histogram for FPAs and FAs.

Results

Over the course of the study, snorkeling and SCUBA surveys covered more than 8000 m of reef habitat over a 12 h period. Seventeen *S. barracuda* and a combined total of 117 *C. cruentata* and *C. fulva* were recorded. Estimated water visibility ranged from 7 m to 30 m (mean = 20 m).

Abundance

Mean barracuda sightings min^{-1} for FPA sites (0.04 sightings min^{-1} ± 0.04 (±SD) and for FA sites (0.01 sightings min^{-1} ± 0.03), were not significantly different from one another (t = 1.53, df = 39 , p = 0.07)(Fig. 2). The number of Barracuda sighted within survey dimensions for fished areas (5 fish) and for fish protected areas (12 fish) produced an average sighting frequency of 0.024 sightings min^{-1} ± 0.04 (±SD) for all survey locations.

C. cruentata and *C. fulva* combined mean SPUE was found to be 1.03 sightings min^{-1} ± 0.72 (±SD) in FAs, and 0.91 sightings min^{-1} ± 0.64 in FPAs (Fig. 3). Mean sighting frequency of *C. cruentata* and *C. fulva* was not statistically different between fished and unfished regions (t = -0.53, df = 17, p = 0.7).

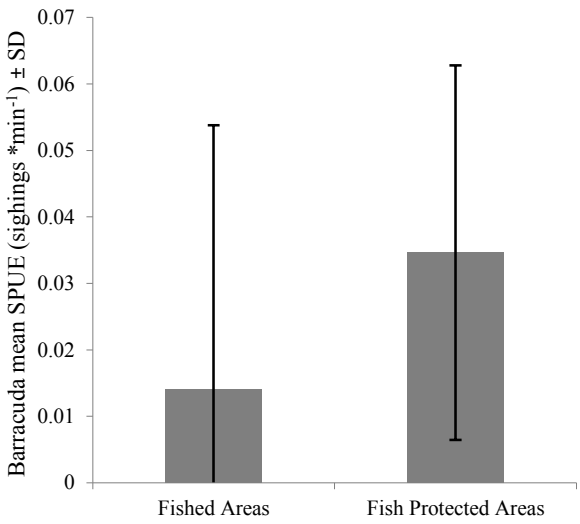

Fig. 2 *Sphyraena barracuda* average sighting frequency (± SD) within FPAs and FAs on the leeward side of Bonaire, DC (n = 40, p = 0.06668)

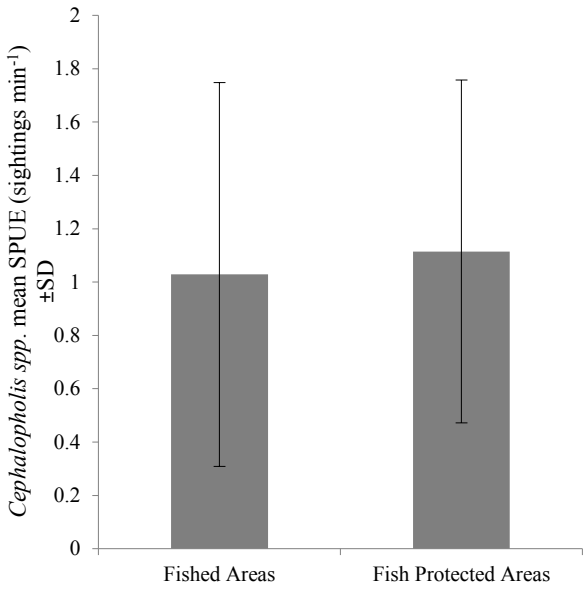

Fig. 3 Combined sighting frequency (sightings min^{-1}) (±SD) for *C. cruentata* and *C.* fulva inside the two FAs and two FPAs along the leeward side of Bonaire (n = 18, p = 0.7932)

Length estimates

Over the course of 20 surveys, there were no sightings of barracuda less than 30 cm in FPA regions (Fig. 4). Whereas 7 individuals were between 30 and 60 cm total length, and 5 individuals were greater than 60 cm total length (Fig. 4). In the FPA regions, 20 graysbys and coneys were less than 10 cm,

37 individuals were between 10 and 20 cm, and 13 individuals were greater than 20 cm.

Survey locations within FAs showed a different assortment of size estimates (Fig. 5). There were zero sightings of *S. barracuda* longer than 60 cm, four sightings of *S. barracuda* between 30 and 60 cm, and one barracuda less than 30 cm (Fig. 4). There were 37 *Cephalopholis spp* less than 10 cm total length, 26 individuals between 10 and 20 cm, and 6 greater than 20 cm total length (Fig. 5).

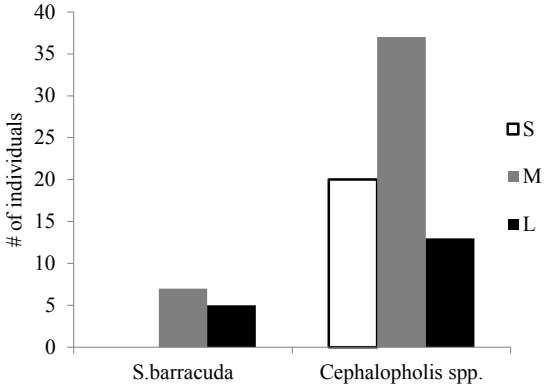

Fig. 4 Number of FPA individuals in each size class (*n = 40). S. barracuda S (*≤ 30 cm), M (30 - 60 cm), and L (> 60 cm) size classes shown. *Cephalopholis spp* size classes include S(≤ 10 cm), M (10 – 20 cm), and L (>20cm)

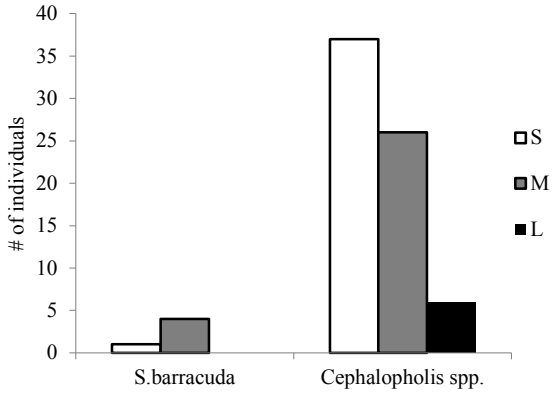

Fig. 5 Number of FA individuals in each size class(*n = 40). S. barracuda S (*≤ 30 cm), M (30 - 60 cm), and L (> 60 cm) size classes shown. *Cephalopholis spp* size classes include S(≤ 10 cm), M (10 - 20cm), and L (> 20cm)

Response to protection ratios

When a fish protected area SPUE to fished area SPUE ratio is compared between the species, *S. barracuda* exhibits a higher response to protection ratio (FPA sighting freq/ FA sighting freq=2.46), compared to the response to protection ratio for graysbys and coneys (0.89)(Fig. 6).

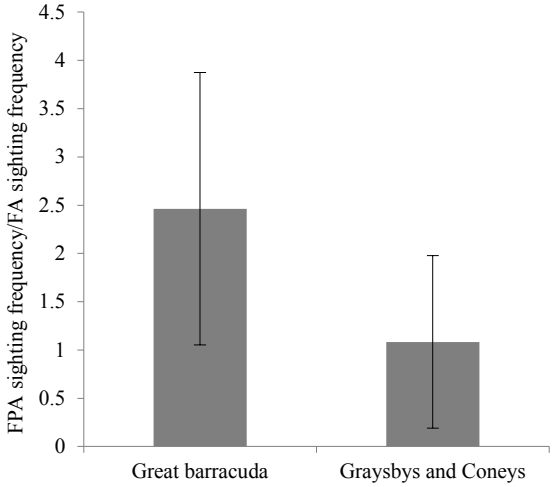

Fig. 6 Ratio of FPA sightings min^{-1} to FA sightings min^{-1} for *S. barracuda* and *Cephalopholis spp.* (graysbys and coneys) ± SD. Higher ratios equate to a higher proportion of sightings within the FPAs

Discussion

No significant difference in abundance was found between FAs and FPAs for any of the three species. Therefore the first hypothesis stating that all species will have higher sighting frequencies in the FPAs must be rejected. This is likely a consequence of several factors. First, the FPAs may not have been around long enough to allow fish populations to rebound. Second, in the case of *S. barracuda* the FPAs may be too small to adequately protect the species from fishing pressure considering the species' propensity for long seasonal migrations and variable range behavior (O'Toole et. al 2011). A third possibility is that some fishing is occurring within the protected areas due to a lack of enforcement. This has the potential to affect all three populations, although the severity of this pressure is unknown at present.

In the FPAs, there was a higher percentage of large barracuda observed compared to fished areas where zero large individuals were seen. These findings may

indicate lessened fishing pressure on large individuals within the FPAs. Larger fish within a population are the most targeted by fisherman and are often rare by consequence (Chiappone et. al 2000). Another possible conclusion is that larger barracuda are showing a higher degree of site specificity as noted by Wilson et. al (2006). This could result in a higher proportional abundance of large barracuda within FPAs, as the larger barracuda stay within the FPA boundaries while smaller individuals roam out.

Considering the data from *C. cruentata* and *C. fulva*, the same trend is apparent. Going from FPA regions to FA regions, the population dynamics appear to shift from a population with the highest percentage of individuals being mid-sized to a population where small sized individuals are the most prevalent. In the presence of fishing pressure, percentages of both mid-sized and large fish decrease while the percentage of small sized fish nearly doubles. This is a prime example of how consistent fishing pressure can affect the structure of a population. Based upon these data trends, the second hypothesis stating that there will be more large-class fish in the FPAs cannot be rejected.

Although differences in abundance between fished areas and fish protected areas were not found to be significant for all three species, there appears to be a trend in the response to protection ratios indicating that barracuda may have responded more positively to protection within FPAs than graysbys and coneys. Considering these findings, the last hypothesis stating that *S. barracuda* would show the greatest response to protection cannot be rejected. This result is most likely because of decreased fishing pressure on great barracuda within the FPAs. These findings support the conclusion drawn by Palumbi 2004, which stated that larger, more mobile reef fish will respond better to protection as compared to smaller more sedentary reef-fish species.

Despite the fact visual counts are an effective and ethical survey method, visibility is often a constraint. For surveys of mobile and relatively scarce fish such as barracuda, clear water is a necessity. On several days of surveying visibility was less than 10 m, possibly skewing the results. Further visual surveys of *S. barracuda*, *C. cruentata* and *C. fulva* may want to set a minimum visibility limit and only conduct surveys on days when visibility is above that limit. Another feature of this study that may have resulted in error were shallow water surface surveys. Shallow water habitats along the leeward coast vary, possibly affecting abundance data for coneys. Additionally, shallow water surveys and shallow point counts seemed generally ineffective in surveying barracuda; the vast majority of individuals were seen in water deeper than 8 m.

In the years to come, more research is needed on the reef-fisheries of Bonaire. At the present time the level of enforcement within the two FPAs is unknown and should be quantified by an unbiased third party.

Furthermore, to effectively manage *S. barracuda* populations, conclusive range data needs to be established for the species. This research will help in the development of methods to protect *S. barracuda* during future years of increasing fishing pressure.

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Sponge abundance and the prevalence of disease on scleractinian coral in Bonaire, Dutch Caribbean

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Abstract
Sponges are important to reef health, fulfilling a number of structural and functional roles. Despite the wide range of functions they provide, limited research on sponges has been conducted. Their ability to filter the water column of nutrients is essential to keeping the water around reefs clean. Coral reef diseases are becoming more widespread due to multiple factors such as pollution and eutrophication. This study assessed the possible relationship between sponge abundance and disease prevalence in Bonaire, Dutch Caribbean, by using 10 m x 2 m belt transects photographed into 1 m x 1 m quadrats. Per transect, 10 randomly selected photographs were analyzed using Image J to find the total area of sponges and live and diseased coral. Less than 1% disease was found at the study sites because the majority of the coral in the quadrats was partially or completely dead. No correlation was found between disease and sponge coverage. However, a significant correlation was found between percent sponges and live coral coverage. Although there was no correlation found between sponge cover and disease prevalence, the relationship between live coral and sponge is of particular importance for reef health monitoring across the Caribbean.

Introduction

Coral reefs provide ecosystem services such as food, income, and coastal protection for millions of people worldwide (Powell et al. 2010). Additionally, the biodiversity found on and around coral reefs is unmatched in the marine environment and can only be compared to tropical rainforests in terms of species diversity. Sponges represent an integral part of this biodiversity, with approximately 200 different species of sponges found on Bonaire alone (Diaz and Rützler 2001). Sponges are important to reefs because of the number of structural and functional components they provide (Bell 2008), including but not limited to: stabilizing corals, serving as food for spongivores, increasing the biomass of coral reefs, providing shelter to certain species and filtering the surrounding water of particles and nutrients not consumed by other organisms (Diaz and Rützler 2001; Wulff 2001). However, because identification of sponges in the field is difficult, they are often overlooked and not as extensively studied as corals and reef fish (Wulff 2001). The most accurate way to distinguish between sponge species that are similar in size, shape and color is through microscopic analysis (Diaz and Rützler 2001). This is much more time consuming than studying corals and reef fish because the latter can be differentiated based on simple pictures and a few descriptive facts.

Nutrients are always present in marine environments but coral reefs are typically found in oligotrophic waters. However, with increased industrialization and construction along the coasts of many islands, nutrient levels are consequently increased, which can lead to eutrophication. The amount of land-based nutrients in the water directly relates to sponge growth and biomass (Wilkinson 1987) and as these pollutants increase so does the stress placed on diseased corals. This has been demonstrated by experimentally enriching yellow band infected colonies of *Montastraea franksi* and its congener *M. annularis* with concentrations of nitrate, phosphorus and ammonium. The concentration values used were within the range of concentrations found within

anthropogenically-enhanced areas, and caused the disease to spread an average of 1.8 times faster than the yellow band infected colonies that did not receive the increased nutrients (Bruno 2003). Other diseases, such as black band and aspergillosis, have also been found to affect more tissue in areas with high levels of nitrogen and phosphorus (Harvell et al. 2007).

Bonaire, Dutch Caribbean, has been subjected to multiple coral diseases (Voss and Richardson 2006), as well as eutrophication by anthropogenic input. Currently, the wastewater and sewage from residential and commercial areas are collected in septic tanks, which are then emptied into either the newly constructed, waste management plant, or into unlined trenches (R. Peachey, CIEE Bonaire Research Station, Bonaire, pers comm). Because the trenches are unlined, the waste has the potential to percolate through the porous limestone and can enter the water table which can affect both human and reef health (van Sambeek et al. 2000; Kayes 2008; R. Peachey, CIEE Bonaire Research Station, Bonaire, pers. comm.).

Since Bonaire has some coral diseases that increase in virulence based on nutrient input and sponges have the ability to remove these nutrients, this study focuses on the relationship between the abundance of sponges and the prevalence of coral disease. In this study, one hypothesis is tested:

H_1: Areas with higher sponge cover have a lower prevalence of disease, and areas with lower sponge abundance will have more diseased corals.

No research has been conducted on the potential relationship between sponges and disease. This study has conservation implications as areas of high sponge abundance could be vital to maintaining reef health because of their ability to limit the effects caused by increased nutrient content in the water.

Materials and Methods

Study Site

Study sites, Cliff and Windsock, were selected based on logistical considerations (Fig. 1). More than one site and depth was selected in order to increase the sample size of the data. Five transects were performed at both sites. Cliff is located north of multiple dive resorts and is characterized by a reef wall running north to south at about 13 m depth. This wall is covered in an assortment of sponges and small corals, which provide shelter to a number of fish and invertebrate species. Windsock's reef begins near the end of a small pier located there around 8 m depth and slopes down to about 30 m. Similar types of corals, disease and sponges are found at both sites.

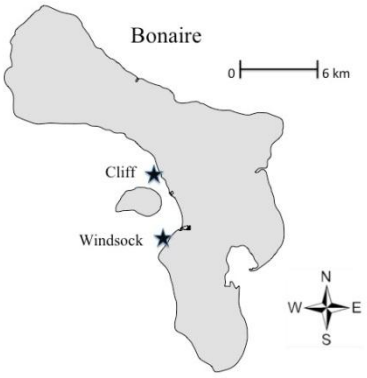

Fig. 1 Study sites of Cliff (12° 10.396' N, 68° 17.375' W) and Windsock (12° 7.962' N, 68° 16.979' W), Bonaire, (N 12°10', W 68°15'), Dutch Caribbean

Photo Quadrats

Percent live coral, diseased coral, and sponge were assessed at each site using a 10 m x 2 m belt transect at depths of 12 m and 15 m. Underwater photographs (Olympus Stylus Tough 8000) were taken using a 1 x 1 m^2 quadrat, for a total of 20 quadrats per transect (modified from Powell et al. 2010). Photographs of each quadrat were taken in two segments in order to increase resolution of the benthos (Fig. 2a).

Data Analysis

Ten of the quadrats were randomly selected from each transect and analyzed using Image J (Collins 2007). The photos were analyzed to find the total area of sponges and live and diseased scleractinian corals (Fig. 2b). A Pearson product moment test was used to test for a correlation between percent live coral cover and percent sponge because the data sets were found to be normally distributed. A Spearman rank correlation test was run on percent sponge cover and percent disease because disease was not normally distributed after transforming the data.

a.

b.

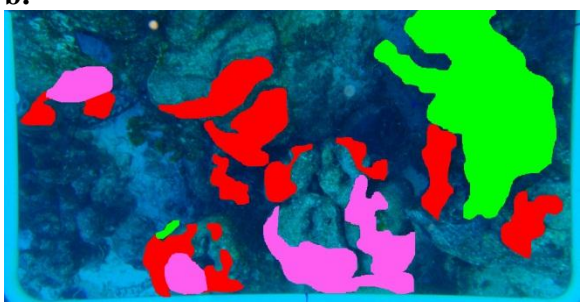

Fig. 2 Segment of quadrat. a) Before analyzing using Image J. b) After analyzing, *green* is sponge, *red* is live coral, *pink* is diseased coral

Results

Overall, the live coral cover for the sites surveyed was 15.52% ± 3.84 (mean ± SD) and the mean sponge and disease coverage was 2.91% ± 1.13 and 0.17% ± 0.30, respectively (Fig. 3).

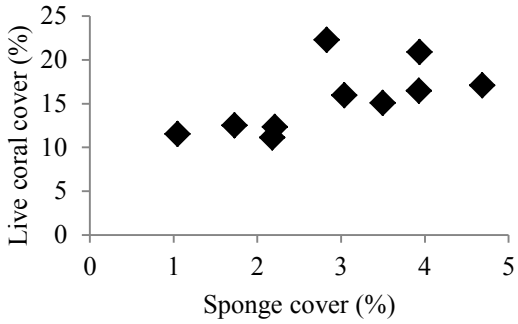

Fig. 3 Mean overall percent cover (±SD) of live coral, sponge, and diseased coral at both Windsock and Cliff

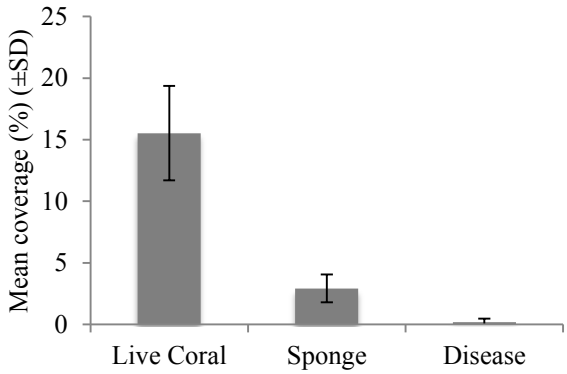

Fig. 4 Mean live coral coverage vs sponge coverage for each transect (r = 0.638, n = 10, p = 0.047)

A significant positive correlation was found between sponge and live coral cover (r = 0.638; n = 10, p = 0.047; Fig. 4). The highest percent live coral coverage was found at Windsock (22.28%) with sponge cover at 2.83%. The transect with the lowest live coral coverage, found at Cliff, was 11.14% with 2.18% sponge coverage.

No significant correlation was found between sponge and disease coverage (r = 0.485, n = 10, p = 0.115; Fig. 5). Of the ten transects surveyed, five had no diseased corals and none had more than 1%. The highest amount of disease coverage was found at Windsock (0.87%) in the same transect which had the highest live coral cover.

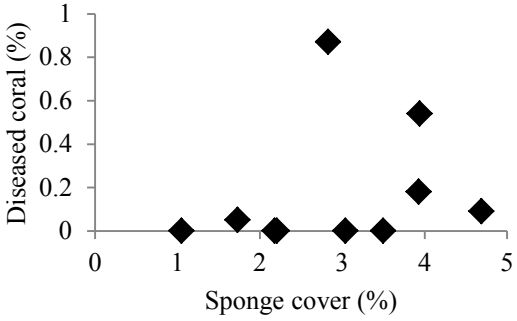

Fig. 5 Mean diseased coral vs sponge coral for each transect (r = 0.485, n = 10, p = 0.115)

Discussion

This study examined the relationship between sponge cover and disease prevalence at sites in Bonaire. Disease at both Cliff and Windsock was very low; neither had disease cover greater than 1%. These results did not support the hypothesis that disease prevalence would decrease in areas with higher sponge cover. This could be due to the fact that Bonaire has been subject to a large amount of bleaching and disease, as well as large storms such as Hurricane Lenny. These three factors stress corals and could have allowed for a greater amount of disease to affect the corals of Bonaire. By the time this study was conducted, a majority of the corals had already died leaving a very low amount of disease still present. Of the disease found, the most common disease was yellow band, which is slow moving, infecting tissues at a rate of 0.5-1.0 cm month^{-1} (Bruno 2003). Measuring the percent cover of dead coral may reveal that it would have been much higher than that found for live coral.

Live coral cover, however, was significantly correlated with sponge cover at the study sites. Powell et al. (2010) in the Wakatobi Marine National Park (WMNP), Indonesia, found similar results when studying coral cover and sponge density. As percent coral cover increased so did the sponge density. This may have to do with a sponge's ability to settle on virtually every available substrate whether it is on, in, or under live or dead coral (Powell et al. 2010). As sponges settle and grow, they have the ability to bind substrate together, which could aid in coral recruitment (Bell 2008). When recruits have a greater chance to settle, it increases the live coral cover and allows for reefs to start rebuilding after being affected by past disturbances.

Future studies should have a greater number of study sites and transects. This would give a larger picture of sponge abundance around Bonaire, and in the Caribbean. Other sites could also have a higher prevalence of disease based on a number of factors such as eutrophication and sedimentation. Measuring dead coral cover would also be helpful in calculating the amount of past disease that has affected an area. If this was combined with long-term studies of sponge abundance, the relationship between sponge cover and dead coral could be further studied to see if sponges chose to settle in areas with a greater amount of dead coral than live because of the amount of substrate available. Another interesting aspect to add to future studies would be measuring the nutrient concentrations around sponges and also near diseased corals. As previous studies have mentioned, an increase in two particular nutrients (ammonium and nitrate) have been shown to increase the rate of spread of yellow band disease (Bruno et al. 2003). With this being the most prevalent disease found on Bonaire and sponges having the ability to filter the water of those nutrients, it would be interesting to see if there are lower concentrations of the nutrients near sponges when compared to water closer to diseased corals.

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Assessing the correlation between coral disease prevalence and fish species richness from 2007-2011 in Bonaire, Dutch Caribbean

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Abstract

Coral reefs have existed for thousands of years and are currently subjected to many threats. Coral disease is of particular concern because of its increasing prevalence; because they reduce coral cover, diseases are likely to affect fish assemblages. This study looks at the factors of coral disease and fish species richness on Bonaire, Dutch Caribbean. The study tests the hypotheses that (1) the amount of coral disease present in Bonaire has increased since 2007 and (2) the species richness of fish assemblages decreases with increasing presence of coral disease. Coral disease prevalence and live coral cover was assessed using an adaptation of the AGRRA benthic methodology by laying 10 m transects at depths of 10 and 12.5 m at Cliff and Windsock, areas that are closed and opened to fishing respectively. In addition, fish species richness was assessed using the roving diver technique by REEF. There was a significant difference between coral disease, as well as live coral cover, between sites and years. Species richness had a significant weak but not significant correlation with coral disease and live coral cover. However, no significant difference was found in the fish species richness between sites or between years.

Introduction

Scleractinian corals are the primary building blocks of modern day coral reefs (Bruno et al. 2007) and have provided a diverse and complex framework for thousands of years (Hughes 1994). However, over the last two decades, coral reefs have been declining because of mass bleaching, macroalgae overgrowth, global warming, coastal pollution, overfishing, destructive fishing practices, and coral disease (Barber et al. 2001; Kuta and Richardson 2002; Jones et al. 2004; Raymundo et al. 2009). Another major threat that raises concern is the impact of overfishing. Overfishing has been occurring for centuries but has seen a sharp increase as fishing technologies have advanced (Pauly et al. 2002). Typically with overfishing, the large reef fish at higher trophic levels are targeted first, leaving much smaller fish at lower trophic levels (Murawski 2000). Thus, a major concern is how the fish biodiversity will be affected as coral reef health continues to decline with increasing fishing pressure (Kuta and Richardson 2002).

In addition to increased fishing pressure, coral diseases have contributed to the worldwide decline of reefs in the last decade (Pandolfi et al. 2003; Sutherland et al. 2004; Bak et al. 2005). Two factors that could be driving coral diseases are high water temperature and nutrient input (Kuta and Richardson 2002; Bruno et al. 2007), which provide a pathway into the environment for pathogenic organisms (Sutherland et al. 2004). Some diseases such as black band, white plague and white pox are bacterial, but the cause of other diseases remains unknown (Kuta and Richardson 2002; Sutherland et al. 2004). The main coral reef builders (e.g. *Diploria labyrinthiformis, D. strigosa, Montastraea* spp. and *Colpophyllia* spp.) appear to be most affected by disease, and thus disease could alter the composition, structure, and functionality of the reef (Sutherland et al. 2004). Moreover, because fish species richness and abundance depend on

structurally complex reefs (Ferreira et al. 2001), fish species richness could see a decline with increased disease prevalence.

Caribbean reefs are greatly affected by these global threats as they have experienced intense fishing pressure and are known as a disease hotspot; consequently these reefs warrant concern (Hughes 1994; Weil et al. 2000; Sutherland et al. 2004). A continuing threat to these reefs is the loss of fish assemblages and the consequent reduction in diversity and community structure (Raymundo et al. 2009). However, marine reserves can help reduce this decline in fish assemblages through the protection they provide (Jones et al. 2004). A study specifically looking at Bonaire and Curacao in the Caribbean concluded that coral cover is decreasing but appears more resilient in shallower reefs (Bak et al. 2005). Since reef fish species richness and abundance have a strong reliance on coral cover (Bell and Galzin 1984), threats such as disease, which can reduce coral cover (Raymundo et al. 2009), could mean a decrease in fish species richness.

A number of studies examining the current state of coral reefs acknowledge the danger that a reef of decreasing coral cover and complexity and increasing disease prevalence has on reef fish populations and species biodiversity (Pandolfi et al. 2003; Jones et al. 2004; Bruno et al. 2007; Raymundo et al. 2009). In Bonaire, there has been a decline in herbivorous reef fish since 1999 (Steneck and Arnold 2009), which are important to reefs because of their ability to control the algal growth on corals (Hughes et al. 2007). To protect these reefs and fish assemblages on Bonaire, two fish protected areas (FPAs) have been implemented and shown to aid in supplying higher abundances of reef fish (Steneck and Arnold 2009).

Using information on Bonaire's open and closed fishing zones, this study has used a dataset spanning five years to focus on the possible relationship between coral disease prevalence and fish species richness in 12.5 m depths. Along each transect, every leeward reefs on Bonaire, Dutch Caribbean from 2007 to 2011 between two sites Cliff and Windsock. The hypotheses of this study are:

H_1: The amount of coral disease present in Bonaire has increased over time

H_2: The species richness of fish assemblages decreases with increasing presence of coral disease.

No studies have directly looked at this relationship of coral disease and fish species richness, hence, this study could aid future research to help assess threats of coral disease and the effects it has on this ecosystem. Also, this study could provide important information for protection and management regarding reefs and fish communities.

Materials and Methods

Study Site

The study was conducted at two sites in Bonaire, Dutch Caribbean: Windsock (12° 7'58.50" N, 68°16'59.40" W), which is open to fishing, and Cliff (12°10'26.44" N, 68°17'26.62" W) which has been closed to fishing since 2008 (STINAPA 2003; Fig. 1). Data were collected over a five-week period from September to November 2011. These sites have been subjected to long-term benthic surveys from 2007 by CIEE Research Station Bonaire and fish abundance surveys since 1994 (REEF 2011), making them ideal sites for this study.

Data Collection

The presence of coral disease was assessed using line intercept transects that followed an adapted Atlantic Gulf Rapid Reef Assessment (AGRRA) benthic survey procedure conducted using SCUBA (Kramer et al. 2005). Six 10 m transects and ten 10 m transects were laid at Cliff and Windsock, respectively, at approximately 10 m and

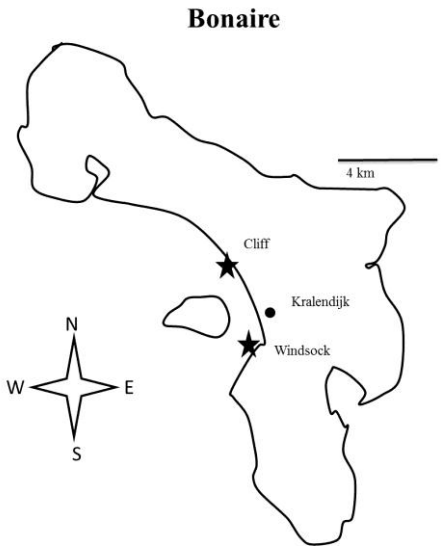

Fig. 1 Site map of two study sites: Cliff (12°10'26.44" N, 68°17'26.62" W) and Windsock (12° 7'58.50" N, 68°16'59.40" W), Bonaire, Dutch Caribbean indicated by *black stars*. Cliff is closed to fishing and Windsock is open to fishing. *Black circle* indicates the capital Kralendijk

live coral colony greater than 10 cm was identified to species level, its dimensions were measured (length, width, height) and any diseases were identified. For this study, coral bleaching was not counted as a disease. The cover of recently dead coral and old dead coral were recorded as recently dead coral (lacking color but identifiable to species level) and old dead coral (defined as unrecognizable and typically accompanied by algal growth). These data were used to calculate percent live coral cover and disease prevalence (i.e. percent of coral colonies infected by disease).

The fish species richness at the study sites was assessed using the roving diver technique (REEF 2011). Three REEF surveys were performed at Cliff and four surveys at Windsock. For this study, the REEF survey was completed after entering the water to a max depth of 24.4 m with 5-10 min spent at approximately each 3.3 m depth increment for a maximum time of 60 min. Fish species richness was measured by adding total different species seen per survey.

Data on coral disease prevalence and fish species richness were collected from the archives of past research from CIEE Research Station Bonaire and the REEF database (REEF 2011). The CIEE Research Station Bonaire has conducted research using AGRRA benthic transects since 2007. From this dataset, 55 and 32 benthic transects were gathered for Cliff and Windsock respectively. All benthic transects included in data analysis were at various depths from 5.8 to 17.0 m with an average depth of 10.5 ± 2.3 m (SD). Due to incomplete data, one benthic transect from Cliff in 2009 was not included in data analysis. From the two types of surveys the REEF database provided, novice level surveys were collected for this study; 33 surveys for Cliff and 26 for Windsock. No transects in 2007 were recorded for Windsock and no novice surveys were found for Windsock in 2008 and 2011.

Data Analysis

All data were tested for normality. A non-parametric two-factor Kruskal-Wallace test was used to analyze the relationship between the prevalence of coral disease (%) over time from 2007-2011 and between study sites Cliff and Windsock. A two-way ANOVA was used to analyze the relationship between (1) the amount of live coral cover (%) over time from 2007-2011 and between sites, and (2) fish species richness over time from 2007-2011 and between sites. Data for two-way ANOVA were log transformed to fit assumptions of normality. The non-parametric Spearman's rank correlation test was used to analyze the association of (1) mean coral disease prevalence (%) and mean fish species richness and (2) mean live coral cover (%) and mean fish species richness.

Results

Surveys from 2007 to 2011 identified 18 different types of coral, 15 to species level and 3 to genus level, between both sites Cliff and Windsock. The REEF surveys identified 184 different fish species for Cliff

throughout 2007-2011 and 176 fish species for Windsock (REEF 2011). According to transect data, disease has not been recorded until 2008 in Cliff and Windsock, and the diseases seen within these transects include white plague, black band, yellow band, white band, and dark spot.

Disease prevalence and live coral cover

A Kruskal-Wallace test revealed significant differences in the prevalence of coral disease at Cliff and Windsock ($\chi^2 = 7.227$, df = 1, p = 0.007) and between years ($\chi^2 = 28.75$, df = 4, p < 0.001; Fig. 2). Cliff had no disease reported in transects in 2007. There was a sharp increase in disease from 2008 to 2009 with consequent years remaining at a higher level of disease for Cliff. The trend for Windsock remained inconclusive (Fig. 2). The highest mean percent of coral disease in Cliff was found in 2009 (24.38 ± 33.58) and 2011 for Windsock (29.97 ± 23.92) with the lowest for Windsock in 2010 (15.20 ± 24.25).

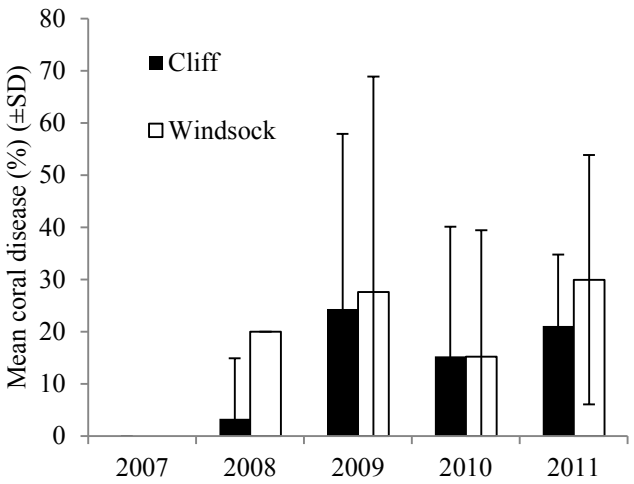

Fig. 2 Mean percent of coral disease (±SD) from 2007-2011 between Cliff (*solid bars*) and Windsock (*open bars*). At Cliff 9, 20, 9, 9, and 14 transects were laid for years 2007-2011 respectively. Windsock: 1, 10, 13, and 18 transects were laid for years 2008-2011 respectively. (Kruskal-Wallace, sites: $\chi^2 = 7.227$, df = 1, p = 0.007; years: $\chi^2 = 28.75$, df = 4, p < 0.001)

A two-way ANOVA indicated a significant difference for the amount of live coral cover (%) between years (F = 4.857, df = 3, 85, p = 0.004) and between sites (F = 5.442, df = 1, 85, p = 0.022; Fig. 3). No transects in 2007 were performed for Windsock. No noticeable trend or pattern can be seen in live coral cover between Cliff and Windsock. The highest percent of live coral cover for Cliff was found in 2007 (17.69 ± 7.58) and lowest in 2008 (12.22 ± 6.43). The highest percent for Windsock was in 2010 (28.60 ± 10.55) with lowest in 2009 (11.32 ± 11.60).

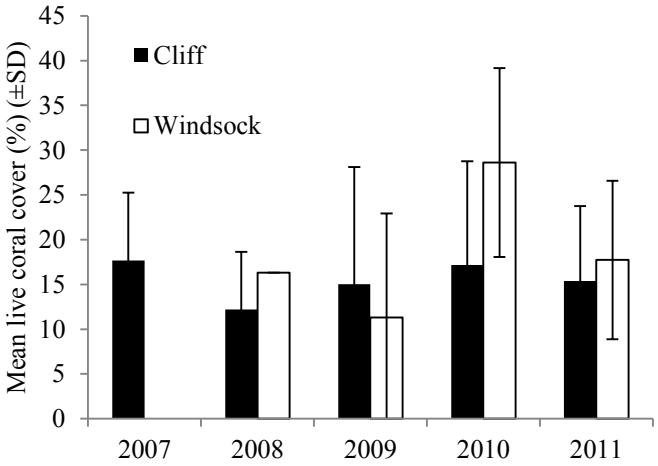

Fig. 3 Mean live coral cover (%) (±SD) from 2007-2011 between Cliff (*solid bars*) and Windsock (*open bars*). At Cliff 9, 20, 9, 9, and 14 transects were laid for 2007-2011 respectively. For Windsock 1, 10, 13, and 18 transects were laid for 2008-2011 respectively. (ANOVA, sites: F = 5.442, df = 1, 85, p = 0.022; years: F = 4.857, df = 3, 85, p = 0.004)

Fish species richness

No significant difference was found in mean fish species richness between sites (F = 0.959, df = 1, 57, p = 0.332) or over time (F =1.507, df = 4, 57, p = 0.212) by using the two-way ANOVA (Fig. 4). Cliff appears to have remained stable over the years while no clear trend can be seen for Windsock. The highest mean number of fish species seen per survey for Cliff was seen in 2007 (66 ± 15) with lowest in 2011 (56 ± 7). For Windsock, the highest was found in 2007 (78 ± 12) with lowest in 2009 (56 ± 14).

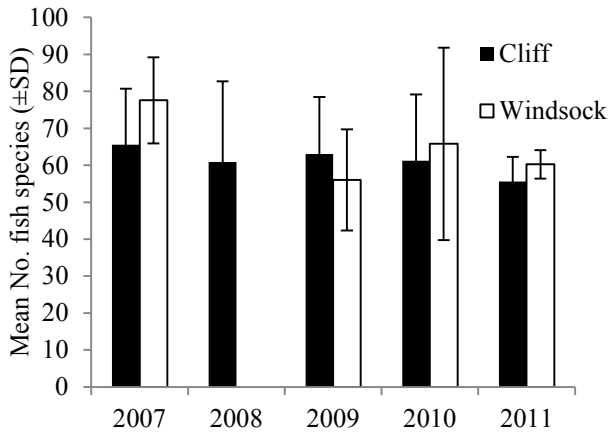

Fig. 4 Mean no. fish species per REEF survey (±SD) from 2007-2011 between Cliff (*solid bars*) and Windsock (*open bars*). At Cliff 9, 9, 8, 5, and 5 transects were laid for years 2007-2011. For Windsock 9, 8, 9, and 4 transects were laid for years 2007, 2009-2011 respectively. (ANOVA, sites: F = 0.959, df = 1, 57, p = 0.332; years: F = 1.507, df = 4, 57, p = 0.212)

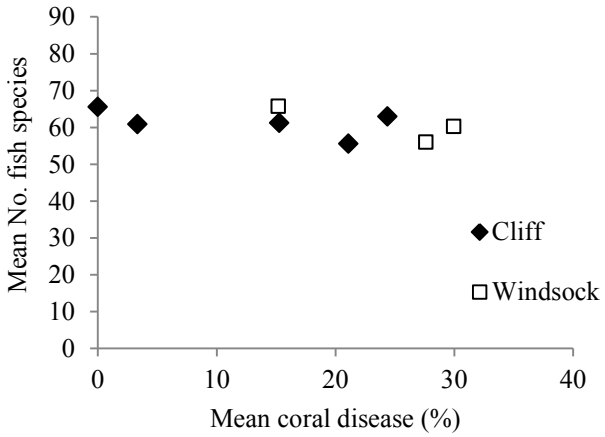

Fig. 5 Correlation of mean no. fish species and mean coral disease (%) between Cliff (*solid diamonds*) for 2007-2011 and Windsock (*open squares*) for 2009-2011. (Spearman's rank, r_S = -0.627, p = 0.096)

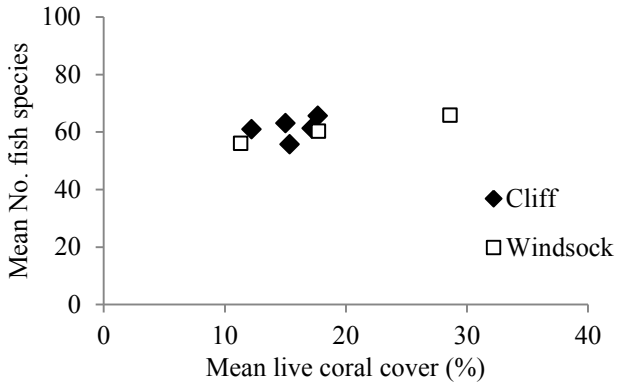

Fig. 6 Correlation of mean no. fish species and mean live coral cover (%) between Cliff (*solid diamonds*) for 2007-2011 and Windsock (*open squares*) for 2009-2011. (Spearman's rank, r_S = 0.530, p = 0.177)

The Spearman's rank correlation test between mean coral disease prevalence (%) and fish species richness revealed a negative but non-significant correlation between both sites Cliff and Windsock over time (r_S = -0.627, p = 0.096; Fig. 5). The same test used to assess the association of fish species richness and the mean amount of live coral cover (%) revealed a positive but non-significant correlation between both Cliff and Windsock over time (r_S = 0.530, p = 0.177; Fig. 6).

Discussion

The increase in coral disease for Cliff in 2008 to 2009 and the higher level of disease from 2009 to 2011 support the hypothesis that disease has increased over time. In addition, a difference was found between Cliff, a fish protected area, and Windsock, an area open to fishing, in coral disease prevalence which appeared to be in 2008. However, the statistical test did not show where the difference lies. The lack of differences found in fish species richness between Cliff and Windsock from 2007 to 2011, as well as no association with coral disease, does not support the hypothesis that as disease has increased over time the fish species richness decreased.

The significant difference in live coral cover between Cliff and Windsock appears to be in 2010 when compared to other years; however, it is difficult to discern a pattern because of the high variation. Moreover, because coral cover displayed no clear trend between Cliff and Windsock over time, this could explain why there was no significant association between coral cover and fish species richness (Fig. 6). For future work addressing coral cover, the association between diseases and live cover could be evaluated to assess whether increasing disease is associated with decreasing cover, where a relationship would be expected (Sutherland et al. 2004).

Effectively managed marine protected areas lowered disease prevalence when diverse fish assemblages were maintained in the Philippines (Raymundo et al. 2009). However, this was not observed in the FPA of the current study. The difference between Cliff and Windsock in terms of disease was surprising considering the five year time period that was examined. This was surprising because a more gradual trend was expected and not the sharp increase that was seen.

No strong association was seen with disease and fish species richness or fish species richness over time. The lack of difference between Cliff and Windsock in fish species richness was unexpected because the FPAs were implemented with the aim of increasing and reestablishing the reef fish community within the protected areas (Steneck and Arnold 2009). No difference in fish species reported between sites does correspond with the amount of fish species seen overall for Cliff and Windsock, which had similar amounts of fish identified from novice surveys from 2007-2011 (REEF 2011). It takes time to induce ecological changes to a system, and because reef ecosystems are constantly affected by anthropogenic threats and changes, FPAs may require more time to show a significant change in fish species richness (McClanahan et al. 2002). Another factor could be the size of the FPAs. These FPAs are approximately 1.2 and 2.6 km long (STINAPA 2003) and may be too small to hold fish populations' daily migratory routes and home ranges which can vary up to 4.6 km (Kramer et al. 1999). These small FPAs could allow cross over between areas closed and open to fishing making it hard for the FPAs to protect the fish assemblages. If this is the case, then a larger protected area may be called for.

The increase that was seen in the prevalence of coral disease on Bonaire could be explained by temperature anomalies which would increase the susceptibility of corals and the virulence of pathogens. Another factor could be environmental stressors such as high nutrient concentrations from runoff into the waters (Bruno et al. 2007). Future studies for Bonaire could look at nutrient levels at these two sites, and other similar sites, as well as studying temperature influences caused by seasonal shifts to note possible effects on the increased coral disease prevalence.

Another future improvement would focus more on assessing fish species richness by conducting more surveys with the same people performing them. The data from REEF are accurate and reliable but the distinction between novice and expert level surveyors could change as people perform more surveys and learn more fish. For this study some surveys gathered from the REEF database appeared high for a novice survey in each year, so a future step would put boundaries on how many fish species define a survey type. Another step would be to use expert surveys instead of novice and reanalyze fish species richness over time and against coral disease and live coral cover to see if the result changed.

Ecological changes take time to occur and more time may be needed to discern the future status of fish species richness and its relationship between coral disease prevalence on Bonaire. Since disease and live coral cover were found to be significantly different between Cliff and Windsock, this could support the movement for more protection to be placed on reefs to reduce future threats to live coral cover as well as more management to combat disease on reef ecosystems. However, more research is needed in determining possible relationships and causes for disease presence on reefs such as temperature, nutrient effects, and other factors, and their effect on species richness of fish assemblages; not just the effects of fishing pressure on reefs.

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Use of trophic structure as an indicator of reef fish assemblages in areas open and closed to fishing

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Abstract
The change in the mean trophic level of fish assemblages can be used as an indicator of fishing pressure. To gain a more detailed understanding of changes in trophic level, trophic spectra can be derived using a 3-point moving average technique. The mean trophic level was used to determine differences between reef fish assemblages inside and outside fish protected areas (FPAs) around Bonaire, Dutch Caribbean. Additionally, mean trophic levels of reefs spanning the entire Caribbean were calculated to enable comparison based on their level of degradation by anthropogenic disturbances. No difference was seen between the mean trophic level of sites inside and outside the FPAs, possibly due to the fact that the FPAs have only been in place for 4 years. Differences in mean trophic levels across the Caribbean were significant but did not correspond to estimated reef health, implying that mean trophic level may not be a good indicator of reef health. Reef fish assemblages seem to be affected by a variety of factors, not simply fishing pressure or reef health.

Introduction

Anthropogenic disturbances, described as forces affecting populations as a result of human activities, have been an increasing stressor on coral reef systems around the world (Benedetti-Cecchi et al. 2001, Burke et al. 2011). Overfishing, pollution, and destructive fishing habits are only a few of the practices that affect reef fish assemblages (McClanahan et al. 2002). These changes may lead to the decline of reef fisheries (Pauly et al. 1998).

Coral reef fishes are an integral part of coral reef ecosystems. Herbivorous fish regulate algae populations, and piscivores prey on herbivores and lower-level consumers (Hughes 2007). This top-down regulation controls prey assemblages and promotes a stable ecosystem (Hairston et al. 1960). Feeding habits within a community can be quantified by trophic levels, determined by where a species fits in the food web based on dietary studies (Hairston et al. 1960). For example, primary producers have a trophic level of 1, whereas primary consumers (herbivores) have a trophic level of 2. This scale continues to a range of 4-5, which describes apex predators (organisms with no natural predators). By analyzing the species density and diversity of a particular area, the trophic structure can be determined (Bozec et al. 2005). The change in trophic structure of a reef fish assemblage can provide insight into disturbances and has been used as an indicator of fishing pressure (Bozec et al. 2005, Gascuel et al. 2005). One may be able to identify different sites affected by overfishing by comparing the trophic structure of reef fish assemblages (Gascuel et al. 2005).

Increased fishing activity has decreased the abundance of both herbivorous and carnivorous fishes (Gascuel et al. 2005). A reduction in the mean trophic level of global fisheries by 0.2 from 1950 to 1994 indicates that there are fewer apex predators available and as such fisheries target smaller fish with lower biomass at lower trophic levels (Pauly 1998).

One method of increasing fishery biomass and abundance is through the use of no-take marine reserves (NTMRs; Russ and Alcala 2004). NTMRs have shown to increase biomass of fishes and biodiversity,

which in turn benefits local fishers (Russ et al. 2008; McClanahan 1999).

This study examined the difference in trophic structures between sites open and closed to fishing around Bonaire. It was hypothesized that:

H_1: the area closed to fishing has a significantly higher mean trophic level than the area open to fishing.

The results of this study provide important insight into the health of the reefs of Bonaire by assessing the structure of current reef fish assemblages and whether these differ between areas open and closed to fishing.

An additional aim of this study was to compare trophic levels of several Caribbean reefs along a gradient of pristine to heavily impacted as identified by Pandolfi et al. (2003) and to assess where the trophic structure of Bonaire's reefs lies in comparison to these. It was hypothesized that:

H_2: the fringing reefs of Bonaire will have a significantly different mean trophic level from the least degraded reef (Belize).

The results of this study provide an assessment of the reef fish assemblages of Bonaire in relation to other reefs found in the Caribbean.

Materials and Methods

Study Sites

All coastal waters from the high tide mark to a depth of 60 m around Bonaire, Dutch Caribbean, and its sister island Klein Bonaire fall within the Bonaire National Marine Park. In 2008, two small fish protected areas (FPAs), defined as an area in which fishing is forbidden except for the collection of baitfish, were designated along the west coast of Bonaire (Fig. 1). The north and south FPAs extend approximately 2.5 and 1 km respectively. This study focused primarily on four sites on the western coast of Bonaire: Barcadera and Windsock, located in fished areas (FAs), and Reef Scientifico and 18 Palm, located within the FPAs (Fig. 1).

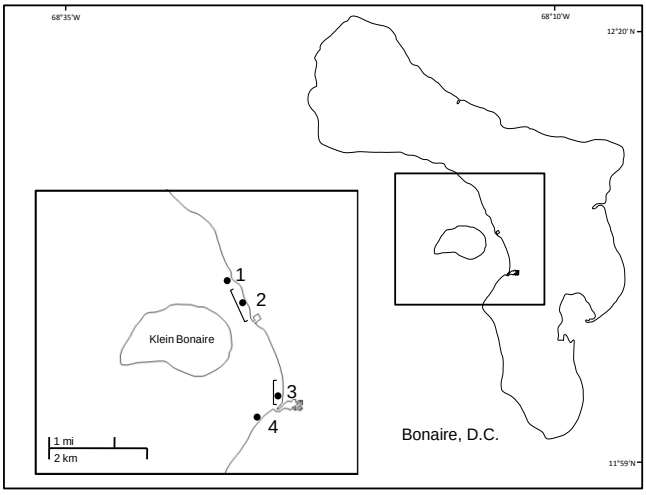

Fig. 1 Map of Bonaire, Dutch Caribbean. *Black dots* designate study sites: Barcadera (1, 12° 11.294' N, 68° 17.819' W), Reef Scientifico (2, 12° 10.396' N, 68° 17.375' W), 18 Palm (3, 12° 8.301' N, 68° 16.611' W), and Windsock (4, 12° 7.962' N, 68° 16.979' W). *Brackets* denote the north FPA (2.5 km long) and south FPA (1 km long)

In order to estimate densities of fish species at each site, it was necessary to use two different underwater visual census protocols: (1) the abundance of larger, less abundant and mobile species of fish were recorded in 50 m x 3 m belt transects; and (2) the more abundant, demersal species were recorded in 10 m x 1 m belt transects. Transect widths were estimated using a t-bar according to Atlantic and Gulf Rapid Reef Assessment protocol (Lang 2010). In order to obtain representative data of the fish assemblages at each site, both dimensions of belt transect were conducted at depths of 2, 5, 10, and 15 m starting at randomly selected points. Data from all transects within each area were combined to obtain abundance of all species identified over a 600 m^2 area, as per Steele and Forrester (2005). Apex predators observed outside the transect were recorded for reference but were not used to calculate density.

Data from Reef Environmental Education Foundation (REEF 2011) were used to calculate mean trophic level of the eight reefs in the comparative study. By selecting certain reefs that were ranked

according to degradation based on scale of human influence, this study attempted to evaluate the relative condition of Bonairean fringing reefs. The mean trophic levels were compared between eight Caribbean countries (listed here in order of increasing degradation): Belize, Bermuda, Cayman Islands, the Bahamas, East Panama, the United States Virgin Islands, and Jamaica (Pandolfi et al. 2003). Sighting frequency and density values were compiled for every species recorded at the eight countries in the past five years, and trophic level data were compiled from Fishbase (Froese and Pauly 2011). The number of individuals was estimated from data collected by REEF and used in data analysis.

Mean trophic levels of the fish assemblages for each site were calculated from density data and species' trophic levels given by Fishbase (Froese and Pauly 2011). A two-way nested ANOVA was used to investigate the difference in trophic levels between sites around Bonaire with FPA vs. FA nested within the site. Abundance data from REEF reports were used to compare mean trophic levels of the eight Caribbean reefs.

Results

A total of 82 fish species from 26 different families were recorded on Bonaire, with 69,768 individuals observed ranging from a trophic level of 2.0 to 4.5 (Table 1). The majority of individuals surveyed were members of the Gobiidae family (weighted trophic level = 2.7). A trophic spectrum based on the density data was created by a trendline using a 3-point moving average technique as per Gascuel et al. (2005; Fig. 2). The trophic spectra of the FPAs and FAs show similar patterns if the Gobiidae family is represented. However, by removing Gobiidae differences in trophic spectra can be seen. Both areas have peaks at 2.0 and 3.0-3.1; however the spectrum for FPA sites peaks around 3.5, whereas the spectrum for FA sites peaks around 3.3. FPA sites had a mean trophic level of 2.71 ± 0.01 whereas FA sites were 2.68 ± 0.01. Mean trophic level was not significantly different between sites inside and outside the FPAs (two-way nested ANOVA: $F = 0.149$, $df = 1$, $p > 0.05$; Fig. 3).

Table 1 Density (individuals 600 m^{-2}) of families surveyed at study sites (n = 4) averaged across the four study sites Windsock, Barcadera, 18 Palm, and Reef Scientifico

Family	Density (individuals 600 m^{-2})
Acanthuridae	30.50
Aulostomidae	4.75
Carangidae	30.00
Chaenopsidae	11.25
Chaetodontidae	5.00
Gerreidae	0.50
Gobiidae	15401.25
Grammatidae	20.50
Haemulidae	48.00
Holocentridae	9.25
Labridae	361.00
Lutjanidae	51.75
Monacanthidae	0.50
Mullidae	33.75
Muraenidae	1.00
Ophichthidae	0.25
Ostraciidae	1.75
Pomacanthidae	1.75
Pomacentridae	1332.75
Priacanthidae	0.25
Scaridae	37.50
Sciaenidae	0.25
Serranidae	41.75
Sphyraenidae	0.75
Synodontidae	0.50
Tetraodontidae	15.50

There was a significant difference between the mean trophic levels of the 8 reefs in the Caribbean-wide study (Fig. 4; $F = 3297.9$, $df = 7$, $p < 0.001$). No significant correlation was found between mean trophic level and degradation based on the study by Pandolfi et al. (2003; $R = 0.1357$, $p > 0.05$). The Bahamas had the highest mean trophic level (3.07 ± 0.01), whereas the lowest mean trophic level was East Panama (2.87 ± 0.03). Belize (mean trophic level = 3.04 ±0.01) had a significantly different the mean trophic level to that of Bonaire (2.92 ± 0.02; $p < 0.001$).

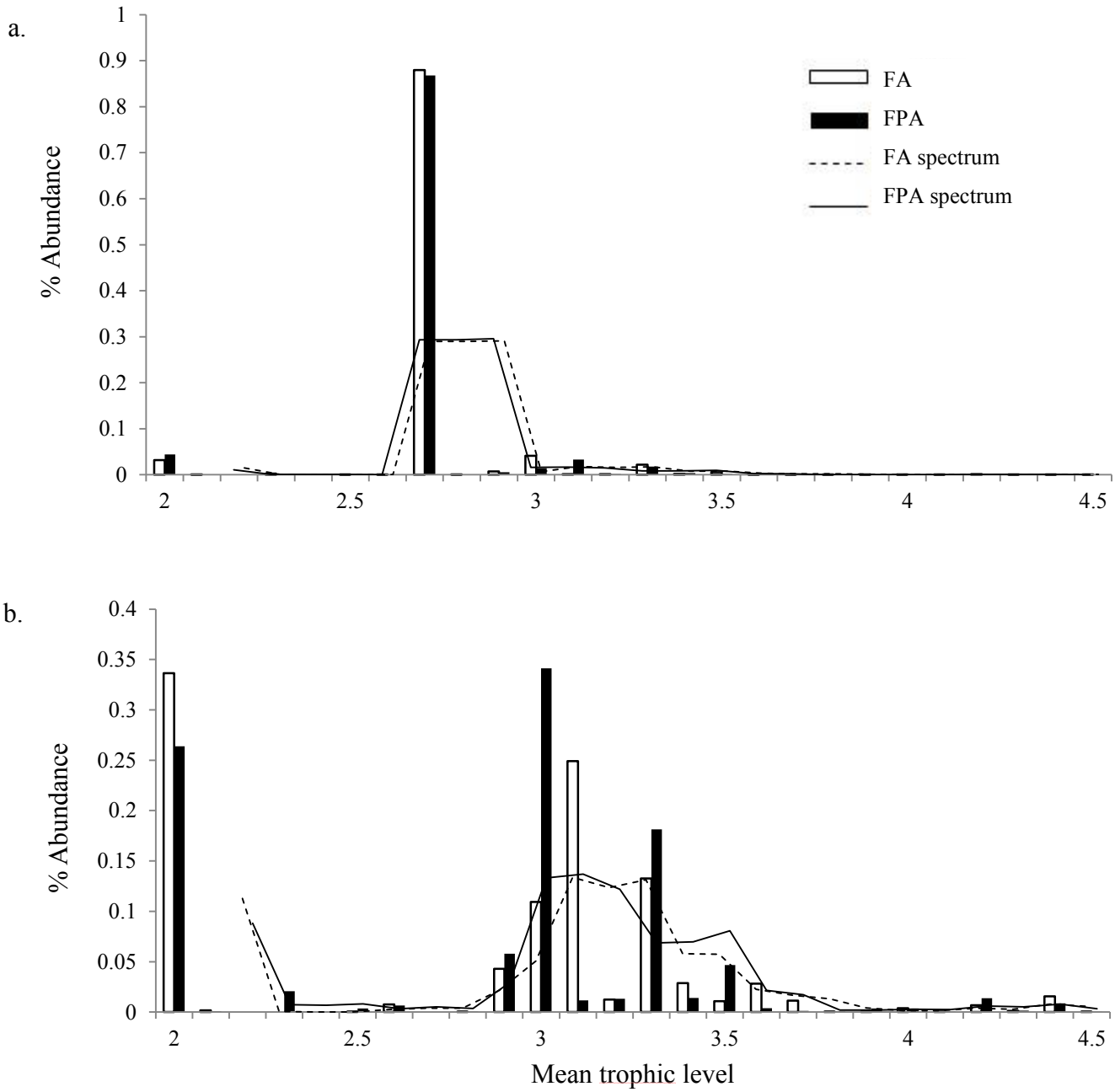

Fig. 2 Trophic spectra of FPA and FA (n = 4). Columns indicate the proportional abundance of organisms per trophic level. Trendlines were created using a 3-point moving average technique. **a.** illustrates the trophic spectra with the Gobiidae family, whereas **b.** omits the Gobiidae community

Discussion

It cannot be said that there is a difference in mean trophic level as a result of a fishing pressure gradient between the FPAs and FAs in Bonaire. This could be due to a variety of factors. FPAs around Bonaire have only been in operation for four years, and the level of compliance is unknown. Thus the intensity of fishing may not be significantly reduced compared to FAs. The fact that the currently established FPAs are so small may also adversely affect productivity (Claudet et al. 2008). Though small, the 0.03 difference in mean trophic level between FPA and FA is interesting considering that Pauly et al. (1998) observed a 0.2 change in the global mean trophic level over 45 years. Also, the researchers directly

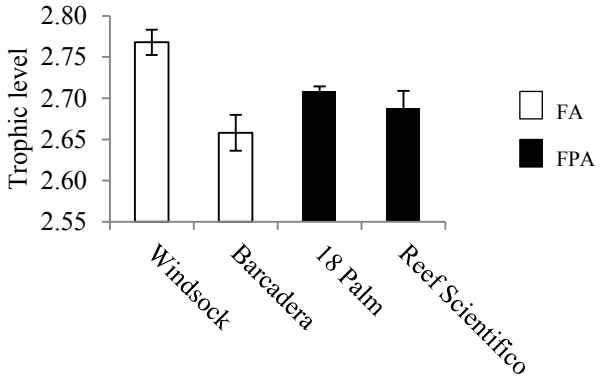

Fig. 3 Mean trophic level of the sites surveyed (± SE; n = 4). *White columns* are sites within the FPA (n = 2), and *black columns* are sites outside the FPA (n = 2). Mean trophic levels were not significantly different between FPA and FA sites (2.71 ± 0.01 and 2.68 ± 0.01, respectively; $p > 0.05$, see details in text)

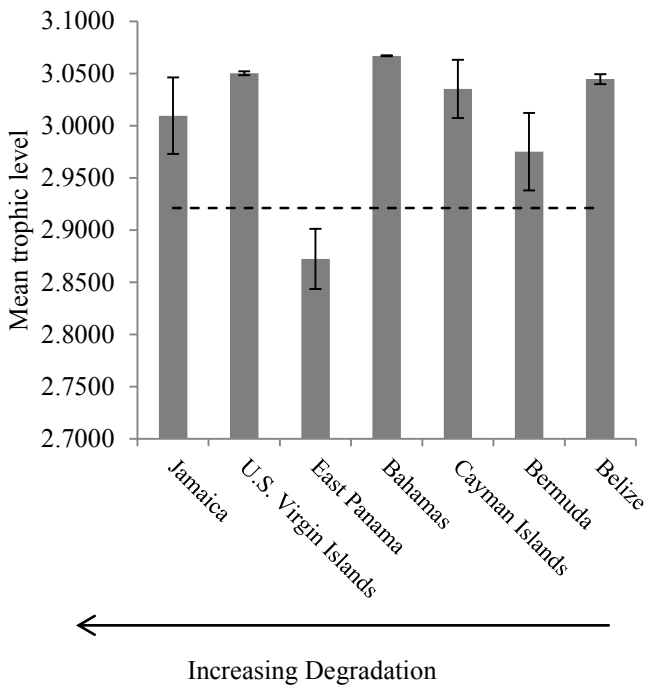

Fig. 4 Mean trophic level of countries in the Caribbean. Reefs are organized left to right from most degraded to least degraded as per Pandolfi et al. (2003). The *dashed line* denotes the mean trophic level of Bonaire in relation to the other sites. No significant correlation has been observed between reef degradation and mean trophic level. There is a significant difference between different reefs ($p < 0.001$, see details in text)

observed more apex predators in the FPA than in the FA. Future studies may indicate that mean trophic level will continue to diverge between the two areas, indicating a positive trend in response to a release from fishing pressure.

The Gobiidae family drastically affected the trophic spectra by altering the mean trophic level due to the sheer size of the Gobiidae community. If this study recorded total length of the organisms, biomass could be estimated and the trophic level could be weighted according to total biomass. If this happened then the contribution of Gobiidae might be much less. Smaller, less mobile fish are often overlooked in trophic studies and may affect assemblages more than is commonly thought (Willis 2001).

The results of the Caribbean-wide study provide some insight into the use of mean trophic levels as a reef health indicator. Although a difference was found between the mean trophic levels of Belize and Bonaire, no relationship was found between mean trophic level and the state of degradation of the reef. It seems that mean trophic level may not be a good indicator of reef integrity. These results are limited by the scope of REEF data: surveys are conducted by recreational divers, sites are not randomly chosen and divers often choose less degraded sites to conduct surveys. Surveyors may also be overlooking smaller, demersal fish because they are focusing on larger pelagic species.

Mean trophic level may be greatly affected by factors other than fishing. For example, invasive species may trigger a change in fish assemblages at healthier reefs. Based on the results it is impossible to estimate how Bonaire reef degradation relates to other Caribbean reefs based solely on mean trophic level. If this study were reproduced in the future, estimating total length of all individuals observed would facilitate estimates of biomass and thus allow researchers to accurately weight the contribution of each individual to the mean trophic level. In addition, in order to gain a better regional perspective, protocols better able to accurately represent a complete reef fish assemblage (such as those employed in

the present study) should be employed rather than REEF. Random surveys conducted by a single research team would yield consistent, more reliable results.

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