Guana Island

Marine Science Month Report

2004 - 2005

Compiled by
Lianna Jarecki, Ph.D.
Marine Science Month Program Coordinator

Reef Symposium in Japan in early July, 2004. This project was expanded this year to include other islands in the BVI to give a better reference to which Guana's reefs may be compared.

Coral Bleaching

During August and September of 2005, the BVI and other nearby areas experienced warmer than normal seawater temperatures (Figure 1). We saw corals beginning to bleach during August, but many more were bleached by the end of September. Bleaching in corals occurs when they eject their symbiotic zooxanthellae (an algae that lives inside the coral tissue) in response to temperature stress. Often the corals die as a result, but they may also recover if the bleaching is not severe.

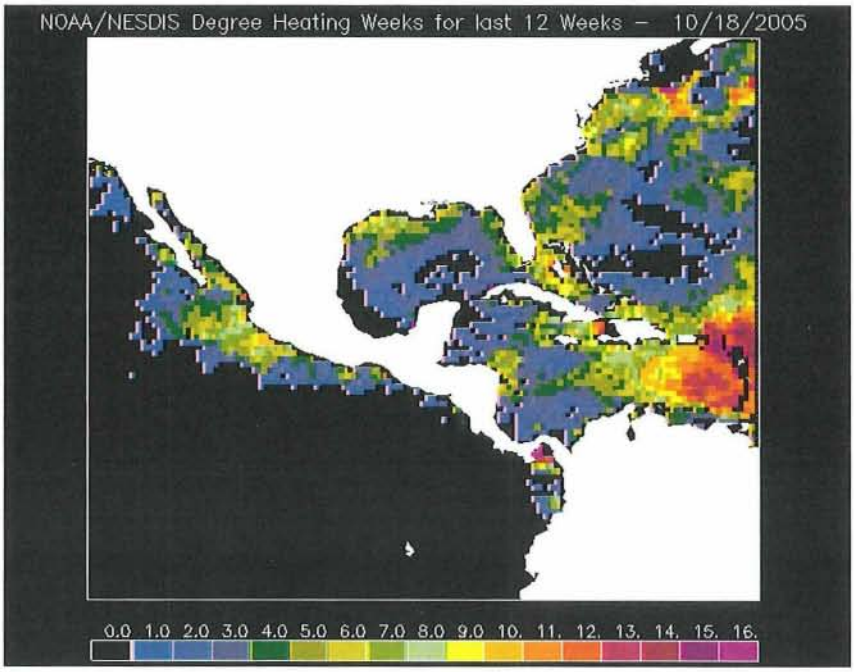

Figure 1: NOAA SST map for August – October. Note the purple color (highest temperature) in the Virgin Islands and eastward.

In October, Graham Forrester and Lianna Jarecki surveyed two of the long-term monitoring sites for bleached corals. Dr. Forrester reports the following:

> We have performed intermittent surveys at the 8 Guana monitoring sites to assess the abundance of recently killed corals, which includes corals recently killed by diseases as well as those killed by bleaching. These surveys were also done at 16 sites throughout the BVI in 2004. In each case, we counted the number of recently dead coral colonies that were > 5 cm in diameter, within a 30 m long x 1.5 m wide transect.
>
> In all of these surveys, the number of dead colonies per transect never exceeded 9, and was typically 3 to 4. In October 2005, we performed identical surveys at two of the Guana monitoring sites (Iguana Head and

> White Bay). **The number of dead colonies per transect (this year) ranged from 169 to 279, close to a 100 fold increase in coral mortality.** (emphasis added)

We will know from our monitoring work next year if any of these corals were able to recover. However, during the coral replanting project (described later), which continued through November, we observed many of the corals in White Bay dying as a result of bleaching and/or secondary infections, particularly black band disease.

Ship grounding

On July 7th, 2004, the Holo Kai, a ship of 165 feet in length, sailed into Muskmelon Bay on Guana Island, where it dropped two very large bow anchors in a small patch of sand, then backed up as it layed out 150 feet of heavy chain across the coral reef and finally placed a third, stern anchor with heavy chain lying across a deeper part of the coral reef. Muskmelon Bay, because of its healthy coral reef and abundant fish populations, is a recently-declared Fishing Priority Area, a popular dive site, and, as a result of the new Fisheries Regulations, a site where any anchoring is illegal.

MSM scientists observed this boat anchored over the reef and set out quickly to survey the damage. We found brain corals the size of large boulders with anchor scrapes, medium-sized and smaller coral heads broken or overturned, and sea fans and soft corals flattened (Figure 2).

Figure 2: Damage to the reef from the Holo Kai's anchor chains. Broken columns of pillar coral (left); anchor scrapes on boulder coral (right).

The area of continuous damage caused by the Holo Kai anchors measured 30,000 square feet, more than 2/3 of an acre. The incidence of damaged corals inside this area was more than ten times that measured in nearby areas. The potential for recovery from such damage is low because corals grow extremely slowly (most grow less than ½ inch per year).

We followed up by contacting and finally meeting with the captain and with the owner of the Holo Kai. We prepared a report for the Conservation and Fisheries Department (CFD) of the BVI government, and we published articles in the local newspapers and on the internet. Unfortunately CFD never pressed charges against the Holo Kai nor took other action to resolve the problem of boats anchoring on or very near coral reefs.

Are Guana's reefs affected by coral diseases?

Scientists: Longin Kaczmarsky; Sonya Kaczmarsky

Recently-discovered coral diseases are often fatal and appear to be spreading through some of the most abundant coral populations in the Caribbean. Mr. Kaczmarsky and his assistant conducted an intensive survey of diseased corals around Guana during 2004. He found that many of Guana's corals are infected by a variety of recognizable conditions. He also recorded rapid spread of infection through individual corals. Mr. Kaczmarsky hopes to continue his studies in the Caribbean (he also monitors reefs in St. Croix) as the basis for his PhD research.

Will Elkhorn Coral populations recover from the devastating infection of the 1980s?

Scientists: Caroline Rogers, PhD, Erinn Muller

Elkhorn coral was the largest and most abundant shallow-water coral in the Caribbean until it was nearly wiped out by an infectious disease during the 1980s and early 90s. The grandeur of former Elkhorn reefs can still be recognized in the coral rubble and standing branches of dead coral in popular snorkelling areas near shore (such as White Bay and Muskmelon Bay). Elkhorn corals now appear to be making a slow come-back. Small colonies are growing on top of former Elkhorn reefs. Dr. Rogers has pioneered the study of Elkhorn coral recovery in the Virgin Islands. She has now expanded her studies to include the reefs around Guana, where she focuses on tracking the growth of individual coral colonies in Crab Cove.

Can we replant areas where Elkhorn coral has died out?

Scientists: Lianna Jarecki, PhD, Graham Forrester, PhD, Linda Forrester, and Caitlin O'Connell-Rodwell, Ph.D.

A project to replant Elkhorn coral on the finger reefs of White Bay, the main beach area of Guana Island, began in 2005. An initial intensive survey showed that live coral accounts for only 5% of these reefs. Elkhorn coral, the fastest growing shallow-water, reef-building coral in the Caribbean, was originally responsible for building the finger reef structures in White Bay, but the surveys showed that Elkhorn coral was extremely rare on these reefs. Elkhorn coral was found growing in other areas of White Bay, where naturally broken-off pieces were collected as material for re-seeding the finger reefs. Thirty-five of these fragments were carefully glued (using underwater epoxy) on to the old, dead reef structures of the finger reefs. They were photographed and measured, and we hope that they will survive and grow, thus improving the ecological and aesthetic value of these reefs.

How do gill parasites affect the lives of reef fish?

Scientists: Rachel Finley, PhD, Graham Forrester, PhD, and Grace Lentini

Dr. Finley based her PhD research on the lives of gobies, a type of small fish, which live at the edge of Guana's reefs. These gobies are often infected by a parasitic copepod that attaches to a fish's gills. Through her studies on Guana, Dr. Finley determined the specific effects of this parasite on the longevity of gobies (parasitized gobies die

younger), and has investigated the mechanism by which the parasites spread through fish populations.

How prevalent are fish parasites around Guana?

Scientists: Paul Sikkel, Ph.D., Donna Nemeth, Ph.D., and Amber McCammon

This group studied the abundance of parasitic flatworms (monogeans) in two closely related reef fish—Ocean Surgeonfish and Blue Tangs—around the Virgin Islands. They found about half of all Blue Tangs, but very few Ocean Surgeonfish, were infected but by monogeans. They will continue to investigate differences in the habits of these fish species that may explain the observed differences in parasite loads.

Interestingly, Dr. Sikkel and his team also found that the rate of parasitic infection in Blue Tangs around Guana was higher, both in the proportion of infected fish and the number of parasites per fish, than their other study sites around St. Thomas and St. John. Rachel Petrik, in previous years while searching the BVI for gobies with gill parasites, similarly found parasite loads to be highest at Guana. We do not yet know the reason for this pattern.

How does the abundance of fish on Guana's reefs compare with that in the Virgin Islands National Park on St. John, USVI.

Scientists: Ralf Boulon and Thomas Kelley

Years of monitoring reef fish populations in the VI National Park on St. John have shown a declining abundance of edible species. Many people believe that the abundance of these threatened fish populations increases as one looks farther east in the Virgin Islands. Dr. Boulon and Mr. Kelley set out to test this belief by applying their survey techniques to the reefs around Guana. They found that the populations of fish on Guana's reefs are similar to those on St. John, except that Guana's reefs appear to be more diverse. They counted 10 more species of fish on Guana than on St. John!

Are West Indian Topsnails endangered by overcollection?

Scientists: Tom Good, PhD, Graham Forrester, PhD, Linda Forrester, Rachel Finley, PhD, and Grace Lentini

The West Indian Topsnail (locally known as whelk) is a delicacy that draws a higher price than lobster or conch in the BVI. These snails are threatened by over-collection on many islands in the Caribbean. Initiated by Dr. Tom Good several years ago, this study seeks to identify how fishing pressure affects the size distribution of individuals making up whelk populations in the BVI. The proportion of large topsnails to small ones is an important indicator of potential reproduction because the capacity to produce eggs increases exponentially with linear increases in body size. The information produced in this study will also be used by Rob Power, at the H. Lavity Stoutt Community College, who is investigating the viability of using topsnails in mariculture.

Education

2004

Summer Youth Program of the Conservation and Fisheries Department and the National Parks Trust

Forty children and eight supervisors visited Guana between July 28th and 29th, 2004 (Figure 3). Dr. Lianna Jarecki guided these children through a field study of beaches and salt ponds during their visit. At White Bay and North Beach, students learned about the importance of beach vegetation, how sand is formed, and they viewed some of the marine life adapted to life in the surf zone. At the salt pond, the children saw flamingos (many for the first time) and other wetland birds, identified mangroves, caught and studied fiddler crabs, and captured aquatic insects in the pond with hand nets.

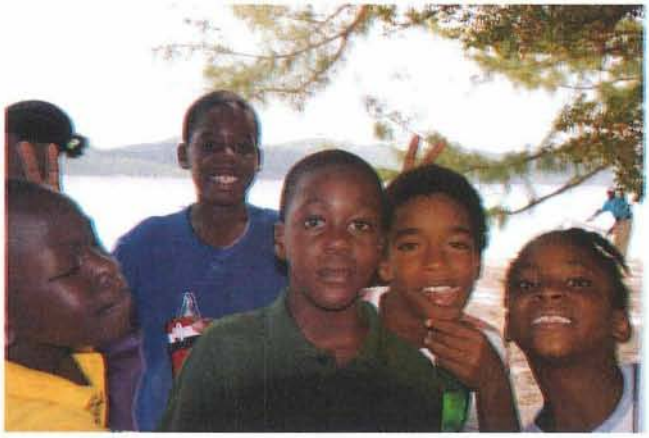

Figure 3: Summer youth program, day on Guana.

Guana provided transportation from Trellis Bay and also provided a beach lunch for the students. On the return trip, the boat toured the mangrove area between Tortola and Beef Island so that the students could compare a mangrove lagoon community to the community they studied at Guana's salt pond.

Tropical Field Ecology, an undergraduate course from Texas Tech University

Gad Perry, PhD, taught this course at the H. Lavity Stoutt Community College from July 15 to July 30th, and on July 27th the students and instructors visited Guana to study examples of species and habitat restoration efforts. This course was also attended by three local people representing the Conservation and Fisheries Department, the National Parks Trust and the Environmental Health Department.

2005

Environmental Youth Program

Dr. Lianna Jarecki gave an environmental presentation to about 50 local children in a local youth program run by the Conservation and Fisheries Department. This took place on Tortola.

Experiential science for high school students

We hosted a local chemistry teacher, his wife, and three high school students from July 25th to 29th, 2005. This special program for students interested in nature was organised by Robert Chalwell and led by Goeff Gordon, a school teacher in Jost van Dyke. The students learned about the MSM projects at mealtime discussions; Christina Leahy, a behavioural biologist and ornithologist, gave the students a half-day bird tour; and our assistants, Sam and Che, taught snorkelling skills and showed the students the White Bay reef mapping project. The students also spent time learning about terrestrial biology with their teacher.

Small-scale field experiments accurately scale up to predict density dependence in reef fish populations at large scales

Mark A. Steele*†‡ and Graham E. Forrester†§

*Marine Science Institute, University of California, Santa Barbara, CA 93106; and §Department of Natural Resources Science, University of Rhode Island, Kingston, RI 02881

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Field experiments provide rigorous tests of ecological hypotheses but are usually limited to small spatial scales. It is thus unclear whether these findings extrapolate to larger scales relevant to conservation and management. We show that the results of experiments detecting density-dependent mortality of reef fish on small habitat patches scale up to have similar effects on much larger entire reefs that are the size of small marine reserves and approach the scale at which some reef fisheries operate. We suggest that accurate scaling is due to the type of species interaction causing local density dependence and the fact that localized events can be aggregated to describe larger-scale interactions with minimal distortion. Careful extrapolation from small-scale experiments identifying species interactions and their effects should improve our ability to predict the outcomes of alternative management strategies for coral reef fishes and their habitats.

spatial scaling

Field experiments provide direct and compelling tests of ecological hypotheses and yield important insights into the nature of species interactions. Despite their scientific rigor, logistics typically constrain field experiments to plots no more than a few square meters in area (1, 2). Conservation and resource management generally occur in areas that are orders of magnitude larger than most experimental plots, so it is vital to determine whether the results of field experiments apply at larger spatial domains. Using small-scale field experiments, we tested empirically whether density dependence detected in reef fishes scales up to exert similar effects over larger areas. We chose to study this topic because density-dependent feedback in population growth determines the long-term stability of populations and their response to exogenous perturbations (3). Defining the dynamical effects of density dependence and identifying its underlying biological causes are, therefore, of both theoretical and practical importance.

Manipulations of population density provide the most direct, rigorous, tests for density dependence. Coral reef fishes are excellent subjects for these tests because they are easily observed and manipulated *in situ*. Density manipulations often reveal strong density-dependent mortality of reef fishes, usually of recently settled juveniles (settlement occurs when planktonic larvae take up permanent residence on a reef) (4, 5). Recent cross-factored manipulations of both population density and putative biological causes of density dependence are particularly informative and have successfully identified the competitive and predator–prey interactions responsible for density-dependent mortality (6–11). Despite their scientific rigor, these experiments have all been performed on tiny patches of habitat (<10 m^2 in area), so whether their results can inform decisions about conservation and fisheries management is questionable. We tested whether density-dependent mortality detected on small habitat patches scales up to have equivalent effects on entire reefs (thousands of m^2 in area). Extrapolating to entire reefs of this size is of practical interest because they match the size of smaller marine sanctuaries and approach the scale at which reef fisheries operate.

We studied the bridled goby (*Coryphopterus glaucofraenum*), a small site-attached fish common throughout the wider Caribbean. Bridled gobies occupy reefs where sand and coral are interspersed because they feed on invertebrates in the sand but seek refuge from larger predatory fishes in crevices at the base of rock and coral. Field experiments show that bridled gobies suffer strong density-dependent mortality on small habitat patches (12). Additional experiments show that this density dependence occurs because, as they become crowded, gobies experience a shortage of the crevices they use as refuges and so become increasingly susceptible to predation (7, 11). Many other reef fishes use structural features of reefs as refuges, and increased predation due to limited shelter may be a common source of density-dependent mortality in reef fishes (10, 11, 13).

Methods

Density Dependence on Entire Reefs. We performed a large-scale study of goby populations over 5 years (1998–2001 and 2003) on five entire reefs spread over 25 km on the Great Bahama Bank near Lee Stocking Island, Bahamas (Fig. 2, which is published as supporting information on the PNAS web site). Each reef contained 3,000–15,000 m^2 of habitat suitable for gobies, a mix of stony corals, gorgonians, sponges, limestone rock, and sand. As is true of most reef fishes, larval gobies settle to reefs nightly throughout a long reproductive season (June through September for bridled gobies), and many die within days of arrival. Accurate estimates of settlement thus require daily monitoring of newly settled fish, which has been prohibitively time-consuming over large areas. To overcome this problem, we developed methods allowing us to monitor settlement throughout the reproductive season at our five sites (14). Further details of methods used to estimate settlement are in the *Supporting Text*, which is published as supporting information on the PNAS web site. To test for density dependence, we tested whether mortality of gobies settling during summer was related to the density of settlers averaged over the entire summer. The actual density experienced by settlers at a site depends on when during the summer they settle and how long they survive, but this index of density provides a simple time-averaged measure of the initial density of a year-class.

Estimating goby mortality after settlement was simplified by the isolation of the five reefs. Each was >100 m from any other suitable habitat and, because gobies do not move among reefs this isolated after settlement, we could infer that any goby disappearing from a reef had died. Mortality estimates also were facilitated by the gobies' short lifespan and the availability of

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†M.A.S. and G.E.F. contributed equally to this work.

‡To whom correspondence should be addressed. E-mail: steele@lifesci.ucsb.edu.

preverified methods for aging this species. Gobies become reproductively mature roughly 3 months after settlement, at about 25 mm in length. Few adults survive from one summer to the next, and this species is effectively an annual. Therefore, we estimated mortality from the date of settlement until late October, the time when each generation reaches its peak adult abundance. Divers counted gobies on the reefs in late October and estimated their body lengths visually. Length estimates were converted to estimates of age, which were used to determine which of the fish present in October had settled during the preceding summer. Mortality was calculated by using the weekly summer counts of settlers and the October count of survivors, assuming that the probability of mortality after settlement was constant and identical for all settlers at a given site in a given year. Further details of methods used to estimate mortality are published in *Supporting Text*.

Because density dependence in bridled gobies is caused by a shortage of refuges, divers estimated the fraction of substratum covered by live and dead coral and small rocks, which, as we showed in ref. 11, is an index of the density of refuges. Further details of methods used to estimate refuge density are published in *Supporting Text*.

Density Dependence on Small Habitat Patches. We also compared the strength of spatially density-dependent mortality on entire reefs to the strength of spatial density dependence measured on small habitat patches. Four separate small-scale studies were used for this comparison. The first two studies were manipulations of goby density on small patch reefs near Guana Island, in the British Virgin Islands. Details of the first manipulation were published in ref. 12, and the second manipulation used nearly identical methods. The two remaining small-scale studies were done during summer and fall 1997 at two of our Bahamian reefs (Rainbow and Windsock). These studies were observational and tested for a correlation between settler density and mortality in the first week after settlement on small habitat patches. Further details of the methods used in the small-scale Bahamian studies are published in *Supporting Text*.

Results and Discussion

Accurate Scaling-Up of Spatial Density Dependence. We first tested for density dependence in space on entire reefs (mortality related to differences in density among sites), because past local-scale experiments on reef fishes all tested for spatial density dependence (4, 5). We used an analysis of covariance model that included terms for effects of settler density (a covariate), differences among years (a categorical factor), and the interaction between the two to test whether mortality was spatially density-dependent. The strength of density-dependent mortality, measured as the slope of the relationship between mortality and settler density, did not change appreciably among years ($F_{4,15} = 0.72$, $P = 0.59$). With this interaction term removed from the model, mortality was shown to differ significantly among years ($F_{1,15} = 4.3$, $P = 0.02$). Most notably, however, mortality on entire reefs increased progressively with settler density ($F_{1,19} = 63.9$, $P < 0.0001$), indicating that spatial density dependence in bridled gobies was detectable on entire reefs (Fig. 1*A*). Not only was spatial density dependence observable at small and large scales, its strength (measured as the slope of a linear regression relating population density to instantaneous per-capita mortality) was similar at small and large spatial scales. Regression slopes from our study of five entire Bahamian reefs fell within the range of slopes measured in four separate studies on small habitat patches, despite differences in scope and methods among the studies (Table 1).

Temporal Density Dependence on Entire Reefs. We also tested for density dependence over time (mortality related to differences in density over time at a single reef). The distinction between spatial and temporal density dependence is important because one does not necessarily lead to the other, and only temporal density dependence can stabilize populations or facilitate their recovery from near extirpation (15). To test whether mortality was temporally density-dependent, we used an analysis of covariance model including terms for effects of settler density (a covariate) and differences among reefs (a categorical factor), and their interaction. Only the three reefs at which settlement varied appreciably among years (Goby Spot, Rainbow, and Windsock) were included in the analysis, although the results were qualitatively similar if the remaining two reefs were included. The slope of the relationship between mortality and settler density did not differ among reefs ($F_{2,6} = 0.61$, $P = 0.566$) and, after removal of this interaction term, there was no detectable variation in mortality among reefs ($F_{2,11} = 1.14$, $P = 0.354$). Bridled goby mortality was, however, higher in years when the density of settlers was highest ($F_{1,11} = 10.1$, $P = 0.009$) and so was temporally density-dependent (Fig. 1*B*).

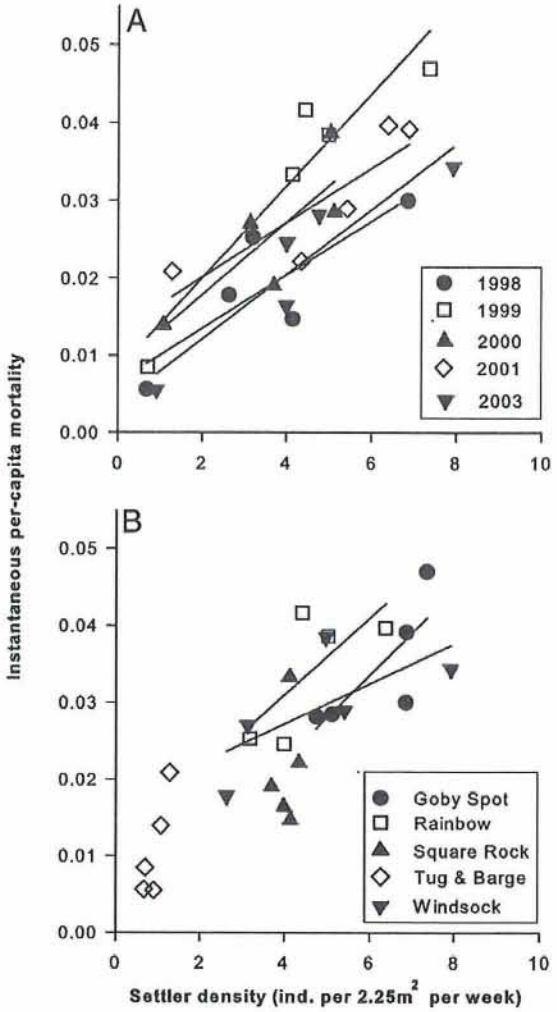

Fig. 1. Density-dependent mortality of bridled gobies on entire reefs. Data grouped by site reveal spatial density dependence (*A*), whereas data grouped by year show temporal density dependence (*B*). Regression lines are fit by using analysis of covariance and, in *B*, are omitted for two sites showing little interannual variation in settler density.

Hypotheses for Accurate Scaling. Why does density dependence scale up in bridled gobies? Theoretical considerations suggest

Table 1. The strength of spatially density-dependent mortality is similar at small and large spatial scales

Mean reef size, m^2	Year	Location and type of study	Focal age group for mortality estimate	No. of reefs or patches	B (±SE)
Entire reefs					
6,200	1998	Bahamas, observational	Settlers to adults	5	0.0035 (±0.0013)
6,200	1999	Bahamas, observational	Settlers to adults	5	0.0060 (±0.0011)
6,200	2000	Bahamas, observational	Settlers to adults	5	0.0047 (±0.0019)
6,200	2001	Bahamas, observational	Settlers to adults	5	0.0035 (±0.0011)
6,200	2003	Bahamas, observational	Settlers to adults	5	0.0042 (±0.0009)
Small habitat patches					
8.4	1994	British Virgin Islands manipulation	Sub-adults and adults	16	0.0035 (±0.0010)
8.5	1995	British Virgin Islands, manipulation	Adults	8	0.0023 (±0.0005)
2.25	1997	Bahamas, observational	Settlers	10	0.0038 (±0.0059)
2.25	1997	Bahamas, observational	Settlers	5	0.0060 (±0.0098)

The strength of density dependence is measured as the slope (B) of a linear regression relating population density (mean number per 2.25 m^2) to per-capita instantaneous mortality.

that density dependence should not scale up in situations where density dependence within small habitat patches interacts with spatial heterogeneity in density (16). Localized heterogeneity in fish density and habitat is obvious on coral reefs, and fish populations on entire reefs are well described as a collection of local patches. Distortion in density-dependent relationships will occur when aggregating the constituent local patches fails to describe the properties of the entire population (16). The degree of distortion, however, depends on the pattern of local heterogeneity in density, the functional form of the local relationship between density and mortality, and other details of demography at small scales. A simulation model tailored specifically to the demography of bridled gobies suggests that aggregation error should be minimal in this species and is thus in agreement with our empirical findings (17).

A second factor critical to the scaling of density dependence is its underlying cause at the local scale (18, 19). Local density dependence in bridled gobies is caused by a shortage of refuges from predation, and so the strength of density dependence at small scales is sensitive to changes in the local availability of refuges. The nature of refuge use in bridled gobies suggests that vulnerability to predation may be roughly approximated simply as the ratio of mean goby density to mean refuge density (11). We do not know whether density dependence has the same cause on entire reefs, but this is the most parsimonious explanation for our findings. Differences among reefs in our index of refuge density (15–33%; Table 2) were slight, compared with those needed to alter the strength of spatial density dependence on small habitat patches (1–58%, ref. 11). To test whether refuge density influenced the strength of spatial density dependence, terms for the effect of refuge density and all of its possible interactions were added to the original analysis of covariance model. Although overall goby mortality was slightly higher on reefs where refuges were sparse, refuge availability did not appreciably alter the slope of spatial relationships between settler density and mortality ($P > 0.2$ for all interaction terms). Limited aggregation error and perhaps the relative homogeneity of refuge density among entire reefs may thus explain why density dependence in bridled goby mortality scaled up accurately.

Table 2. Differences among five reefs in the availability of refuges for gobies

Reef	% hard substratum
Goby Spot	15.0 (7.0–23.0)
Rainbow	19.4 (14.1–24.6)
Square Rock	30.2 (21.8–38.5)
Tug and Barge	33.1 (18.9–47.1)
Windsock	21.0 (17.1–24.8)

Shown are means (with upper and lower 95% confidence intervals) of the percentage of the bottom covered by hard substratum (rock, coral, and rubble), which is an index of the density of refuges for gobies.

Data on mortality at equivalent small and large spatial scales are available for only one other species of reef fish, the lemon damsel (*Pomacentrus moluccensis*). Small-scale experiments confirm that juvenile lemon damsels occupying small habitat patches suffer density-dependent mortality (13) within days of settlement (20). In contrast, a large-scale study on the lemon damsel showed that adult abundance on entire reefs increased in direct proportion to the prior density of older juveniles, indicating that mortality of older juveniles and adults was effectively density-independent (21). The large-scale study, although impressive and unparalleled in scope, was not designed to measure mortality in the first days or weeks after settlement, and therefore it cannot be used to ascertain whether density dependence detected in young juvenile lemon damsels scales up.

Implications for Conservation and Management. The fact that we can extrapolate from experiments on bridled gobies is encouraging because it suggests that small-scale manipulations may be used to inform fisheries management. Analyses of large-scale survey and catch data show that temporally density-dependent mortality is common in commercially harvested demersal (bottom-oriented) fishes but usually occurs only in small juveniles (22). Despite its importance for the stability of these populations under harvesting and other anthropogenic perturbations, we know little about the underlying interactions responsible for temporal density dependence at the juvenile stage. Although the bridled goby is not exploited by humans and is distinguished from many exploited species by its small adult size, limited home range, and short life cycle, there are hints that small juveniles of some harvested species are vulnerable to predators and use structural habitat features in ways reminiscent of bridled gobies (22, 23). Identifying the type of species interaction responsible for density dependence may help devise effective management strategies for specific stocks (24). Different strategies are suggested, for example, if density dependence among juveniles is caused by a shortage of habitat that provides shelter from predators than if it results

from competition for food. Focused experimentation at local scales should allow us to better define the causes of density dependence in juveniles of harvested species, and our results illustrate that careful extrapolation of the results to larger spatial domains is possible.

Our results also demonstrate that density-dependent mortality occurs at spatial and temporal scales relevant to fisheries and marine reserves on coral reefs. No-take reserves are rapidly gaining favor as a tool for conservation and fisheries management. Fishing usually targets adults of larger species and so, once fishing is halted, adult populations of these species often build quickly within reserves (25). Long-term benefits to the fishery are anticipated once the offspring produced by these protected brood stocks begin subsidizing populations outside the reserve. The adults of many harvested species are generalist piscivores that prey on a variety of smaller fishes, and they are often the agents of density-dependent mortality in those smaller fishes. As adult populations of piscivores build within reserves, the increase in density-dependent predation that they inflict may fall partly on their own juveniles and so may offset some of the anticipated benefits of protection from harvesting. Increased density-dependent predation on unexploited species like bridled gobies also could have unexpected community-wide influences (26). More mechanistic studies on how fished and unfished species respond to building population densities are thus needed to evaluate the long-term success of marine reserves.

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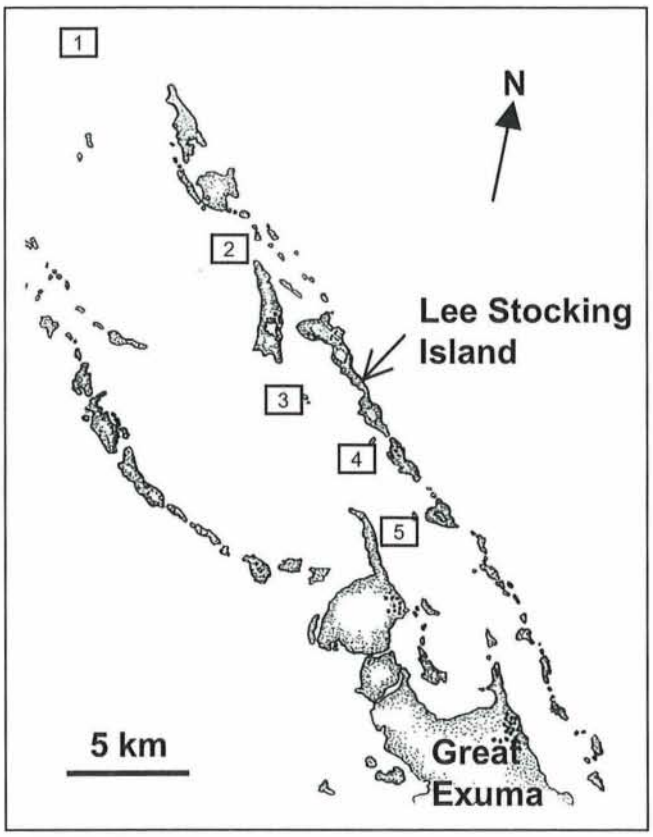
Supporting Fig. 2. Map showing our five Bahamian study sites: (1) Goby Spot, (2) Rainbow Reef, (3) Tug & Barge, (4) Windsock Reef, (5) Square Rock. Our base for fieldwork was the Caribbean Marine Research Center on Lee Stocking Island.



Supporting Text

Additional Details on Methodology

Measuring Goby Settlement on Entire Reefs. We measured settlement at each site over several consecutive weeks during the summer of each year. Within each year, all sites were sampled over the same time period, but the duration and precise timing of the sampling period varied from year to year (duration: 7–12 weeks; start dates: June 7–July 9; end dates: Aug. 23–Sept. 7). We measured settlement during summer because year-round monitoring showed that little settlement occurs at other times of year. Settlement was measured in five 1.5 × 1.5 m plots of natural habitat at each reef. The habitat on these plots was unaltered, except that it was caged to protect the recently settled gobies from predators, which are the primary cause of death of young gobies and can rapidly distort goby settlement patterns (1). Cages were built of plastic netting on poly(vinyl chloride) pipe frames. The mesh size (5 mm) was large enough to allow settling gobies to pass through but small enough to exclude virtually all predators. In previous work, we found no evidence that the cages themselves influenced settlement rates of the gobies (1). The settlement plots were located in a stratified random fashion. They were placed at random within 10-m-long sections of reef, which were themselves spaced by 40–50 m.

Scuba divers used hand nets to collect settling gobies that accumulated on the plots each week. Earlier work showed that weekly collections of recent settlers from caged areas accurately captured cumulative daily patterns of settlement (1). After collection, gobies were measured in the lab, and recent settlers (fish that had settled in the preceding week) were distinguished from older residents by their size, using known size–age relationships (ref. 1 and M.A.S., unpublished data).

Measuring Goby Mortality on Entire Reefs. To test whether mortality after settlement was density-dependent, we censused the reefs in late October to see how many of the summer's settlers had survived to this date. By late October, gobies that had settled during the summer had reached 20–40 mm in length. Many of the gobies were already

sexually mature by this time, because maturity is reached at about 25 mm (M.A.S., unpublished data, and refs. 2 and 3). To estimate goby density, divers counted gobies within quadrats that were distributed at each site by using a stratified, random design. From 1998 to 2001, we sampled 25 1.5 × 1.5 m quadrats per site. Quadrats were located by placing five 50-m-long transects so that they were roughly equally spaced within the goby habitat at the site. Each transect was divided into five 10-m-long segments, and a quadrat was placed at a random distance from the start of each segment. In 2003, we sampled 4 × 4 m quadrats at each site. Quadrats were placed by dividing the site into segments and placing one quadrat at a randomly located position within each segment. Sixteen quadrats were sampled at Goby Spot, Rainbow, and Windsock, and 8 quadrats were sampled at Square Rock and Tug and Barge. In each quadrat, all gobies were counted and assigned to one of seven size classes, each of which was 5 mm wide. Counts were made by a single diver throughout the entire study (M.A.S.), and visual estimates of size were typically within 2 mm of actual sizes, based on field tests in which gobies were captured and measured after their length had been estimated visually (M.A.S., unpublished data).

We converted the measured size structure to age structure by using statistical relationships between size and age. We measured age by using rings that form daily in the otoliths. We verified that these rings form daily by performing a field mark–recapture study of recent settlers whose otoliths had been chemically labeled (M.A.S., unpublished data and ref. 4). Using collections of at least 60 gobies per site, we developed statistical relationships between body size (length in millimeters) and age in days for each site, based on counts of daily growth rings in the otoliths (M.A.S., unpublished data). Using these otolith-based relationships between size and age, we converted the size classes from the autumn censuses into age classes. We were then able to determine, based on their estimated age, which of the fish present in late October had settled during the preceding summer's settlement monitoring.

We estimated goby mortality from the day of settlement to the date of the October census, making separate estimates for each year at each site. The calculation focused on

cohorts or groups of individuals that arrived at a given site as settlers during the same week (x = the number of settlers in a cohort). For simplicity, all settlers were assumed to have arrived exactly at the midpoint of the week (so t_i is the time in days on which the ith cohort settled, and t_0 is the beginning of summer). The number of cohorts (i_{max}) is the number of weekly collections. To estimate the instantaneous per capita mortality rate (m) from our data, we assume that individuals of all ages experienced the same mortality rate during the entire time interval from the date on which settler collection began until the final autumn population count (X = the number of surviving individuals and T = the day of the autumn census).

Under the assumption of constant mortality, the number of individuals that settled on day t_i and also remained alive on day T is

$$x_{t_i} e^{-(T-t_i)m},$$

and consequently

$$X_T = \sum_{i=1}^{i_{max}} x_{t_i} e^{-(T-t_i)m}.$$

The preceding equation was solved iteratively to estimate m.

Measuring an Index of Refuge Density on Entire Reefs. Because density dependence in bridled gobies is caused by a shortage of refuges, each year a diver estimated an index of refuge availability for bridled gobies within a set of 25 plots (each 1.5 × 1.5 m) that were located in a stratified random fashion at each site. The diver hovered above each plot, and made a visual estimate of the fraction of area covered by different types of substratum: live coral, sand, sea grass, limestone pavement, rubble (pieces of rock <25 cm in diameter), rocks and dead coral (>25 cm in diameter), and other substrata. The

combined percent cover of live coral, rock/dead coral, and rubble was used as a measure of the density of crevices used by gobies to shelter from predators (5).

Density Dependence on Small Habitat Patches. We used four separate small-scale studies to compare the strength of spatially density-dependent mortality on entire reefs to that on small habitat patches (6). Two of these studies were done at Rainbow and Windsock reefs in the Bahamas. For both of these studies, we collected recently settled gobies on a set of paired plots (2.25 m^2) for 3 weeks by using methods described in ref. 1. Settlers were collected from both types of plot weekly. One plot of each pair was caged to exclude predators, so the number of recently settled gobies collected per week (S) is thus an estimate of the actual settlement rate. The second plot in each pair was uncaged, and so the number of settlers collected each week (R) reflects the number that settled minus the number killed by predators before collection. Assuming that actual settlement to both plots in a pair was identical, we estimated the daily instantaneous per capita mortality rate (m) of the gobies on each pair of plots during each week as:

$$m = \frac{(\ln(S) - \ln(R))}{7}.$$

For each pair of plots, goby density and mortality were then averaged across the 3 weeks sampled. For all four small-scale studies, we then calculated linear regressions relating mean goby density (number per 2.25 m^2) to instantaneous per-capita mortality. The strength of density dependence (the slope of the regression line) in each of the small-scale studies was thus directly comparable to strengths calculated for entire reefs.

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NOTES

DISTRIBUTION AND ABUNDANCE PATTERNS IN CARIBBEAN ROCKY INTERTIDAL ZONES

Thomas P. Good

Investigations of temperate rocky intertidal zones have identified patterns of distribution and abundance of organisms and mechanisms responsible for the observed patterns. Models for the organization of rocky shore communities (Menge and Sutherland, 1976; 1987) predict that in areas of reduced environmental stress, such as low wave action, the relative influence of predation and competition on organism distribution and abundance increases. With increasing wave action, predation becomes progressively less important relative to physical factors, and competition among sessile taxa becomes increasingly important, particularly in areas of high recruitment. Finally, at high wave action sites, physical factors impact organisms directly or indirectly, thus decreasing the importance of biological factors.

Tropical intertidal zones have been described as more physically benign than their temperate counterparts with respect to wave exposure and winter stresses (Menge and Lubchenco, 1981; Brosnan, 1992). Examination of intertidal zones on the Pacific coast of Panama found predation to predominate. Intense consumer pressure from mollusks, crabs, and fishes combined with low recruitment resulted in low biomass of sessile taxa, dominance of crustose algae, and indistinct vertical zonation (Menge and Lubchenco, 1981; Menge et al., 1983; 1985; 1986a,b; Lubchenco et al., 1984; Menge, 1991).

Generalizing from these studies to a global temperate-tropical model of the organization of rocky shore communities (Brosnan, 1992) is problematic. First, tropical intertidal zones may actually be more stressful than temperate shores, particularly in terms of temperature and desiccation stress (Moore, 1972). Furthermore, substantial variability among tropical locations exists. Wave-induced stresses appear to be more important in Costa Rica than in nearby Panama (Ortega, 1986). Also, the eastern shorelines of both countries, as well as the rest of the Caribbean Sea, experience very different physical regimes than those on the Pacific coast. Tidal amplitudes on the Pacific coast are ~ 6 m (Menge and Lubchenco, 1981), several times greater than on the Caribbean coast, which could affect intertidal distribution and abundance patterns.

Rocky shores in the Caribbean Sea have been described (Stephenson and Stephenson, 1950; Lewis, 1960; Voss and Voss, 1960; Brattstrom, 1980; Britton and Morton, 1989; Dawes et al., 1991), but not assessed quantitatively. Here, I report on patterns of distribution and abundance of sessile and mobile taxa in rocky intertidal zones at sites differing in wave exposure in the Caribbean Sea.

METHODS

The study was conducted in August 1999 and 2000 on Guana Island, a small (340 ha.) privately owned island located in the British Virgin Islands (B.V.I.; Fig. 1). Six sites similar in topography—sloping ledges adjacent to sandy beaches—were selected to define intertidal conditions observed around the island. Northern and eastern shore sites (Long Point, North Bay, Bigelow Beach) are exposed to prevailing easterlies, whereas southern shore sites (Harris Ghut, White Bay, Crab Cove) are protected from winds (Fig. 1). Maximum wave-force dynamometers (Bell and Denny, 1994) detected force readings (mean Newtons ± SE) at exposed sites (8.8 ± 1.2) sig-

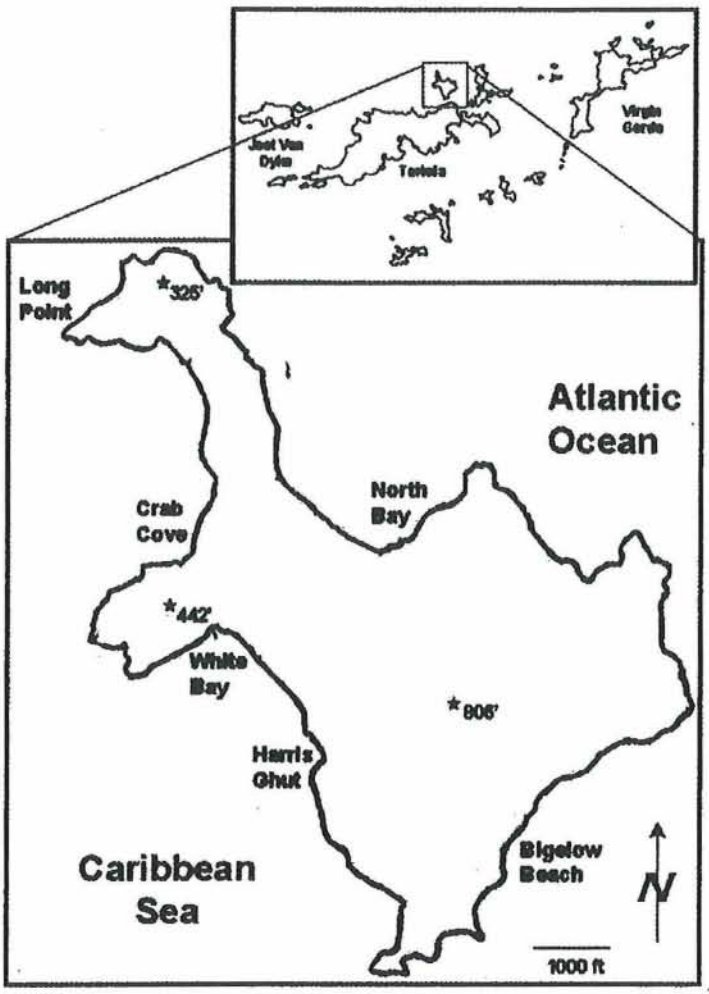

Figure 1. Map of the British Virgin Islands (B.V.I.) in the central Caribbean Sea, showing the location of Guana Island, and the study sites. Numbers next to stars represent elevation of local peaks in feet.

nificantly greater than at protected sites (4.1 ± 0.2; Mann-Whitney U = 40.0; P = 0.005). Tides are semi-diurnal and tidal amplitudes are small (0.5–1.0 m). Annual water temperatures in the B.V.I. vary between 24.4° and 28.9°C, and annual air temperatures vary from 22–31°C.

Guana Island's intertidal flora and fauna are typical of Caribbean Sea rocky shores, with narrow (< 1 m) zones characterized by the dominant taxa. The upper intertidal zone is typified by barnacles and erect green algae, while the lower intertidal zone is typified by crustose and turf algae. Mobile taxa include gastropods, limpets, chitons, crabs, urchins, and fish. In 1999, patterns of algal and invertebrate abundance and distribution were quantified at each site using 6–10, 25 × 50 cm quadrats placed at random meter intervals along a 30 m transect parallel to shore in each of two intertidal zones during low tides. For each quadrat, percent cover of sessile taxa was determined from 48 point-intercepts, and mobile taxa were counted. The distance from 12 random points in the quadrat grid to the substratum was measured to the nearest millimeter, and substratum heterogeneity indices (mean substratum depth, proportion of measurements > 3.0 cm, coefficient of variation) were calculated from these data for each site (Lubchenco et al., 1984). In 2000, West Indian topshells (*Cittarium pica*) and predatory gastropods were counted in three, 10 × 1 m band transects in the lower intertidal zone at all sites.

Nested ANOVA were used to test the effect of wave exposure (fixed) and site (random) within exposure on percent cover of sessile taxa (erect algae and sessile invertebrates), density of mobile

taxa, and species richness for both upper and lower tidal zones. As intertidal zones were defined by taxon differences, such as crustose algae in the lower zone, comparisons between zones were limited to t-tests. T-tests compared densities of mobile consumers between protected and exposed shores. Where data transformation did not achieve homogeneous variances, non-parametric Mann-Whitney U-tests were used. Regression analyses tested the relationship of measures of species richness and diversity (S; Shannon-Weaver H′) as a function of substratum heterogeneity indices per plot (Menge et al., 1985).

Results

The percent cover of sessile taxa varied between wave exposed and protected shores in the lower zone, but not the upper zone (Table 1). In the upper zone, bare space averaged 56% cover of the substratum, while crustose algae (*Lithothamnion* spp.), sessile invertebrates (barnacles *Chthamalus fragilis* Darwin; vermetid snails), and erect algae such as *Cladophora prolifera* (Roth) Kuetzing, *Chaetomorpha aerea* (Dillwyn) Kuetzing averaged 44% cover. Percent cover (mean ± SE) of sessile taxa (erect algae and invertebrates) was similar at exposed (39.7 ± 6.3) and protected shores (35.8 ± 2.4; Fig. 2A; Table 1A), although it varied among sites within shore exposure (Fig 2A; Table 1). In the lower zone, by contrast, bare space averaged only 2% cover of the substratum, while crustose algae, barnacles, and erect algae such as *Bryopsis plumosa* (Hudson) C. Agardh, *Caulerpa racemosa* (Forsskal) J. Agardh, *Sargassum polyceratium* Montagne, *Padina gymnospora* (Kuetzing) Sonders, *Turbinaria turbinata* (Linnaeus) Kuntze, *Turbinaria tricostata* Barton, *Laurencia papillosa* (Hudson) Lamouroux, *Dictyota mertensii* (Martius) Kuetzing, *Acanthophora spicifera* (Vahl) Boergeson and *Amphiroa brasiliana* Decaisne averaged 98% cover. Percent cover (mean ± SE) of sessile taxa was greater at exposed shores (74.0 ± 4.4) than at protected shores (26.0 ± 5.4; Fig. 2B; Table 1), but varied little among sites within shore exposure (Table 1).

Densities of mobile taxa also varied between exposed and protected shores in the lower zone but not the upper zone (Table 2). In the upper zone, densities (mean ± SE) of herbivorous gastropods as *Littorina ziczac* (Gmelin, 1791), *Littorina mespillum* (Muhlfeld, 1824), *Nerita peloronta* Linnaeus, 1758, *Nerita tesselleta* Gmelin, 1791, *Nerita versicolor* Gmelin, 1791, *Fissurella* spp., *Acmaea leucopleura* Gmelin, 1791, *Acmaea antillarum* (Sowerby, 1831), *Chiton tuberculatus* Linnaeus 1758, *Acanthopleura gran-*

Table 1. Nested ANOVA of wave exposure and site within exposure on percent cover of sessile taxa (erect algae and sessile invertebrates) in the upper ($r^2 = 0.80$) and lower ($r^2 = 0.75$) tidal zones, respectively.

Upper tidal zone					
Source	SS	df	MS	F	P
Exposure	0.04	1	0.04	0.1	0.8
Site (exposure)	1.2	4	0.4	28.7	< 0.001
Error	0.4	29	0.02		
Lower tidal zone					
Source	SS	df	MS	F	P
Exposure	2.4	1	2.4	135.3	< 0.001
Site (exposure)	0.1	4	0.02	0.6	0.7
Error	0.9	30	0.03		

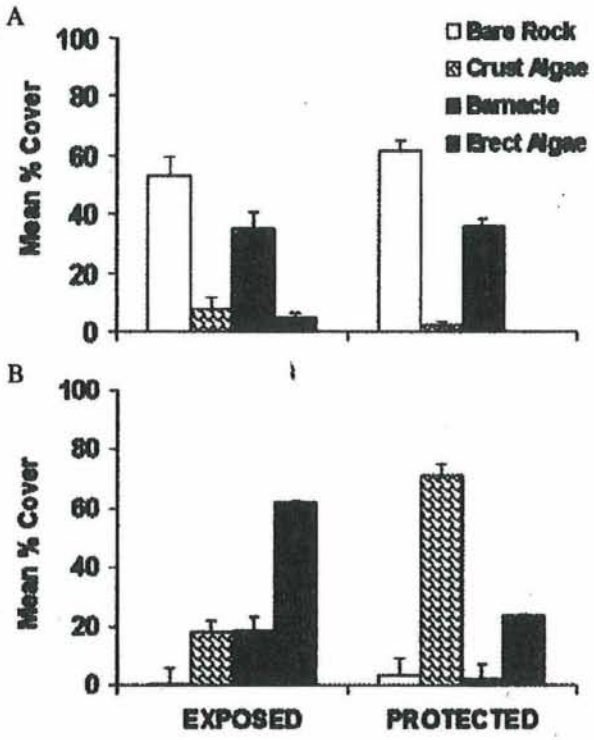

Figure 2. Percent cover (mean ± SE) of bare space, barnacles, crustose algae, and erect algae in (A) upper intertidal zones and (B) lower intertidal zones at protected and exposed shores on Guana Island.

ulata (Gmelin 1791), *Ceratozona squalida* (C. B. Adams, 1845), and *Cittarium pica* (Linnaeus, 1758) were similar at exposed and protected shores, although they varied among sites within shore exposure (Fig 3B; Table 2A). Densities of the predatory gastropods *Purpura patula* (Linnaeus, 1758), *Stramonita haemastoma floridana* (Conrad, 1837), *Stramonita rustica* (Lamarck, 1822), and *Thais deltoidea* (Lamarck, 1822) were similar at exposed and protected shores (Fig 3A; U = 174, P = 0.056). Densities of the crabs *Pachygrapsus transverses* (Gibbes, 1850) and *Microphrys bicornutus* (Latreille, 1825) were similar at exposed and protected shores (Fig 3A; U = 143, P = 0.75), as were

Table 2. Nested ANOVA of wave exposure and site within exposure on herbivorous gastropod density (indiv. quadrat^{-1}) in the upper ($r^2 = 0.55$) and lower ($r^2 = 0.72$) tidal zones, respectively.

Upper tidal zone					
Source	SS	df	MS	F	P
Exposure	59,607.9	1	59,607.9	1.4	0.3
Site (exposure)	175,219.9	4	43,804.9	5.8	0.001
Error	218,572.4	29	7,536.9		
Lower tidal zone					
Source	SS	df	MS	F	P
Exposure	109,933.6	1	109,993.6	18.4	0.01
Site (exposure)	23,911.9	4	5,977.9	3.6	0.02
Error	48,672.9	29	1,678.4		

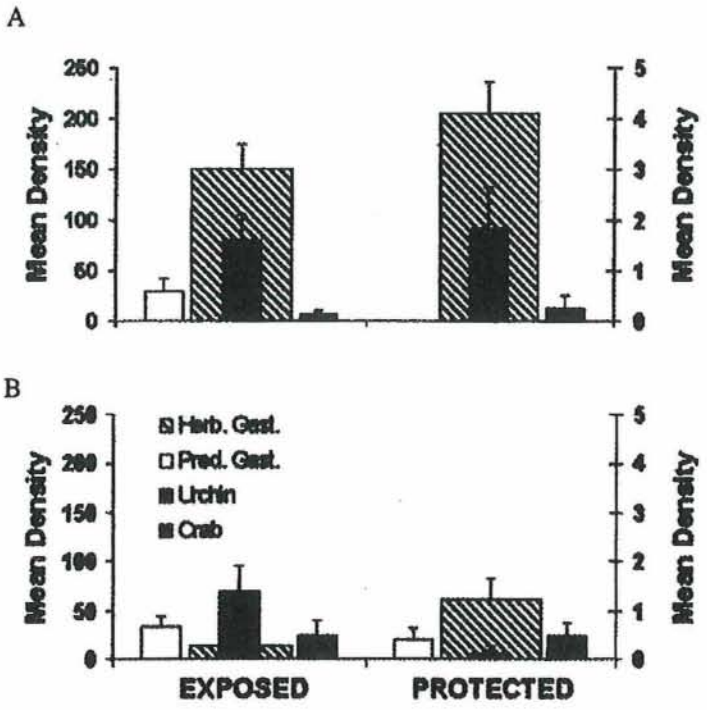

Figure 3. Density (mean number ± SE) of mobile consumers per quadrat in (A) upper intertidal zones and (B) lower intertidal zones at protected and exposed shores on Guana Island. The left axis corresponds to herbivorous mollusks density (striped bars), while the right axis corresponds to carnivorous mollusks, urchins, and crab density.

densities of urchins *Echinometra lucunter* (Linnaeus, 1758; Fig 3A; U = 132, P = 0.81). Fishes, primarily clingfish species *Tomicodon fasciatus* (Peters, 1860), *Acyrtus rubiginosus* (Poey, 1868), *Arcos macrophthalmus* (Günther, 1861), were too rare in quadrats to analyze quantitatively. In the lower zone, herbivorous gastropod densities were greater at protected shores, although densities varied among sites within shore exposure (Fig. 3B; Table 2). Predatory gastropod densities were similar at exposed and protected shores (Fig 3B; $t_{33} = 0.96$, P = 0.3). In 2000, densities (mean 10 m^{-2} ± SE) of *P. patula* were greater at exposed shores (0.6 ± 0.11) than at protected shores (0.03 ± 0.10; $t_{25} = 3.8$; P < 0.001); *Stramonita* and *Thais* spp. densities were similar at exposed (0.15 ± 0.06) and protected shores (0.19 ± 0.05; $t_{25} = 0.45$, P = 0.7). Densities of crabs were similar at exposed and protected shores (Fig 3B; U = 146, P = 0.61), but densities of urchins were greater at exposed shores (Fig 3B; U = 202, P = 0.05). Clingfish were again too rare in quadrats to analyze quantitatively.

Species richness (mean number $quadrat^{-1}$ ± SE) was greater in the lower zone (11.8 ± 0.5) than in the upper zone (7.3 ± 0.5; $t_{69} = -6.5$; P < 0.001). This was true for both sessile (6.9 ± 0.3 in lower; 3.7 ± 0.3 in upper; $t_{69} = -6.9$; P < 0.001) and mobile species (4.8 ± 0.4 in lower; 3.5 ± 0.3 in upper; $t_{69} = -2.6$; P = 0.01). Species richness did not differ between exposed and protected shores, or among sites within shore exposure in the upper or lower zone (Table 3); the same pattern held if sessile and mobile species were analyzed separately. Sessile species richness was not explained by mean depth measurements (D_h) ($r^2 = 0.01$, $F_{1,40} = 0.4$, P = 0.5) or the proportion of depth measurements > 3.0 cm (P_h) ($r^2 = 0.02$, $F_{1,40} = 0.9$, P = 0.3). Species richness of mobile taxa was not correlated with mean depth

Table 3. Nested ANOVA of wave exposure and site within exposure on species richness (number of species) of sessile and mobile taxa in the upper ($r^2 = 0.16$) and lower ($r^2 = 0.13$) tidal zones, respectively.

Upper tidal zone					
Source	SS	df	MS	F	P
Exposure	2.7	1	2.7	0.3	0.6
Site (exposure)	37.6	4	9.4	1.3	0.3
Error	203.9	29	7.0		
Lower tidal zone					
Source	SS	df	MS	F	P
Exposure	11.4	1	11.4	1.1	0.4
Site (exposure)	40.9	4	10.2	1.0	0.4
Error	303.3	29	10.1		

measurements ($r^2 = 0.02$, $F_{1,40} = 0.8$, $P = 0.4$) or the proportion of depth measurements > 3.0 cm ($r^2 = 0.01$, $F_{1,40} = 0.5$, $P = 0.5$). The patterns for species diversity (H′) were similar to those for species richness.

Substratum heterogeneity, as measured by the extent and variation in substratum depth measurements, was low overall on Guana Island (Table 4). The ranges of mean depth (2.5–5.2 cm) indicate that substratum variability was similar in the two zones. In the upper zone, substratum heterogeneity, as measured by the standard deviation of depth measurements, did not differ between exposures ($F_{1,4} = 0.9$, $P = 0.4$) or sites within exposures ($F_{4,33} = 1.2$, $P = 0.3$). In the lower zone, the standard deviation of depth measurements did not differ between exposures ($F_{1,4} = 0.05$, $P = 0.8$) or sites within exposures ($F_{4,28} = 1.2$, $P = 0.4$).

Discussion

Patterns of percent cover of sessile taxa and densities of mobile taxa in the intertidal zone of Guana Island suggest that predation, as indicated by density of mobile consumers, may be inversely related to the environmental stress of wave action. Sessile taxa such as erect algae and barnacles dominated at wave-exposed sites, especially in lower zones, and mobile taxa were more abundant under the more benign environmental conditions at protected sites. Consumer pressure, as indicated by density of mobile taxa, varies between sites with different wave exposures. While consistent with predictions of the environmental stress model (Menge and Sutherland, 1976; 1987), the patterns at exposed shores were inconsistent with the characterization of tropical intertidal zones as being relatively benign environments (Menge and Lubchenco 1981; Brosnan 1992). At Guana Island, environmental stress in the form of wave action may indirectly influence variation in community structure as it does in temperate intertidal zones, and this influence may rival the effects of or act in concert with temperature and desiccation stresses in tropical intertidal zones.

Distinct vertical zonation patterns in the Caribbean also contrasted markedly with the indistinct patterns found in Panama (Menge and Lubchenco, 1981; Lubchenco et al., 1984). On Guana Island, the yellow upper and pink lower zones were apparent even from

Table 4. Substratum depth measurements (cm) in the upper and lower zones at sites on Guana Island.

	Protected			Exposed		
Upper tidal zone	White Bay	Crab Cove	Harris Ghut	Bigelow Beach	North Beach	Long Point
Mean ± 95% CI	2.5 ± 0.5	3.0 ± 0.5	3.2 ± 0.8	4.0 ± 1.1	3.8 ± 0.5	3.0 ± 0.6
Range of x's	1.0–4.7	2.7–3.5	2.3–4.2	2.5–5.5	3.3–4.5	1.8–4.4
CI range	0.9–5.5	2.3–4.8	1.2–6.1	1.8–7.3	2.6–5.3	1.3–5.7
CV	76.1	51.5	75.2	65.0	36.0	56.9
n_1, n_2[a]	3, 36	3, 36	3, 36	2, 24	3, 36	3, 36
Lower tidal zone	White Bay	Crab Cove	Harris Ghut	Bigelow Beach	North Beach	Long Point
Mean ± 95% CI	2.7 ± 1.0	4.0 ± 1.0	2.6 ± 0.7	2.5 ± 0.6	5.2 ± 1.2	4.9 ± 0.7
Range of x's	1.4–4.6	2.3–7.2	2.5–2.6	1.9–3.2	2.6–6.8	4.1–5.7
CI range	0.9–5.2	1.7–9.1	1.6–3.9	1.2–4.0	2.0–8.5	3.3–7.1
CV	75.0	76.2	66.5	53.1	68.6	40.1
n_1, n_2[a]	3, 36	3, 36	3, 36	2, 24	3, 36	3, 36

[a] n_1 = number of quadrats, n_2 = number of points over entire transect

a distance, as they are throughout the Caribbean Sea (Dawes et al., 1991). Upper zones were dominated by bare space and had low species richness, whereas lower zones had higher species richness and were dominated by erect algae and barnacles, particularly at exposed sites. Such algal dominance, particularly low on the shore, is observable along exposed shores throughout the British and U. S. Virgin Islands and has been described for wind and wave-exposed shores throughout the Caribbean (Lewis, 1960; Voss and Voss, 1960; Brattstrom, 1980; Britton and Morton, 1989).

On Guana Island, herbivorous mollusks were the conspicuous intertidal consumer guild, as is true of other tropical locations (John et al., 1992; Williams 1994). Although protected shores contained high densities of littorinid gastropods, limpets, and chitons, the largest gastropods in the system—the West Indian Topshell (*C. pica*) and the wide-mouthed rock snail (*P. patula*)—were found in higher densities at exposed sites, particularly lower on the shore. Despite being globose and up to 110 mm wide, *C. pica* are common at sites with at least moderate wave-action throughout the Caribbean Sea (Lewis, 1960; Brattstrom, 1980).

Other mobile taxa present in quadrats in low densities, but in greater densities in subtidal areas—large crabs, urchins, and fishes—may also contribute to the observed patterns (Menge et al., 1983; Lubchenco et al., 1984). Large mobile crabs such as *Grapsus grapsus* (Linnaeus, 1758), *Percnon gibbesi* (H. Milne-Edwards, 1853), and *Plagiusa depressa* (Fabricius, 1775), normally associated with areas of patchy or sparse algae (John et al., 1992), were common at protected sites on Guana Island but retreated to subtidal areas and were thus not sampled in quadrats. Subtidal areas dominated by urchin burrows were crustose algae "barrens" that reached into the intertidal zone along protected shores around the island. Juvenile and adult reef fish were also common at protected sites and may use subtidal reefs near the sampling areas as refuges (John et al., 1992). Tidepools at Long Point containing juvenile fish and urchins were completely dominated

by rustose algae and encrusting corals. The overall effect of highly mobile taxa on lower zones may be substantial given these intertidal areas are not far from subtidal zones.

Patterns of substratum heterogeneity differed considerably from those of Panama; indices of substratum heterogeneity at Guana Island were one-half to two-thirds of those calculated for Panama sites (Lubchenco et al., 1984). Such heterogeneity influences patterns of space occupancy and abundance (Reimer, 1976a,b) by influencing desiccation stress (Williams, 1993; 1994) and may influence predator-prey interactions (Bertness et al., 1981; Hay 1981). Prey refuges at Guana Island appear to be primarily provided by dense, turf-forming algae rather than the rocky substratum.

The environmental stress model (Menge and Sutherland, 1976; 1987) provides a useful starting point for interpreting intertidal patterns of distribution and abundance at Guana Island. Observed patterns of sessile and mobile taxa are consistent with intertidal communities where predation is important. However, rather than swamping all other factors, consumer pressure may be influenced by variation in environmental harshness, such as between wave exposures (Ortega, 1986). Competition, which is an important factor in Brazil (Sgrott Sauer Machado et al., 1992; 1996) and Costa Rica (Sutherland and Ortega, 1986), may play a role at exposed sites, where erect algae and *Balanus* barnacles overgrow each other. Recruitment intensity among sites may also play a role, as it did for *Chthamalus fissus* Darwin, 1854 in the Bay of Panama, Costa Rica, and the Gulf of California (Sutherland, 1990). The relative importance of physical factors, predation, competition, and recruitment for community organization may also change seasonally (Ortega, 1987), even within the same shore level (Williams, 1993; 1994; Williams et al., 2000). Experimental approaches are needed to assess the relative influence of physical and biological factors on patterns observed along Caribbean rocky shores and to gain a broader understanding of tropical intertidal communities in general.

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ADDRESSES: *Department of Ecology and Evolutionary Biology, Brown University, Providence, RI 02912.* PRESENT ADDRESS: *Conservation Biology, Northwest Fisheries Science Center, National Marine Fisheries Service, 2725 Montlake Boulevard East, Seattle, Washington 98112, E-mail: <tom.good@noaa.gov>.*

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Is There a Relationship between Proximity to Sewage Effluent and the Prevalence of Coral Disease?

LONGIN T. KACZMARSKY[1], MATTHEW DRAUD[1], AND ERNEST H. WILLIAMS[2]

[1] *Long Island University, C.W. Post, Department of Biology, Brookville, NY 11548-1300*
[2] *Department of Marine Sciences, University of Puerto Rico, P.O. Box 908, Lajas, Puerto Rico 00667-0908*
Corresponding author: longin.kaczmarsky@fiu.edu

Abstract.—We studied the prevalence of black-band disease (BBD) and white plague type II (WP) in two sites (Frederiksted and Butler Bay) within a St. Croix coral community that varied in relative exposure to sewage outflow. During sewage discharge events, fecal coliform and *Enterococci* data indicated impact area was limited to Frederiksted. We gathered data from seven belt transects in each of the two sites during the summer of 2001. We sampled 1046 colonies in 343 m^2 in Frederiksted and 2399 colonies in 302 m^2 in Butler Bay. There was significantly (Chi-square: df = 1, p = 0.0001) more disease in the impacted site with 13.6% of colonies of locally susceptible species infected (n = 566) versus the up current site that had 3.7% (n = 1344). Prevalence was highest for *Diploria clivosa*, with 23.7% of total colonies (n = 76) infected in the sewage impact site, which was significantly (Chi-square: df = 1, p = 0.0001) more than in the Butler Bay site where only 2.5% of the total colonies (n = 320) were infected. Recorded mortality, due to WP type II, was most severe for *Dichocoenia stokesi* with 26% of the infected colonies (n = 38) dying in just two months or 9.4% of the total *D. stokesi* sample population (n = 107). The results of the study suggest a relationship between a high prevalence of BBD and WP type II and exposure to sewage.

Keywords.—St. Croix, coral-disease, sewage, white-plague, black-band-disease

INTRODUCTION

Reports of coral reef disease have risen in the last two decades (Hayes and Goreau 1998; Richardson 1998; Harvell et al. 1999; Williams and Bunkley-Williams 2000; Porter et al. 2001). Disease events are recognized as important factors affecting coral community composition, structure, and dynamics (Weil et al. 2003). While the exact causes of coral diseases are largely unknown (Goreau et al. 1998; Richardson 1998), they are often assumed to be linked to either direct or indirect anthropogenic stresses (Hallock et al. 1993; Santavy and Peters 1997; Geiser et al. 1998; Williams and Bunkley-Williams 2000). Despite the contention that human-caused environmental perturbations are associated with higher levels of coral disease, there is minimal quantitative support for the hypothesis (Harvell et al. 1999; Williams and Bunkley-Williams 2000; Kuta and Richardson 2002) and links to specific disturbances are unclear (Bruckner 2002).

Sewage is one of the most significant pollutants affecting the coastal environments of the Wider Caribbean Region, with only about 10% of the sewage generated in the Central American and Caribbean island countries being properly treated (UNEP 1994). In the U.S. Virgin Islands alone, waste loads from domestic sources contain 440 tons per year of suspended solids, 250 tons per year of nitrogen and 132 tons of phosphates (UNEP 1994). Pollution, sewage, and/or elevated nutrient levels (the usual result of sewage discharges) have been linked by a number of studies to coral mortality (Smith et al. 1981; Jokiel 1986; Mate 1997).

While causal connections had not been established, two studies did make rigorous quantitative correlations between elevated nutrients and disease (Kim and Harvell

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2002; Kuta and Richardson 2002). In a more recent study, however, nutrient enrichment was shown to significantly increase the severity of coral disease progression in two Caribbean coral diseases, aspergillosis and yellow band disease (Bruno et al. 2003). Hatcher et al. (1989) predicted because of variations in background nutrient levels coral community responses to nutrient loading from sewage input would have no simple pattern.

A number of researchers have made the link between nutrient inputs from such sources as sewage and alterations in marine bacterial communities that result in environmental degradation and coral disease (Mitchell and Chet 1975; Colwell and Sayler 1978; Brown and Howard 1985; Dustan and Halas 1987; Hayes and Goreau 1998).

Sewage effluent may be a possible source of coral pathogens (Patterson et al. 2002). Pathogen host shifts or expansions of their host ranges that might include corals are distinct possibilities (Smith et al. 1996; Harvell et al. 1999; Weir et al. 2000). Despite a few studies that have attempted to examine the relationships between coral disease and water pollution (Mitchell and Chet 1975; Antonius 1981), the role of anthropogenic influence remains unclear (Green and Bruckner 2000). These relationships however, are beginning to be recognized as some of the most important yet poorly understood aspects of coral disease.

The first author (LK) had lived on St. Croix from 1986 to 2000 and in that time observed numerous outbreaks and mortality events affecting the local marine life (gorgonians, urchins, sea grass, and coral species in the genera *Diploria, Porites,* and *Acropora*), particularly around the Frederiksted sewage outfall. During many of those events visual swim surveys of other nearby sites revealed substantially less or no impact. One of the most striking mortality events in the Frederiksted site occurred in the fall of 1999 when a large portion of the dominant gorgonian, the Rough sea plume, *Pseudopterogorgia acerosa*, was killed off by an *Aspergillosis* outbreak (laboratory confirmed; unpublished data). This was the first time this species was observed infected by this disease. These events were the motivation for a more scientific investigation of the site.

Thus, our primary goal was to examine the relationship between the prevalence of coral disease and the relative exposure level to sewage effluent in two ecologically similar coral reef sites on the west shore of St. Croix. Using a matched pair comparative approach, we tested the simple prediction that disease prevalence is higher in a site that is adjacent to a frequently used sewage bypass outfall (Frederiksted) than in a site within the same coral community that is remote and upstream from the same sewage outfall (Butler Bay).

Methods and Materials

Study Sites

This study examined coral disease frequency in two geomorphologically and ecologically similar sites within a continuous hard pavement fringing coral community, which extends for approximately 10 km along the west coast of St. Croix in the U.S. Virgin Islands (Fig. 1). Site selection minimized natural variability, while highlighting the difference in sewage exposure. Transects were permanently marked and locations recorded using GPS for future community and population studies.

The west coast of St. Croix is in the lee of the island and is generally less exposed to oceanic swells and the high-energy wind-generated waves found on the east end. The pavement of the narrow shelf (<150 m) has a gentle to moderate slope with a sparsely to moderately developed fringing coral community. Geomorphologically, the western coast is fairly similar along most of its length. A typically weak to sometimes-moderate current running parallel with the shoreline regularly reverses but usually runs north to south. Daily tidal range is small, typically 0.2 m (Kjerfve 1981). Biological connectivity between the sites appears high thus ecological differences between the sites would not likely be due to spatially broad factors such as water temperature, hurricanes, overfishing, or major recruitment events but rather would be

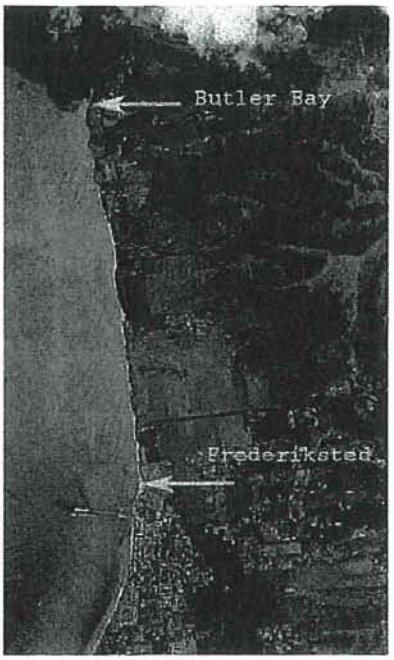

FIG. 1. Map and photo of study sites. The map of St. Croix (top) and the aerial photo (left) show the location of the two study sites (approx. 2.5 km apart). The town of Frederiksted can be seen at the bottom of the photograph.

more consistent with small-scale and short-term local factors.

The Frederiksted site, at 17°74.0′N, 64°87.5′W, is close to the town of Frederiksted (>10,000 people). The sewage system normally carries all city sewage to a central island treatment facility. An increasing trend in sewer system failures over the past 15 years has caused, at times, approximately half the islands untreated sewage to be diverted from the treatment facility and funneled to Frederiksted's sewage bypass outfall (A. Hutchins, V.I. Dept of Planning and Natural Resources; J. Bradford, St. Croix Dept of Public Utilities; pers. comm. 2001). Many cases of sewage bypasses in St. Croix are not reported (pers. obs., 1990-2002; J. Casey, US EPA Regional Director, pers. comm. 2001). EPA records show that between March 19, 1997 and June 27, 1999 (the only USEPA records available for the period closest to the time of the study), 90 sewage bypass events were reported for all of St. Croix, which totaled 15.3 million gallons of untreated sewage discharged at several different sites along the shorelines. Between 1997 and 2000, the only volumes reported by the Virgin Islands Department of Planning and Natural Resources (VIDPNR) at the Frederiksted study site were for two events: October 1, 1997 totaling 434 375 liters, and 3 270 586 liters on September 14, 2000. Although five other bypass events were recorded during the first half of 2000 in Frederiksted, no volumes were documented. These figures are presented here as an indication of the severity of some sewage system failures. The sewage-influenced study site (Frederiksted) begins about 7 m west of the sewage outfall where bypass events, as frequent as three times a month, are observed. Visible plumes associated with these events are only noticeable when it rains and are relatively small (<100 m wide). Additionally, an indeterminate volume of sewer line leakage from corroded pipelines flows continuously into the outfall drainage (pers. obs.; Hutchins, pers. comm.).

The Butler Bay study site is near Estate Butler Bay, which is a very lightly populated area, and is located 2.5 km north (up current) of the sewage outfall in Frederiksted at 17°75.3′N, 64°88.8′W. While it is ecologically similar it is shielded from regular exposure to sewage discharge, as indicated by the water quality tests that follow.

As an indication of the limited spatial impact of these sewage bypass events, which shows a pattern restricting it to the Frederiksted study site, we used recent laboratory analyses on water samples taken from sites along the west coast that were supplied by the VIDPNR (unpublished data, V. Vilanueva-Mayor). They reported on November 18, 2003, during a sewage bypass event, that seawater fecal coliform levels in Frederiksted were 1460/100 mL and *Enterococci* levels at 1880/100 mL while at the same time just 0.5 km north samples were 1/100 mL for fecal coliforms and 0/100 mL for *Enterococci* and at Butler Bay fecal coliforms and *Enterococci* were 0/100 mL and 5/100 mL, respectively. During an event on December 17, 2002 at Frederiksted, fecal coliforms and *Enterococci* were 3760/100 mL

and 7560/100 mL, respectively, while at the same time 1.5 km north fecal coliforms were 4/100 mL and 0/100 ml for *Enterococci*, and just 0.5 km to the south of Frederiksted, fecal coliforms were 2/100 mL and 3/100 mlL for *Enterococci*. The combination of the narrow shelf, wave action and the straight, open coastline appears to limit the spread of the effluent along the coast.

Survey and Analytical Techniques

Seven 2 meter wide belt transects approximately 10 meters apart were used in each site with depth ranging from 1 to 4 meters. The beginning of the first transect was determined haphazardly by swimming out to the beginning of the coral communities and blindly dropping the rolled up transect measuring tape from the surface and beginning where it landed. Following a north compass heading, roughly parallel with the shoreline, each subsequent transect began approximately 10 meters away. Transects lengths varied slightly because permanent transect end markers could not always be adequately secured at the 20 m mark (our first choice) because of the presence of sand or loose rubble but were approximately 20 m each. Generally, we extended the transect more than 20 m rather than shortening it. This is what accounted for the difference between sites in total reef area examined (see below). Each transect was divided into numbered 2 m^2 plots to help relocate individual colonies for long-term monitoring. Positioning of the survey areas deliberately avoided large areas of sand and seagrass.

Each coral colony including gorgonians and milleporid were identified to species using Veron (2000) and the maximum height and width were recorded to the nearest cm on all colonies 3 cm wide or larger. Each colony (infected and uninfected) was assigned a logbook number with an exact location. In total, 302 m^2 were examined in Butler Bay and 343 m^2 in Frederiksted. Data were recorded using snorkel and SCUBA surveys and conducted over a two three week period from the last week of June to the second week of July and in August 2001. The diseases and syndromes were identified using Richardson (1998), NOAA (1999), and Williams and Bunkley-Williams (2000). For each infected colony numbered, aluminum tags were attached nearby to the substratum with masonry nails and photographs were taken. Two small nails were also inserted a few centimeters behind the area of active disease progression (or necrotic zone) in a line perpendicular to the area infected. Using a flexible ruler, with its edge flush with the two nails, the distance from the first nail to healthy tissue was recorded. After at least a month and as sea conditions permitted, tagged colonies were visited a second time, and again photographed and measured to estimate the rate of disease progression on a per day basis. All diseased corals were photographed with a Sea and Sea Motormarine 35 mm camera and lens; model MX-10 with 32 mm lens and close-up attachment. To verify and characterize the putative pathogens, samples of black-band disease and white-plague-infected tissues were collected and examined microscopically and cultured by the authors and at the laboratory of Dr. L. Richardson, an expert in pathogens of Caribbean corals.

Disease prevalence data were divided into the following categories : 1) All species observed with BBD, 2) All species observed with WP type II, 3) Elliptical star coral, *Dichocoenia stokesi* with WP type II, 4) Knobby brain coral, *Diploria clivosa* with BBD, 5) *D. clivosa*, with WP type II, 6) Symmetrical brain coral, *Diploria strigosa*, with BBD, 7) *D. strigosa*, with WP type II, 8) Massive starlet coral, *Siderastrea siderea*, with WP type II, 9) Great star coral, *Montastrea cavernosa*, with BBD, and 10) Blushing star coral, *Stephanocoenia michelinii*, with WP type II.

We reviewed the results of the only available water quality tests closest to the time of our study, which were conducted by the EPA (March 2000) and DPNR (Department of Planning and Natural Resources of the Virgin Islands; June 2000 to June 2001) on the Frederiksted site and from a sampling site located one km south of our Butler Bay site. Water temperature was recorded daily between 12 pm and 5pm during the study using a laboratory thermometer made and

calibrated by H-B Instruments Co. and tested against thermometer standards traceable N. I. S. T. with an accuracy of ±1 degree. Because of the openness of St. Croix's west coast, its lack of tidal flats, and small tidal range (<0.2 m) the temperature changes very slowly thus the time range here makes little difference. Water samples were collected from the two sites almost simultaneously and tested for total suspended solids and turbidity on June 21 2001 and July 3 2001. Benthic sediment samples were analyzed for grain size composition from nine locations in the Frederiksted site and four from the Butler Bay site during the June/July survey period. Rainfall records for the survey periods were supplied by a private field station on Hermon Hill, St. Croix operated by Dr. Ken Haines. Rainfall data for the survey period was compared to eight year averages in order to assess potential influence of any major run-off events occurring during the survey.

Statistical Methods

Chi-square contingency tables (Yates corrected) (Zar 1996; Fowler et al. 1998) were used to determine if the prevalence of diseased colonies significantly differed between sites and size-classes. Spearman's rank correlation analysis (Fowler et al. 1998) was used to determine if a significant relationship existed between colony density and prevalence of disease. Differences in densities of each species were analyzed using the Mann-Whitney U-test. Scleractinian coral diversity was determined by the Shannon-Weiner diversity index (H'n) and evenness was determined using Pileou's evenness component (Magurran 1988). Rarefaction analysis was used to compare the species richness of the two sites (Krebs 1999). Rarefaction standardizes samples from different sites to a common sample size and estimates the number of species expected in a random sample of individuals taken from a site.

Results

Three types of disease/syndrome were observed: 1) black-band disease (BBD), 2) a form of white plague type II (WP type II), and 3) dark spot syndrome (DS). There was a high prevalence of DS observed on colonies of *Siderastrea siderea*, and Lesser starlet coral, *S. radians*, in both sites. Other coral diseases/syndromes observed less frequently: aspergillosis on a few gorgonians, sponge overgrowth on a few corals, and a few cases of blotchy partial bleaching were recorded. These were excluded as too few for statistical analysis. No white-band or yellow-band disease were observed in the study sites but were observed elsewhere around St. Croix during the study period.

Between Site Differences

A higher prevalence of disease occurred in the Frederiksted site, with 7 of 10 species or categories examined had significantly more disease (Table 1, Fig. 2). Of the 21 species of scleractinian coral recorded and examined in the transects of both sites, 8 had BBD and/or WP type II (Table 2). Of 1046 colonies examined in Frederiksted, 566 colonies were of species susceptible to disease, 77 (13.6%) of which were infected. Of the 2399 colonies examined in Butler Bay, 1344 colonies were of species susceptible to disease of which 50 (3.7%) were infected.

Frederiksted had significantly more coral disease than Butler Bay for 1) BBD 2.7% (14 of 521) versus 1.0% (13 of 1255) (Chi-square: $df = 1$, $p < 0.013$); 2) WP type II 11.4% (63 of 552) versus 3.1% (38 of 1223) (Chi-square: $df = 1$, $p < 0.0001$); and 3) BBD and WP type II combined 13.6% (77 of 566) versus 3.7% (50 of 1344) (Chi-square: $df = 1$, $p < 0.0001$).

Coral diversity was lower in Frederiksted (H'n = 1.99) compared to Butler Bay (H'n = 2.28). Eveness was similar: 0.75 in Frederiksted and 0.73 in Butler Bay. Rarefaction analysis estimated species richness (including milleporids) was higher in Butler Bay, with 21 species, than Frederiksted, with 14 species. Using the Mann-Whitney U-test colony densities of large massive species were significantly lower in Frederiksted for *M. cavernosa* (93% fewer colonies;

TABLE 1. A comparison of White Plague type II (WP) and Black Band Disease (BBD) prevalence between Butler Bay and Frederiksted study sites. The "All species" groupings include only the species found to be susceptible to the disease(s) in this study. Asterisks denote significant differences. *D. labyrinthiformis* was only observed with disease outside the transects as was the case with BBD-infected *S. siderea* (only a few colonies), which is why there is limited data for these species.

Species or grouping	Butler Bay	Total colonies	Frederiksted	Total colonies	p value
All species, WP and BBD	3.7%	n = 1344	13.6%	n = 566	0.0001*
All species, BBD only	1.0%	n = 1255	2.7%	n = 521	0.013*
All species, WP only	3.1%	n = 1223	11.4%	n = 552	0.0001*
D. stokesi WP only	41.4%	n = 70	59.5%	n = 37	0.15
D. clivosa, WP and BBD	2.5%	n = 320	23.7%	n = 76	0.0001*
D. clivosa BBD	2.5%	n = 320	14.5%	n = 76	0.0001*
D. clivosa WP	0%	n = 320	9.2%	n = 76	0.0001*
D. strigosa WP and BBD	1.7%	n = 363	6.8%	n = 279	0.002*
D. strigosa BBD	1.4%	n = 363	0.7%	n = 279	0.93
D. strigosa WP	0.3%	n = 363	6.1%	n = 279	0.0001*
S. siderea WP	1.8%	n = 443	7.7%	n = 143	0.002*
M. cavernosa BBD	0%	n = 121	7.1%	n = 14	0.19
S. michelinii WP	0%	n = 18	57.1%	n = 7	0.004*

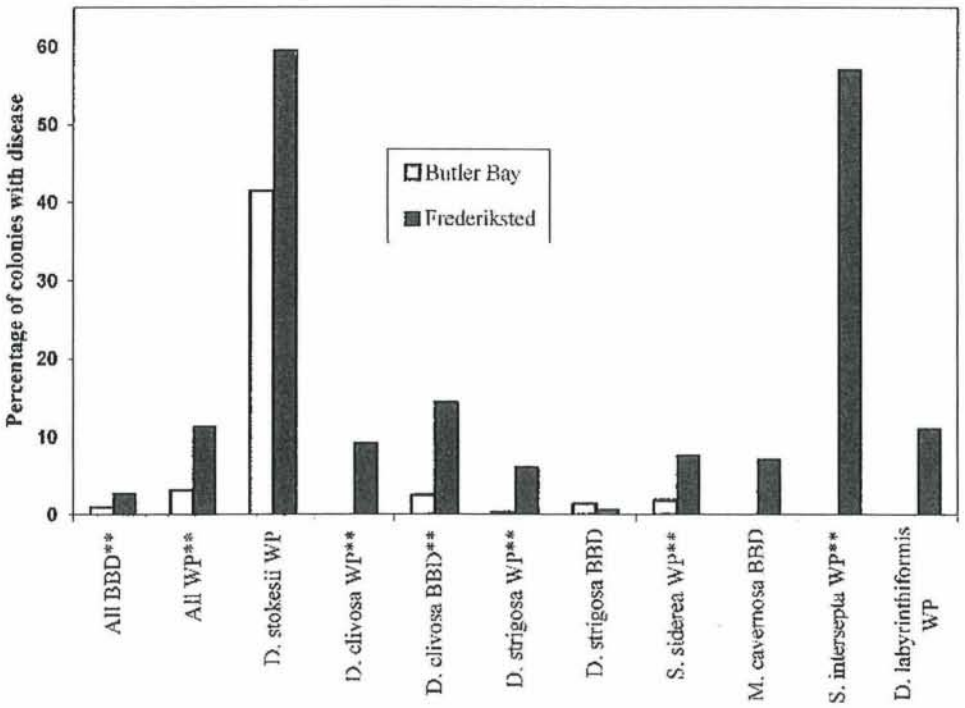

FIG. 2. Percentage of diseased coral colonies of common species. Only species that were susceptible to white plague type II (WP type II) and/or black band disease (BBD) in these study sites were included. ** indicates a statistically significant difference between sites using the chi-square test, $p < .05$.

p = 0.0209, n = 8), *S. siderea* (80% fewer colonies; p = 0.0209, n = 8), *S. michelinii* (77% fewer colonies; p = .0209, n = 8), and *D. stokesi* (67% fewer colonies; p = 0.0209, n = 8). The colony density of *Siderastrea radians*, a small massive brooder, was significantly higher in Frederiksted (394% more colonies; p = 0.0127, n = 14).

TABLE 2. Coral species found susceptible to disease in the St. Croix study sites, Frederiksted and Butler Bay. WP = white plague type II, BBD = black band disease, DS = dark spot syndrome.

Species	Diseases
Diploria clivosa	WP/BBD
Diploria strigosa	WP/BBD
Diploria labyrinthiformis	WP/BBD
Dichocoenia stokesi	WP
Siderastrea siderea	WP/BBD/DS
Siderastrea radians	DS
Stephanocoenia michelinii	WP
Montastrea cavernosa	BBD
Dendrogyra cylindrus	WP

WHITE PLAGUE TYPE II

Seven species of coral (*Dichocoenia stokesi, Diploria strigosa, D. clivosa, Diploria labyrinthiformis, Stephanocoenia michelinii, Siderastrea siderea,* and *Dendrogyra cylindrus*) had WP type II (Table 2, Fig. 3). The mean rate of disease progression for all tagged colonies infected with WP type II (n = 100) was 0.76 mm per day (S.D. = 1.26 mm: range 0.10-8.6 mm per day).

Prevalence of WP type II was also significantly higher in Frederiksted for the following infected species: *D. clivosa* 9.2% versus 0% ($p < 0.0001$); *D. strigosa* 6.1% versus 0.3 % ($p < 0.0001$); *S. siderea* 7.7% versus 1.8% ($p < 0.0015$); and *S. michelinii* 57.1% versus 0% ($p < 0.0038$). However, there was no significant difference in disease prevalence between sites for the most severely WP type II-infected species, *D. stokesi*, with 59.5% in Frederiksted and 41.4% in Butler Bay (Table 1, Fig. 2).

The prevalence of WP type II was significantly higher in small (<10 cm) colonies of D. *strigosa* than in larger colonies (chi-square: df = 1, $p < 0.0261$). All other species with disease were examined for size-related effects and results were inconclusive.

BLACK-BAND DISEASE

In the study sites, black band disease infected five species: *D. strigosa, D. clivosa, Montastrea cavernosa, S. siderea,* and *D. labyrinthiformis.* These were infected at least once (Table 2). The mean rate of disease progression for all tagged colonies infected with BBD (n = 27) was 1.45 mm per day (S.D. = 1.10; range 0.32-5.8 mm per day). Positive identification of the pathogens responsible for BBD was microscopically confirmed from the collected samples (L. Richardson, pers. comm.).

The prevalence of BBD was significantly higher in Frederiksted for the most severely BBD-infected species, D. *clivosa* 14.5% (11 of 76 colonies) versus 2.5% (8 of 320 colonies) (Chi square: df = 1, $p < 0.0001$). In Frederiksted the impact of combined colonies of D. clivosa with both type of infections (BBD and WP type II 23.7% of colonies diseased), in ecological terms, is severe and significantly higher than Butler Bay (2.5%) (Chi-square: df = 1, $p < 0.0001$). No WP type II was observed on *D. clivosa* in Butler Bay. There was no significant difference between sites in the prevalence of BBD for either *D. strigosa* or *M. cavernosa* (Table 1, Fig. 2).

MORTALITY RATE

During the survey periods high mortality occurred among the populations of *D. stokesi* infected with WP type II. During the second survey period in August, 10 of the 38 (26%) WP type II-infected colonies of *D. stokesi* tagged in the first survey had died. This translates to 9.4% of the entire sample population of *D. stokesi,* (n = 107) died in approximately 2 months.

DISEASE CLUMPING AND COLONY DENSITY

Spearman rank correlation analyses revealed no significant correlations between the prevalence of WP type II or BBD and colony density in either site. For example, for all species susceptible to WP type II in Frederiksted, Rs = 0.18 and p = 0.70, n = 7. Colony density for all species susceptible to WP type II in Frederiksted versus percent affected by WP disease was as follows: transect 1, 0.10 colonies per m^2 vs. 0.0%; transect 2, 0.31 colonies per m^2 vs. 16.6%; transect 3, 1.34 colonies per m^2 vs. 12.1%; transect 4, 2.22 colonies per m^2 vs. 21.0%;

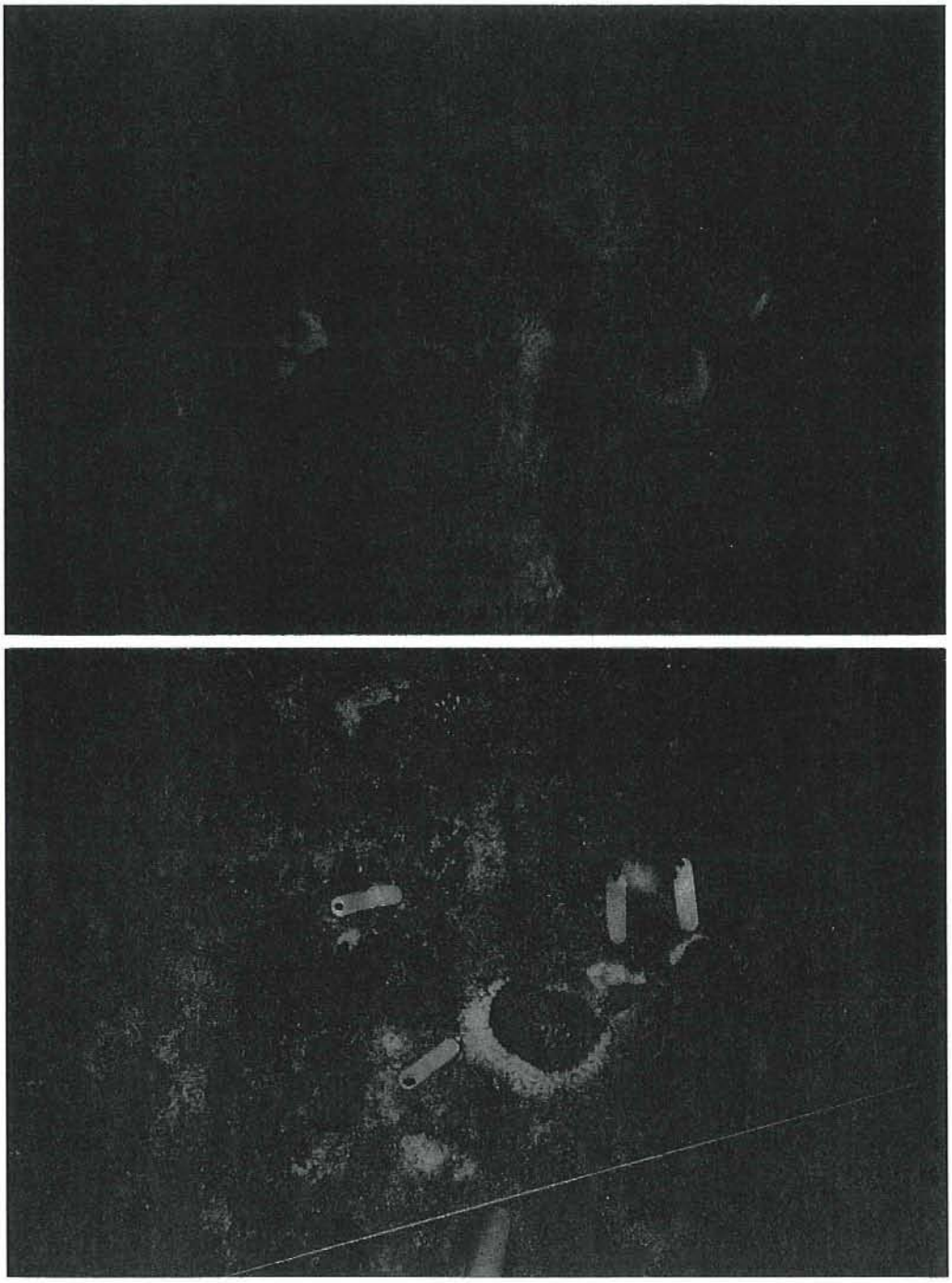

FIG. 3. Photographs showing patterns of contagiousness as observed in the Frederiksted study site. Viewed from above. Top) A cluster of seven colonies all with white plague type II, five *D. strigosa* and two smaller *S. siderae*. Bottom) Four WP type II-infected colonies in close proximity; *D. stokesi*, *D. strigosa*, and two *S. siderea* colonies (from left to right).

transect 5, 2.69 colonies per m^2 vs. 8.8%; site 6, 2.19 colonies per m^2 vs. 8.1%; site 7, 1.22 colonies per m^2 vs. 12.8%.

Environmental Data

Tested environmental parameters were as follows: turbidity averaged 0.31 NTU's (n = 2) in Frederiksted vs. 0.31 NTU's (n = 2) in Butler Bay, total suspended solids averaged 9.3 mg/L (n = 2) in Frederiksted vs. 15.8 mg/l in Butler Bay (n = 2) and the clay/silt fraction of the sediment (<0.0625 mm) averaged 5% (n = 9) in Frederiksted vs. 6% (n = 4). VIDPNR turbidity data averaged 0.78 NTU's (n = 7) in Frederiksted and 0.89 NTU's (n = 4) in Butler Bay (June 2000 to June 2001). The rainfall, June through August 2001,was normal (8 previous years average 5 to 9 cm per month). However, for May 2001 just prior to the survey, rainfall was more than three times higher than normal (27 cm vs. 8 cm). Water temperature averaged 29°C during the June/July period and 30°C in August.

Discussion

Our results indicate significantly higher prevalence of BBD and WP type II in a site exposed to persistent sewage discharge (Frederiksted) than in an ecologically similar up current site (Butler Bay). Environmental data during episodes of sewage bypasses show, by fecal coliform and *Enterococci* levels, that the impact appears to diminish after only a few hundred meters. Thus the Butler Bay site is clearly less exposed to effluent. The disease data was stronger for BBD than WP type II. Considering all the species that were susceptible to both diseases in this study there was a significant difference in overall disease prevalence, with nearly 14% of 566 susceptible colonies diseased in the Frederiksted site as compared with less than 4% of 1344 in Butler Bay. In fact all but one of the 8 species (i.e. *D. stokesi*) of coral susceptible to BBD and/or WP type II had statistically significant higher disease prevalence in the Frederiksted site. As non-parametric analysis was used and therefore Type II error is more likely, there's the possibility that WP II-infected *D. stokesi* colonies were in fact significantly more prevalent in Frederiksted as well. Acquiring normally distributed data from percentages (even after transformation) is not typical and applying parametric tests to data such as this one might not be appropriate. Using nonparametrics in our study was in fact more conservative than using parametrics in terms of finding the significant differences.

Diploria clivosa forms massive colonies in the shallow waters along the west end of St. Croix and is the most dominant coral species contributing to shallow-water community structure, possibly having replaced the once dominant *A. palmata*. Our results are particularly striking when we draw a comparison between disease prevalence of *D. clivosa* in the two sites. In Frederiksted, *D. clivosa* was more likely to be diseased than in the Butler Bay site. The prevalence of BBD on this species in the Frederiksted site (14.5%) is one of the highest levels reported for any single species to date. Other large massive coral species, *Siderastrea siderea*, *Stephanocoenia michelinii*, and *Diploria strigosa*, were also significantly more affected by one or both of the diseases in Frederiksted. We speculate that sustained levels of disease similar to those reported in this paper may lead to (and perhaps already have caused) considerable restructuring of the shallow coral community in the Frederiksted site. An increase in nutrients may play a role in increased virulence and fitness of coral pathogens (Bruno et al. 2003). We suspect that past sewage impacts have resulted in reductions in coral species richness, diversity and densities.

Because we are concerned with the potentially negative impact of sewage and dark spot syndrome is generally thought not to cause mortality (Bruckner 2002), the data collected for DS were not analyzed here. The results of this study would have been essentially the same as we obtained using just BBD and WP type II data alone even if all syndromes and diseases had been pooled because there were so few incidents of diseases/syndromes other than WP type II and BBD.

White Plague type II

The WP type II seen here formed a distinct white line or bleached band between

healthy tissue and exposed skeleton. The typical infection appears to start from the base. The combined disease prevalence of WP type II (5.7%; n = 1775) was higher in this study than in all other Caribbean studies except Nugues (2002), who found 11% of the susceptible colonies infected during an outbreak in St. Lucia in March 1998. Based on WP progression rate (a distinguishing characteristic of the WP disease types) in our two sites, which ranged between 0.1 mm to 8.6 mm per day, it is not clear whether the WP seen here is more similar to WP type I (3.0 mm/day, Dustan 1977) or type II (up to 2 cm/day, Richardson 1998). Alternatively, specific colonies, species, and/or microhabitats may vary in the expression of disease virulence (i.e. disease progression rates).

Ten tagged WP type II-diseased colonies were dead during the second survey period. The exact disease progression rates for these particular colonies could not be determined and could have been much faster for these colonies. Recorded per day rates for these colonies that died sometime between the two survey periods were thus, minimum values, which ranged from 0.18-1.23 mm per day. However, these minimum values were consistent with the other observed rates (0.10-8.6 mm per day) and thus were included in the calculated average rate.

It should be noted that in calculating the prevalence for the category "all species infected with WP", colonies of *D. labyrinthiformis* found in the transects were included for the following reason. While only healthy colonies of *D. labyrinthiformis* were found within the transects, infected colonies were found outside the transects, nearby. Since this makes it a locally susceptible species and it was found uninfected within the transects, the number of healthy colonies in the transects was added to the total number of susceptible colonies found in the transects (healthy and diseased) used in calculating the prevalence in the category "all species infected with WP". No colonies from outside the transect were included in the counts.

A sample examined microscopically had revealed numerous gram-negative rods and flexi-rods. The gross morphology of the rods appeared to be the same as the pathogen identified in the white plague type II outbreak in Florida coral reefs (Richardson 2001, pers. comm.). Plate culturing produced many uniform round whitish translucent colonies 1.5 to 3 mm in diameter after 4-day incubation at room temperature on nutrient agar. The colonies darkened slightly to yellow and became more opaque after one more week of incubation. Recently, Denner et al. (2003) identified and described the causative agent of white plague type II with similar gross morphology and characteristics and, based on detailed polyphasic taxonomic characterization, proposed a new genus and species name, *Aurantimonas coralicida.* In addition, *D. stokesi*, which was severely diseased in this study, appears to be susceptible to type II, and not to type I, disease (Richardson 1998).

The significantly higher prevalence of WP type II in small (<10 cm) colonies of *D. strigosa* might be explained by their greater contact with sediment-associated pathogens. However this was not tested; other factors that differ between small and large colonies could be involved. Since these smaller colonies would perish more quickly than larger colonies and be lost from the count, the apparent preference for small colonies was probably greater than our data suggested. All other species with disease were examined for size-related effects and results were inconclusive. While large colonies (>10 cm in height) for most species and disease type had significantly higher disease prevalence, this is not necessarily indicative of higher susceptibility, as they may simply take longer to die or have more time to recover than smaller colonies. However if small colonies have significantly higher disease rates then this variable is not relevant. This was the case only for *D. strigosa*.

Black-Band Disease

Seven species of Caribbean coral were reported as the most susceptible to BBD: *Colpophyllia natans, D. strigosa, D. labyrinthifor-*

mis, Montastrea cavernosa, M. annularis, M. faveolata, and *M. franksii* (Rutzler et al. 1983; Edmunds 1991). Bruckner et al. (1997) and Kuta and Richardson (2002) recently included *D. clivosa* in the list of most susceptible. Our study supports the addition of *D. clivosa* to that list.

For *S. siderea,* and *D. labyrinthiformis,* infected colonies were only found outside transects, nearby, thus this makes these species locally susceptible. These were also species found uninfected within transects, so the number of healthy colonies found in the transects were included in the total number of colonies (healthy and diseased) used to calculate disease prevalence for the category "All species infected with BBD."

The mean rate of disease progression for all colonies infected with BBD in St. Croix was 1.45 mm per day (S.D. = 1.10), which is about half the reported average in other studies (Rutzler et al. 1983; Kuta and Richardson 1997). However the rates are typically highly variable within and between studies and can occasionally reach 2 cm/day.

Contagiousness

Antonius (1985) found BBD to be infectious *ex situ*. However, contagiousness of BBD and WP type II in the field is still in question (Edmunds 1991; Kuta and Richardson 1996; Bruckner and Bruckner 1997; Richardson et al. 1998a; Nugues 2002). Our correlation analysis found no statistical support of either disease being contagious. However, our qualitative observations lead us to think they might be. For example, several adjacent *D. clivosa* colonies were observed sharing circular patches of BBD infection indicating possible contagiousness. However this may also be explained by the two colonies sharing a common initial injury that led to disease. Although our data doesn't substantiate it, colonies infected with WP type II in St. Croix often visually appeared clumped (Fig. 3). Nugues (2002) also found no statistical evidence of clumping in the WP type II outbreak in St. Lucia. While clumped distributions would suggest contagiousness via the spreading of pathogen among adjacent colonies, alternately clumped distributions could result from adjacent colonies sharing an environment highly favorable to the pathogen.

Populations, Community Structure and Disease

We found that coral species richness, diversity and densities were lower in the Frederiksted site, which had significantly higher BBD and WP type II prevalence. We speculate that chronic exposure to sewage and the resulting high incidence of disease has contributed to this reduction. Kuta and Richardson (2002) also found coral diversity to be lower in sites with BBD-infected corals compared to sites without BBD. In St. Lucia, Nugues (2002) suggested WP type II could progressively deplete two of the most important reef frame-building coral species. Higher colony densities in our sites were not associated with higher disease prevalence as one might expect with density-dependent factors.

Implications

From our study we speculate that BBD and WP type II may be killing a high proportion of ecologically important scleractinian corals (i.e. *D. clivosa* and *D. stokesi*) in some St. Croix near-shore coral communities and may be responsible for restructuring these communities. For example, the prevalence of WP type II among the *D. stokesi* population here is among the highest thus far reported in the Caribbean for any species. Because our sample area is small, however, this localized high level of infection would not necessarily parallel the larger scale trends showing lower prevalence seen in other studies (e.g. Weil et al. 2003). More than 25% of the infected *D. stokesi* colonies died in less than three months in our study. We speculate that this translates into an estimated mortality rate of almost 10% (both sites combined) for this species within at least the area between sites, which extends across approximately 25% of the fringing west coast coral community. *Dichocoenia stokesi* is the fourth

most abundant large massive coral in this community. This level of mortality due to WP type II could cause considerable restructuring. Despite a severe outbreak of white plague type II in the Florida Keys in 1995 that highly affected *D. stokesi*, Richardson and Aronson (2003) initially concluded that it had minimal long-term impact as this species began recolonizing these reefs within a year. However, since then the population of these new recruits has plummeted (Richardson, pers. comm.). Ongoing monitoring of the size-class frequencies in our study sites in St. Croix will reveal if any long-term impact occurs as a result of these disease events.

According to Kuta and Richardson (1997), BBD acts selectively on important reef framework species and stated that this may result in reef degradation and changes to the reef community structure. On a local scale, BBD appears to cause considerable restructuring in St.Croix by selectively removing *D. clivosa* from the sewage-impacted site. *Diploria clivosa* nears 100% cover in some areas of the up current Butler Bay site and although it is at a lower density in the shallow range of the Frederiksted site (1-1.5 m) it is still dominant there. There are several very large *M. cavernosa, S. michelinii* and *D. cylindrus* colonies in the deeper range (2-4 m) of both study sites. All three of these were susceptible to either WP type II or BBD. We speculate that the increasing trend in sewage discharges over the past 15 years (J. Casey, J. Bradford, A. Hutchins; pers. comm.) has contributed to a decline in these large massive species in Frederiksted. *M. cavernosa* and *S. michelinii* were much more abundant 5 to 15 years ago in Frederiksted (pers. obs.), to which some large dead colonies of these species now attest. While the exact cause of death is unknown, the death of these large colonies coincided with the period of increasing frequency of sewage bypasses. Also more abundant 10 to 15 years ago, in both Frederiksted and Butler Bay, were *Acropora palmata* (now completely absent in the Frederiksted site) and *D. stokesi* (large tracts of dead colonies provide evidence). We suspect the significantly higher density of *S. radians* in the sewage-impacted site reveals it is opportunistically making use of space recently made available as a result of coral disease, at least in part.

Sewage and Other Potentially Synergistic Influences

Our study documents significantly higher disease prevalence at the Frederiksted site, which is directly and chronically exposed to raw sewage discharge, than the similar nearby Butler Bay site that is not directly exposed to sewage. The March 2000 U. S. EPA water quality report on St. Croix documented the presence of high nutrient levels and toxic contaminants in the water and sediments in this sewage-impacted site, an indication of chronic sewage exposure. We propose our study links higher disease prevalence with exposure to sewage and hypothesize that chronic sewage exposure can lead to higher rates of coral disease, especially considering the consistently statistically lower prevalence of disease in the ecologically similar upstream study site that is not exposed to sewage discharge. However, cause and effect has not been conclusively shown. We do realize that the two study sites are not identical in every way except sewage discharge and thus recognize several other potential sources of stress, which may contribute to increased coral disease in Frederiksted. For example, foot traffic from recreational (i.e. snorkelers and bathers) and fishing activities (i.e. seine netters and spear fishers) is higher in Frederiksted and likely results in more physical damage to corals there. It has been shown that the BBD infection rate is higher among damaged corals (Antonius 1985). In addition, recent landscape modifications adjacent to the Frederiksted study site of both marine (the recent installation of a vertical sea wall without any energy-absorbing toe protection, i.e. boulders) and terrestrial features (the recent addition of a large paved lot and partial filling of an adjacent drainage canal) have likely increased total suspended solids (TSS) and terrestrial run-off in the waters around Frederiksted. While a link between TSS and disease may exist, we do not

believe it accounts for the significant differences in disease measured in this study since our water samples indicated that TSS might possibly be higher in Butler Bay (15.8 mg/l, n = 2) than in Frederiksted (9.3 mg/l, n = 2). There is severe shoreline erosion at the Butler Bay site that might account for the higher TSS levels there, as opposed to anthropogenic pollution. Our measures of the silt/clay fraction of the surrounding sediments and turbidity (as well as turbidity data from VIDPNR) were similar in both sites. However, environmental sampling was limited and a more thorough sampling would be required to draw more robust conclusions about the role of these parameters. On St. Croix's north shore near the town of Christiansted a similar chronic sewage system failure problem exists resulting in large volumes of raw sewage that are bypassed into coastal waters. It is quite possible that coral disease there is also found in higher prevalence than adjacent reefs.

Not all coral diseases are exacerbated by environmental impacts (Goreau et al. 2000). Mona Island in the Mona Passage off Puerto Rico represents a more pristine environment than the waters near the main island of Puerto Rico. However, the corals around Mona Island have in the last few years suffered more from the yellow-band syndrome and *Cliona* (a coral-killing sponge) damage than the main island (Williams and Bunkley-Williams 2000; Williams, unpubl. data).

Our study presents some evidence that supports the idea that some coral diseases with wide ranges can have more intense impacts at local scales in association with anthropogenic influence. Furthermore, while BBD and WP type II have been observed to occur in remote areas not influenced by exposure to sewage and/or high nutrient levels, the results presented here suggest that their prevalence may significantly increase in localized areas as a result of exposure to such insults.

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A Comparison of Fish Densities Between Guana Island, British Virgin Islands and Virgin Islands National Park, U.S. Virgin Islands

Ralf H. Boulon, Jr., MS
Thomas Kelley, MA

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Introduction

From July 4 to July 10, 2004, we surveyed eleven coral reef sites along the western side and southern end of Guana Island, BVI, to quantify reef fish populations at those sites (Fig. 1). The goal was to compare fish species occurrences and densities to similar sites in Virgin Islands National Park (VINP) on St. John.

Coral reefs and associated reef fish populations are declining throughout the world at an alarming rate, much of it due to historical overfishing and other anthropogenic impacts (Jackson, et al. 2001). This has been well documented in Jamaica and is being documented throughout the Caribbean (Hughes 1994). For various reasons more fully described in other publications and documents, but including substantial marine-based tourism, significant terrestrial development and associated sedimentation, and harvest of reef fish by both commercial and recreational fishermen, the marine resources of VINP have experienced serious and increasing stressors, which have accelerated over the past 15 years (Rogers and Beets 2001; Beets and Rogers 2003; Garrison et at 1998; Garrison, et al 2004). Guana Island and the associated near shore reefs have experienced similar but substantially reduced stress. Fishing pressures on the Puerto Rico Bank are thought to represent a gradient from highest (western PR) to lowest (eastern BVI). Guana Island, being near the eastern extent of the Puerto Rico Bank, should lie near the end of the gradient with the least fishing pressure and therefore the better remnant fish populations.

VINP was awarded International Biosphere Reserve status by UNESCO in 1976, among the first in the world and as a representative example of Lesser Antillean Ecosystems. Reef fish census indices, rugosity and benthic habitat characterization and analysis have been conducted in VINP for approximately two decades. A quantitative and qualitative comparison of Guana Island coral reef fishes with similar and existing VINP datasets may show important differences which can provide management justifications for resource managers.

Methods

The Bohnsack/Bannerot stationary sampling fish census methodology (Bohnsack and Bannerot 1986) was employed without significant modification, for investigator consistency and maximum comparative value with existing, similar long-term VINP

datasets. A number of sites were censused within each selected reef. Sites were selected to include relatively continuous reef within the entire 15M diameter cylinder. Adjacent sites were located along the same approximate depth contour. Sufficient distance was established between the census points to avoid overlap or disturbance from the previous census. GPS (Garmin 72) data was recorded near each initial starting point from the anchored boat (Table 1). Following fish population data collection, reef characteristics and related information are gathered. Two divers performed two censuses each at each site.

Table 1. Guana Island Fish Census Sites:

Site #	Location	Latitude	Longitude
1	South of Guana Head	18 28.454'	64 34.883'
2	South Muskmelon	18 28.722'	64 37.747'
3	Middle White Bay	18 28.235'	64 34.504'
4	Mid/South Muskmelon (Crab Cove)	18 28.834'	64 34.744'
5	Mid Muskmelon	18 28.950'	64 34.750'
6	Mid/North Muskmelon	18 29.067'	64 34.764'
7	North Muskmelon	18 29.137'	64 34.890'
8	Mid/South White Bay	18 28.143'	64 34.438'
9	South White Bay	18 28.065'	64 34.424'
10	South White Bay	18 27.894'	64 34.425'
11	South Bigelow	18 28.055'	64 33.883'

The method attempts to sample all fish within a 15 M diameter cylinder, extending from the sea floor to the water surface. A stationary SCUBA diver ascertains the central point and uses a 7.5 M tape to determine one reference edge. A small diameter white PVC pipe measuring 1 M in length with a 30 cm ruler attached perpendicular to one end, is used to estimate fish lengths. After remaining stationary for a few minutes for biota equilibration, each observed fish species is recorded for 1 minute in each cardinal direction; all new species observed within the final minute are also recorded. The researcher hovers above the central point and rotates slowly about 90-degrees each minute. The goal is to observe and immediately record all species within the cylinder on a prepared datasheet, during the 5-minute sample period. No statistical data on the observed species is recorded during this period with the exception of moving schools, which must be counted when observed

Following the 5-minute observation and recording period, statistical data for each species observed in the 5-minute sample period are estimated and recorded, including the number of individuals, maximum, minimum and mean estimated lengths for each species. Often the measuring device can be quickly employed to remove underwater magnification errors without apparently frightening most fish. While recording during the initial 5 minutes, the entire cylinder is theoretically observed, this requires the investigator to periodically scan above and below.

Following the initial 5-minute sampling period, data on commercially important groupers and snappers are recorded separately to indicate they were observed outside of the recording period. Grouper and snapper observed within the initial census period are tabulated with other fish data. The excel database accommodates fish data before and after the 5-minute sampling period.

One single-page, formatted plasticized datasheet is used to record reef fish census data initially, and reef characteristic information subsequently, for each sampling point. Reef characterization data include maximum depth, diver depth (at central recording location), minimum depth, substrate slope, surface relief coverage and metric percentages, abiotic and biotic footprint estimates e.g. gross substrate type, percent sand, macroalgae and coral cover. The number and species of observed lobsters, conch and sea turtles are also recorded with water temperature, visibility, presence and type of fishing gear and total census time. Fish were identified to species based on descriptive characteristics provided by several well-known fish identification guides (Stokes 1984; Randall 1968; Humann 1991).

This method biases towards the larger, gregarious, common and non-cryptic fish species and is widely employed to ascertain the most commonly observed reef fish inhabitants. Researchers understand that small, secretive, rare and cryptic fish are often missed. The method is relatively fast and highly reproducible, allowing a larger sample size per unit effort. The method is not suitable for strong currents, heavy surge, deep depths or low visibility.

Results

Forty four censuses at 11 different sites around Guana Island resulted in data being collected on a total of 67 common eastern Caribbean reef fish species (Table 2). A total of 4,188 individuals were counted during the censuses. An additional five species were observed in the census cylinder but outside of the five minute sampling period. Notes were kept on observations of commercially harvested macro-invertebrates (conch, lobster, octopus) but few were observed.

The most commonly observed fish was the Masked Goby, followed by Striped Parrotfish, Brown Chromis, and Princess Parrotfish (Table 3). Forty four species were observed ten or less times during the study period. Few commercially important macro-invertebrates were observed during the study period.

The St. John data set was collected by Rikki Dunsmore and a number of assistants. Ninety two censuses were made around the island of St. John in a variety of hard-bottom habitats. These censuses resulted in data collection on 57 common eastern Caribbean reef fish (Table 2). A total of 22,008 individuals were counted during the censuses, of which Masked Goby accounted for 14,356. Bluehead Wrasse, Princess Parrotfish, and Blue Tang were the next most abundant species.

Three groups of species, parrotfish, snappers, and groupers, representing grazers and carnivores, were compared in greater depth between St. John and Guana Island. In general, when parrotfish were observed during a census at Guana Island, they tended to be more abundant (Table 4). However, with the exception of Princess Parrotfish, all others were more abundant overall in St. John censuses. Also, with the exception of Princess Parrotfish, the species rankings among parrotfish were the same for both islands.

While three species of snappers were observed in Guana Island, five species were observed in St. John (Table 5). Four of the species were more abundant on St. John, however, Yellowtail Snapper were slightly more abundant overall on Guana Island. The same species of small groupers were observed at both islands (Table 6). While none were very abundant, the most common species at Guana Island was Red Hind. On St. John, the Graysby was the most common small grouper species. No large groupers were observed at either island.

Discussion

Comparison of data sets from Guana Island and St. John show remarkably similar species and numbers (when corrected for differing numbers of censuses). What is interesting is the fact that ten more species were observed, in half the number of censuses, at Guana Island, while only three species were observed at Guana Island that were not observed at St. John. It is interesting that no damselfish were recorded from St. John. This may have been an intentional oversight on the part of the Principal Investigator as it is certain that many damselfish exist around St. John. When one compares number of individual fish counted per census, St. John found 239 fish per census compared to 95 fish per census at Guana Island. However, when one takes into account that on St. John, Masked Goby accounted for 156 fish per census, compared to 27 per census at Guana Island, the numbers are not all that different.

Similarities also exist in numbers of snappers and groupers, two very important groups of predators in Caribbean reefs. Both are found to be at low levels in both locations, suggesting that overfishing may be playing a similar role on both islands. Past reports by fishermen from the USVI and the BVI painted a picture of a significantly overfished USVI while the BVI, with fewer fishermen, was said to have many more reef fish remaining. The comparison of data sets between Guana Island and St. John does not support this however. Both sites have few large predators, many smaller herbivorous and planktivorous species, and moderate amounts of invertevores, much as one would expect to find in reef fish populations being subjected to relatively strong fishing pressure. This phase shift in dominant species is similar to that observed throughout the Caribbean (Jackson, et al. 2001; Hughes 1994).

It is recognized that 44 censuses from one – six day period of time are only a snapshot of what is going on at Guana Island and these results must be accepted as such. Additional sampling would result in a more robust, multi-year data set to compare with existing and on-going St. John data collection. We would like to do more substantantive statistical comparisons of species relative abundances as well as a size comparison of species

between the two sites but this will require additional data from Guana Island to be able to make valid comparisons with the larger, and growing, St. John data set.

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Figures and Tables

Figure 1. Aerial of Guana Island showing approximate locations of fish census sites, July 2004. Refer to Table 1 for site location names and GPS coordinates.

Table 2. Guana Island Fish List, July 2004. Acronyms represent accepted species code.

Acronym	Scientific Name	Common name
	Species observed during census period	
BAVE	*Balistes vetula*	Queen Triggerfish
HOCI	*Holocanthus ciliaris*	Queen Angelfish
POPA	*Pomacanthus paru*	French Angelfish
POAR	*P. arcuatus*	Gray Angelfish
SCTA	*Scarus taeniopterus*	Princess Parrotfish
SCCR	*S. croicensis*	Striped Parrotfish
SCVE	*S. Vetula*	Queen Parrotfish
SPCH	*Sparisoma chrysopterum*	Redtail Parrotfish
SPVI	*S. viride*	Stoplight Parrotfish
SPAU	*S. aurofrenatum*	Redband Parrotfish
SPRU	*S. rubripinne*	Yellowtail Parrotfish
EUDI	*Eupomacentrus diencaeus*	Longfin Damselfish
EULE	*E. leucostictus*	Beaugregory
EUPL	*E. planifrons*	Three Spot Damselfish
EUVA	*E. variabilis*	Cocoa Damselfish
EUPA	*E. partitus*	Bicolor Damselfish
EUDO	*E. dorsopunicans*	Dusky Damselfish
MICH	*Microspathadon chrysurus*	Yellowtail Damselfish
ABSA	*Abudefduf saxatilis*	Sargeant Major
ACCH	*Acanthurus chirurgus*	Doctorfish
ACCO	*A. coeruleus*	Blue Tang
ACBA	*A. bahianus*	Ocean Surgeonfish
HYPU	*Hypoplectrus puella*	Barred Hamlet
HYUN	*H. unicolor*	Butter Hamlet
HYCH	*H. chlorurus*	Yellowtail Hamlet
HYGU	*H. guttavarius*	Shy Hamlet
SPSP	*Sphoeroides spengleri*	Bandtail Puffer
CARO	*Canthigaster rostrata*	Sharp Nose Puffer
LATR	*Lactophrys triqueter*	Smooth Trunkfish
LABI	*L. bicaudalis*	Spotted Trunkfish
LAPO	*L. polygonia*	Honeycomb Cowfish
SYIN	*Synodus intermedius*	Sand Diver
HAFL	*Haemulon flavolineatum*	French grunt
HACA	*H. carbonarium*	Caesar Grunt
HAPL	*H. plumieri*	White Grunt
HASC	*H. sciurus*	Bluestriped Grunt
HAAU	*H. aurolineatum*	Tomtate

SETI	*Serranus tigris*	Harlequin Bass
SETA	*S. tabacarius*	Tobaccofish
EPFU	*Epinephelus fulvus*	Coney
EPAD	*E. adscensionis*	Red Hind
EPCR	*E. cruentatus*	Graysby
GOOC	*Gobiosoma oceanops*	Neon Goby
COPE	*Coryphopterus personatus*	Masked Goby
HORU	*Holocentrus rufus*	Squirrelfish
HOAS	*H. ascensionis*	Longjaw Squirrelfish
MYJA	*Myripristis jacobus*	Blackbar Soldierfish
PSMA	*Pseudopeneus maculatus*	Spotted Goatfish
MUMA	*Mulloidichthys martinicus*	Yellow Goatfish
CHCA	*Chaetodon capistratus*	Foureye Butterflyfish
CHST	*C. striatus*	Banded Butterflyfish
OPAT	*Ophioblennius atlanticus*	Redlipped Blenny
HARA	*Halichoeres radiatus*	Puddingwife
HABI	*H. bivittatus*	Slippery Dick
HAGA	*H. garnoti*	Yelowhead Wrasse
THBI	*Thalassoma bifasciatum*	Bluehead Wrasse
CHCY	*Chromis cyaneus*	Blue Chromis
CHMU	*C. multilineatus*	Brown Chromis
OCCH	*Ocyurus chrysurus*	Yellowtail Snapper
LUAP	*Lutjanus apodus*	Schoolmaster Snapper
LUMA	*L. mahogoni*	Mahogany Snapper
LUSY	*L. synagris*	Lane Snapper
CARU	*Caranx ruber*	Bar Jack
CABA	*C. bartholomaei*	Yellow Jack
SCCA	*Scomberomorus cavalla*	King Mackerel
SCRE	*S. regalis*	Cero
CACA	*Calamus calamus*	Saucereye Porgy
AUMA	*Aulostomus maculatus*	Trumpetfish
ECCA	*Echidna catenata*	Chain Moray
	Species observed outside of census period	
EQPU	*Equetus punctatus*	Spotted Drum
LUJO	*L. joco*	Dog Snapper
ALSC	*Aluterus scriptus*	Scrawled Filefish
SPPI	*Sphyraena picudilla*	Southern Sennet
GYVI	*Gymnothorax vicinus*	Purplemouth Moray
	Octopus briareus	Reef Octopus
	Scyllarides aequinoctialis	Spanish Lobster

Table 3. Total number of censuses in which each species was observed (Guana Island, N = 44 and St. John, N = 92), and the total number of individuals counted.

Species	Island	#Census	# Indivs.	Island	#Census	# Indivs.
BAVE	Guana	3	5	St. John	3	3
HOCI	Guana	4	6	St. John	11	11
POPA	Guana	2	4	St. John	3	3
POAR	Guana	3	5	St. John	17	23
SCTA	Guana	29	274	St. John	46	66
SCCR	Guana	24	332	St. John	92	1029
SCVE	Guana	4	7	St. John	17	20
SPCH	Guana	1	1	St. John	8	9
SPVI	Guana	28	94	St. John	92	490
SPAU	Guana	29	81	St. John	92	450
SPRU	Guana	3	13	St. John	13	34
EUDI	Guana	5	5	St. John	0	0
EULE	Guana	29	51	St. John	0	0
EUPL	Guana	27	88	St. John	0	0
EUVA	Guana	10	19	St. John	0	0
EUPA	Guana	23	73	St. John	0	0
EUDO	Guana	32	135	St. John	0	0
MICH	Guana	21	51	St. John	28	89
ABSA	Guana	6	29	St. John	18	142
ACCH	Guana	23	82	St. John	37	72
ACCO	Guana	31	187	St. John	92	711
ACBA	Guana	24	83	St. John	92	437
HYPU	Guana	28	38	St. John	52	69
HYUN	Guana	2	2	St. John	25	32
HYCH	Guana	2	3	St. John	10	10
HYGU	Guana	2	2	St. John	9	9
SPSP	Guana	4	6	St. John	4	4
CARO	Guana	23	45	St. John	58	91
LATR	Guana	1	1	St. John	11	11
LABI	Guana	2	2	St. John	0	0
LAPO	Guana	1	1	St. John	0	0
SYIN	Guana	2	2	St. John	10	10
HAFL	Guana	26	93	St. John	61	377
HACA	Guana	6	22	St. John	7	17
HAPL	Guana	1	1	St. John	11	32
HASC	Guana	2	3	St. John	17	54
HAAU	Guana	6	52	St. John	1	3
SETI	Guana	5	10	St. John	17	20
SETA	Guana	2	2	St. John	12	22
EPFU	Guana	2	2	St. John	8	8
EPAD	Guana	18	21	St. John	10	10
EPCR	Guana	5	5	St. John	46	49
GOOC	Guana	3	4	St. John	0	0

COPE	Guana	26	1202		St. John	60	14356
HORU	Guana	7	10		St. John	22	53
HOAS	Guana	5	5		St. John	0	0
MYJA	Guana	4	6		St. John	9	14
EQPU	Guana	0	0		St. John	4	4
PSMA	Guana	19	36		St. John	33	53
MUMA	Guana	1	7		St. John	24	79
CHCA	Guana	33	80		St. John	59	137
CHST	Guana	3	4		St. John	7	9
OPAT	Guana	1	1		St. John	0	0
HARA	Guana	3	3		St. John	15	15
HABI	Guana	9	29		St. John	92	515
HAGA	Guana	32	204		St. John	92	435
THBI	Guana	27	171		St. John	92	1129
CHCY	Guana	3	7		St. John	18	56
CHMU	Guana	10	296		St. John	18	120
OCCH	Guana	17	146		St. John	64	272
LUAP	Guana	2	3		St. John	20	107
LUJO	Guana	0	0		St. John	1	1
LUMA	Guana	0	0		St. John	5	19
LUSY	Guana	1	5		St. John	6	23
CARU	Guana	8	15		St. John	52	223
SCCA	Guana	1	1		St. John	0	0
SCRE	Guana	2	2		St. John	2	3
CACA	Guana	2	3		St. John	10	18
AUMA	Guana	8	9		St. John	15	16
ECCA	Guana	1	1		St. John	0	0

Table 4. Frequency of occurrence (# indivs./# censuses observed) for parrotfish species observed at Guana Island and St. John. Numbers in parentheses are frequency of occurrence based on total number of censuses (Guana – 44, St. John – 92).

Species	Guana Is. Total	St. John Total	Guana Is. # Census	St. John # Census	Guana Is. Freq. of Occur.	St. John Freq. of Occur.
SCTA	274	66	29	46	9.48 (6.23)	1.43 (0.72)
SCCR	332	1029	24	92	13.83 (7.55)	11.18 (11.18)
SCVE	7	20	4	17	1.75 (0.09)	0.85 (0.18)
SPCH	1	9	1	8	1.00 (0.02)	1.13 (0.09)
SPVI	94	490	28	92	3.36 (2.14)	5.33 (5.33)
SPAU	81	450	29	92	2.98 (1.84)	4.89 (4.89)
SPRU	13	34	3	13	4.33 (0.07)	2.62 (0.14)

SCTA =	*Scarus taeniopterus*	Princess Parrotfish
SCCR =	*S. croicensus*	Striped Parrotfish
SCVE =	*S. vetula*	Queen Parrotfish
SPCH =	*Sparisoma chrysopterum*	Redtail Parrotfish
SPVI =	*S. viride*	Stoplight Parrotfish
SPAU =	*S. aurofrenatum*	Redband Parrotfish
SPRU =	*S. rubripinne*	Yellowtail Parrotfish

Table 5. Frequency of occurrence (# indivs./# censuses observed) for snapper species observed at Guana Island and St. John. Numbers in parentheses are frequency of occurrence based on total number of censuses (Guana – 44, St. John – 92).

Species	Guana Is. Total	St. John Total	Guana Is. # Census	St. John # Census	Guana Is. Freq. of Occur.	St. John Freq. of Occur.
OCCH	146	272	21	10	6.95 (3.32)	27.2 (2.96)
LUAP	3	107	2	8	1.50 (0.07)	13.39 (1.16)
LUSY	5	11	5	11	1.00 (0.11)	1.00 (0.12)
LUMA	0	19	0	5	0	3.80 (0.21)
LUJO	0	1	0	1	0	1.00 (0.01)

OCCH = *Ocyurus chrysurus* Yellowtail Snapper
LUAP = *Lutjanus apodus* Schoolmaster Snapper
LUSY = *L. synagris* Lane Snapper
LUMA = *L. mahogoni* Mahogany Snapper
LUJO = L. Joco Dog Snapper

Table 6. Frequency of occurrence (# indivs./# censuses observed) for grouper species observed at Guana Island and St. John. Numbers in parentheses are frequency of occurrence based on total number of censuses (Guana – 44, St. John – 92).

Species	Guana Is. Total	St. John Total	Guana Is. # Census	St. John # Census	Guana Is. Freq. of Occur.	St. John Freq. of Occur.
EPAD	21	10	18	10	1.17 (0.41)	1.00 (0.11)
EPFU	2	8	2	8	1.00 (0.05)	1.00 (0.09)
EPCR	5	46	5	46	1.00 (0.11)	1.00 (0.50)

EPAD = *Epinephelus adscensionis* Red Hind
EPFU = *E. fulvus* Coney
EPCR = *E. cruentatus* Graysby

Coral planting on the White Bay Finger Reefs

Lianna Jarecki, PhD

Introduction

The finger reefs at White Bay (Figure 1) were constructed over centuries by colonial coral animals, particularly Elkhorn coral (*Acropora palmata,* Figure 2). Elkhorn coral is one of the fastest growing corals; it was once the most abundant shallow-water coral in the Caribbean; and it was responsible for building all of the reef crests still visible around the BVI today. Sadly, Elkhorn coral and its sibling species, Staghorn coral, were nearly wiped out by a coral disease that swept through the Caribbean in the late 80's and again in the 90's. Both species are now uncommon, and scientists consider them threatened species. They are currently candidates for protection under the Endangered Species Act (USA).

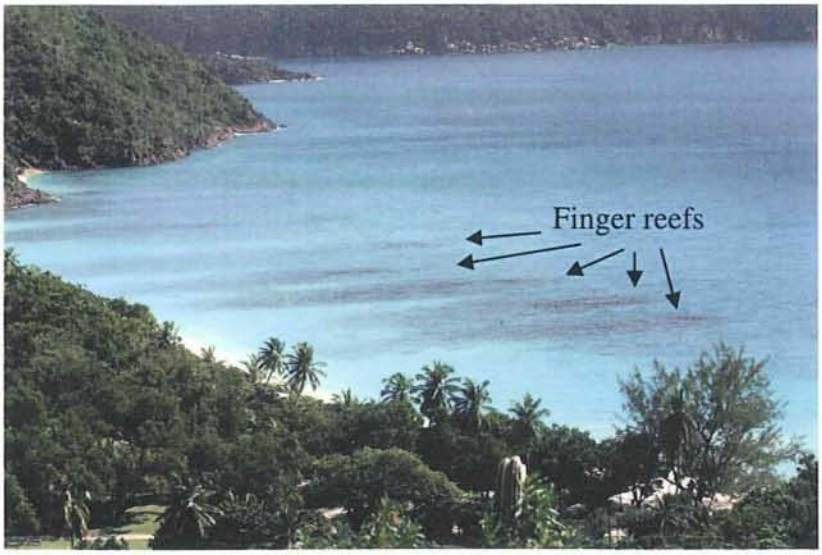

Figure 1: The finger reefs at White Bay, Guana Island.

Figure 2: BVI Elkhorn reef in 1991

This project aims to accelerate the natural regeneration of Elkhorn populations on the finger reefs of White Bay by transplanting coral fragments from other areas. Elkhorn coral was chosen the following reasons: it is the species responsible for initially forming the White Bay reefs; it is a fast-growing and beautiful branching coral (Figure 2); and it is does not appear to be colonizing in most of the finger reefs area on its own. The reason for the latter is not clear, as it is regenerating well in other areas in White Bay and around Guana. Currently the landscape of the finger reefs is mainly the dead skeletons of old Elkhorn coral, covered with algal turf and fire coral (Figure 3). This is unfortunate for snorkelers since the finger reefs are the closest reefs to the Guana Island Hotel beach house.

Figure 3: Old dead Elkhorn branches covered with algae.
White Bay finger reefs, 2005.

Our goal is to improve the biological and aesthetic value of these reefs by increasing the proportion of live coral cover. Its success will be measured by bi-annual assessments of the benthic (bottom) community on and between the finger reefs as well as measurements of each transplanted coral. We expect the transplanted corals to grow into large branching crowns that will eventually seed other parts of the reef with naturally-broken off fragments of their branches.

Methods

Mapping and baseline surveys

The finger reefs were mapped by Sam Frederick and his brother, Che, during July and August 2005. The Fredericks also demarcated 18 study transects, running along each side of each finger reef and also between the fingers (Figure 4). The transects were between 17 and 30m long, and each transect was marked at both ends with a steel stake (marking pin) and an aluminum number tag (see Figure 6).

A point-intercept transect method was used to sample each transect. A measuring tape was run from one marking pin to the opposite pin, and the substrate under each 10cm mark on the tape was recorded on a data sheet. Linda Forrester coordinated the initial

survey, and we subsequently modified the classification categories recorded in the data sheet in order to minimize error in the transect counts.

In November, 2005, a full survey of 16 transects was conducted. Twelve of the original 19 transects were included, plus four additional transects placed within each inter-reef area, parallel to the existing permanent inter-reef transect (Figure 4). These additional transects were necessary to balance the sampling effort on the finger reefs with that between the finger reefs. The 16 transects represent four reef/inter-reef units, with each unit including one inter-reef and the facing sides of opposite finger reefs (Figure 4). Each transect was counted four times, twice by each of two researchers, Lianna Jarecki and Caitlin O'Connell-Rodwell. The repetition in sampling was necessary to quantify the inherent variation in the method within and between researchers, such that any differences recorded in future monitoring must be greater than this inherent variability before it is considered to be valid.

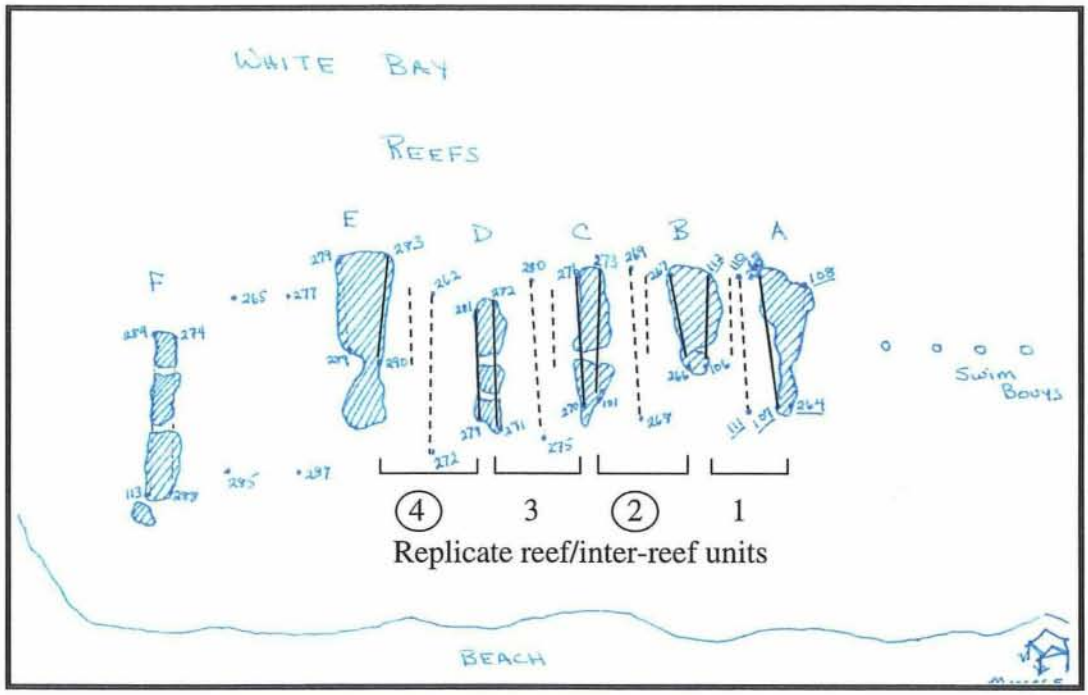

Figure 4: Map of White Bay finger reefs, showing reef transects (solid lines) and inter-reef transects (dashed lines). Four replicate reef/inter-reef units are each comprised of one inter-reef and the facing sides of two opposite reefs. Elkhorn fragments were transplanted to both sides of replicates 2 and 4 (circled).

Transplanting method

Two alternate reef-inter-reef units (2 and 4) were selected for transplanting, leaving the other two (1 and 3) as controls (Figure 4). This approach allows for long term comparison of the effects on the reef community of transplanting to that without transplanting within the same habitat. That is, the non-transplanted replicates will allow us to measure the change in coral cover that occurs without our intervention.

Fragments of coral for replanting were found by searching areas of White Bay (outside of the finger reef area) and the eastern edge of neighboring Muskmelon Bay for broken-off

pieces of Elkhorn coral. No fragments were taken that were attached to substrate and/or appeared to be situated in a position conducive to growth. No fragments were broken off the substrate or off of established colonies. All fragments collected were taken from rubble or sand areas, as these substrates were considered to be inferior to the transplant sites (Figure 5).

Figure 5: Loose Elkhorn fragment being collected from the Harris Ghut region of White Bay.

Fragments were photographed *in situ* and then transferred to a large seawater container on a boat for the short trip to the finger reefs. Gloves were worn to prevent transfer of bacteria from hands to the corals. Upon arrival at the transplant sites, the fragments were placed underwater so that they were in the seawater container on the boat for a maximum of 30 minutes.

Each coral fragment was affixed to a dead part of the reef structure by the following method. Both a dead part of the fragment and the substrate on which it was to be fixed were first scrubbed with a wire brush (Figure 6).

A two-part underwater patching compound (similar to epoxy), available in half-gallon containers, was used for fixing the corals to the substrate. Approximately equal portions of each part were placed into a wet gallon-sized zip-lock freezer bag, pre-filled with seawater (to keep the epoxy from sticking to the sides of the bag). These were mixed underwater at the transplanting site by kneading the outside of the bag until an even color was achieved and the epoxy took on a clay-like consistency (this took about 15 minutes).

Epoxy was placed on the cleaned attachment site on the reef structure, and the coral fragment (cleaned side down), was placed on top, embedding the bottom of the fragment into the epoxy. The epoxy was allowed to harden for at least 24 hours before measurements were taken.

Corals were marked with large numbered plastic tags (cow neck tags), which were attached to the reef with plastic cable ties near each transplant (Figure 7). Each coral was photographed, and each was measured in three dimensions: largest length, width perpendicular to the first measurement, and height from base attachment. The number of branches of greater than 4cm width was also recorded.

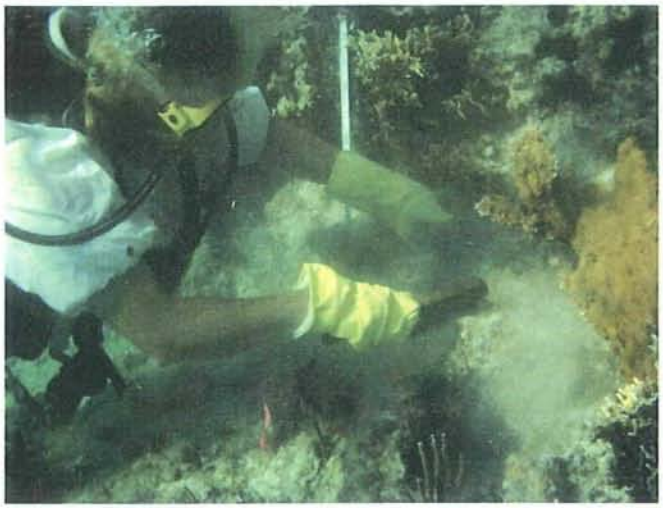

Figure 6: Liz Kintzing cleaning substrate before applying epoxy and mounting the coral transplant. A transplanted fragment is visible on the right side of photo. The transect line (measuring tape) is also visible, running from the end marking pin in the lower center of the photo.

Figure 7: Transplanted fragment number 131.

For comparison with the transplanted fragments, ten small Elkhorn corals were located outside of the transplant area but within the eastern part of the finger reef area (Figure 4). These corals had colonized naturally (they were not transplanted) as larval recruits or as fragments from larger corals. These ten corals were marked, measured and will be remeasured annually along with the transplanted fragments. Thus they will provide a reference growth rate for natural colonizers that can be used to compare with our transplanted fragments.

Results

Current condition of the study area:

The initial (present) state of the White Bay finger reefs can be summarized by the following statistics, calculated from the intensive November survey of 8 reef transects and 8 inter-reef transects:

- 84% of the reef structures and 92% of the inter-reef areas are composed of algae-covered dead coral or sand (Figure 3),

- Only 5% of the reef structure and 4% of the inter-reef areas are covered by living coral,
- 91% of the living coral on the reef structures (3% on the inter-reefs) is fire coral, which is not a true coral but a distant relative with a severe sting,
- Elkhorn coral was not present on any of the sample points.

Success of corals transplanted in August

Of the nine Elkhorn coral fragments that were transplanted in August, seven appeared to be in perfect condition despite a major bleaching event (see introduction to MSM report) that killed many of the surrounding corals. Only one fragment showed evidence of bleaching. This was a large piece that was transplanted near the top of the reef. Twenty-five percent of this fragment was dead in November, but the remainder looked as though it would recover. One other fragment did not fare as well. This was a smaller piece that appeared in November to have been eaten by a damselfish. Damselfish are abundant on the finger reefs, and they are known to bite coral polyps. Eighty-eight percent of this fragment was fully dead in November; a small portion appeared by its color to be surviving, but all of its polyps were bitten down. It will probably not survive.

Thus, the overall survival rate of the August transplants was 89%. Of the survivors, 88% showed no significant change in size.

Transplanting

The nine corals transplanted in August ranged in size from 10 to 51cm in the longest dimension (ave. 25cm, s.d. 16cm). These fragments were collected in a rubble area about 3m deep near Harris Ghut in White Bay.

Twenty-six additional Elkhorn corals, ranging from 4cm to 40cm (ave. 19cm, s.d. 8cm) were transplanted in November. Most of these fragments were collected at the base of the rock wall that runs between the dock at the end of White Beach and the point at the western end of White Bay. Very few fragments were seen in the Harris Ghut area in November. Two fragments were taken from the base of the rock wall at the eastern side of Muskmelon Bay (which neighbors White Bay) on Guana. The smallest coral, measuring only 4 x 3 cm, died shortly after transplanting.

The corals were transplanted to four of the eight reef transects as indicated in Figure 4.

Discussion

Our surveys this year showed that the finger reefs of White Bay are in very poor shape. Live coral accounts for only 5 % of the bottom, and the original reef structures are covered with algae. The little coral that exists on the reefs is nearly all fire coral. Furthermore, the old reef structures, built by the growth of Elkhorn corals before the epidemic of the 1980s and '90s, is being broken down through natural processes of bioerosion and storm waves. As the three-dimensional structure of these reefs diminishes, so do the number of small caves, holes and crevices that host a great diversity of mobile animals (fish, worms, snails, sea stars, etc.) associated with coral reefs.

New coral growth counteracts erosion by replacing lost reef structures. Elkhorn coral is a particularly fast-growing coral, with growth rates up to 10 centimeters per year

(Gladfelter et al. 1978). We will determine the growth rate of the transplanted corals by measuring them annually.

Once the corals establish themselves and grow into the typical Elkhorn crowns (see Figure 2), they will seed new colonies nearby. They do this by natural fragmentation, a common form of asexual reproduction in branching corals such as Elkhorn. Waves break off branches of coral, which may then grow where they fall, thus developing a new coral colony. Thus we expect the transplanting project to expand naturally, with the transplanted corals eventually seeding a denser population of Elkhorn coral on and around the finger reefs. This process may take decades, but we expect to see evidence of growth and fragmentation within the next five years. We will measure the effect of the transplanting effort on the wider reef area by comparing the transplanted areas (reef/inter-reef areas 2 and 4 (as shown in Figure 4) with the control areas (reef/inter-reef areas 1 and 3).

During MSM 2006, we will transplant additional fragments. Our preliminary results suggest that we can expect a survival rate greater than 80% in the transplanted corals. We will also address two further questions:

1) How does the survival rate of loose coral fragments in the collection areas compare with their counterparts that are collected and transplanted? In areas where we find coral fragments (hopefully between 20 and 40 fragments), we will collect half for replanting and leave half where we found them. The fragments that are left will be marked and measured. The transplanted fragments will also be measured. Both sets will be monitored quarterly to determine the survival and growth rates. We expect that the survival rate of the corals left in the collection area will be lower than that of the replanted corals.
2) How does the growth rate of naturally-attached Elkhorn corals in the finger reef area compare with that of transplanted corals. Several small Elkhorn colonies occur in the finger reef area to the east of the transplanted area. These corals have colonized the reef either as larvae or as successful fragments of larger corals, and they are firmly attached to the reef (unlike the fragments we collect for transplanting). Such corals will be marked and measured so that their growth rates can be compared to the transplant growth rates.

The fragments planted this year already give the reef a more colorful and alive appearance. Assuming that these new corals grow as expected, an increasing proportion of the reef will be covered by live coral in the future. This will be enormously beneficial to the reef community. Reef fish diversity, in particular, should improve as it is tightly linked to the amount of coral on a reef (Jones et al. 2004). More coral and more abundant and diverse fish will also improve the value of these reefs to the Guana Island Hotel and the snorkelers who enjoy watching the reef ecosystem.

Acknowledgements

This project was funded by the Guana Island Wildlife Sanctuary. I thank Gloria Jarecki especially for her interest and concern, and The Conservation Agency for assistance with logistics. Essential collaborators were Dr. Graham Forrester, Linda Forrester and Dr.

Caitlin O'Connell-Rodwell. I am also indebted to Sam and Che Frederick, Liz Kintzing and Tim Rodwell for their dedicated assistance with the field work.

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Coral disease on Guana Island reefs July 2004

Longin Kaczmarsky

Field surveys for coral disease prevalence in July, 2004 around Guana Island, BVI indicated that the species most affected by disease was *Acropora palmata*, or Elkhorn coral (Table 1). The signs indicated that most diseased colonies were affected by white pox disease, a smaller number by white band disease and, two cases where colonies were affected by both. The prevalence was highest in White Bay and lowest in North Bay. Statistical analyses are pending.

Two gorgonian species, *P. americana* (Slimy sea plume) and *G. ventalina* (Common sea fan), also appeared to have a relatively high infection rate of black band disease and putatively, aspergillosis. It should be noted that these assessments are on a purely visual basis and without laboratory confirmation of the pathogen. Thus, it may be more proper in this case to refer to aspergillosis as "gorgonian degenerative syndrome". Sea temperatures at 1-5 m deep during the survey period ranged from 28.0C to 29.5C.

Contact: Longin Kaczmarsky, Florida International University, Dept of Biological Sciences, Miami, FL solonnie@hotmail.com 201-704-8065.

Table 1. Coral species most diseased in Guana Island July 2004

(1 to 5 m depths; sites n = 8). Sites were located in White Bay, Muskmelon Bay, North Bay, and Bigelow Bay. WPD = white pox disease, WBD = white band disease, ASP = aspergillosis, DSS = dark spot syndrome, WP = white plague, BBD = black band disease, YBD = yellow band disease. Only species observed to have disease are listed here, for example the most abundant species, *S. radians*, is not included.

% diseased	species	total # colonies	diseases	species abundance: rank
35.2	*Acropora palmata*	182	WPD, WBD	8
33.9	*P. americana*	59	ASP, BBD	19
19.5	*G. ventalina*	190	ASP, BBD	7
13.6	*A. cervicornis*	22	WBD	22
12.5	*C. natans*	8	WP	26
11.7	*S. siderea*	60	WP, DSS, BBD	18
11.5	*G. flabellum*	96	ASP	14
9.0	*M. annularis*	257	WP, BBD, YBD	4
6.3	*Plexaura sp.*	207	ASP, BBD	5
5.9	*D. labyrinthiformis*	17	WP	23
5.7	*D. strigosa*	176	WP, BBD	10
3.9	*D. clivosa*	104	BBD, WP	13
2.5	*F. fragum*	199	WP	6
1.7	*P. homomalla*	58	ASP	20
1.4	*A. agaricites*	651	WP	3
1.4	*M. sulpherea*	71	ASP	16

THE ELKHORN REEF IN MUSKMELON BAY, GUANA ISLAND, BRITISH VIRGIN ISLANDS

Caroline S. Rogers, PhD
Erinn M. Muller
March 18, 2005

Introduction

On August 7, 2004, we surveyed the elkhorn (*Acropora palmata*) reef in Crab Cove, Muskmelon Bay, off the west side of Guana Island (Fig. 1). The overall goal was to collect data that would serve as a baseline for determining if this reef is recovering or not. Here we use the term "recovery" to refer to an overall increase in the amount of living elkhorn coral. The shallow elkhorn patch reef in Crab Cove has a topographically complex structure, evidence of a former impressive and actively growing reef (Fig. 2). Elkhorn coral suffered dramatic losses from white band disease (Fig. 3) and hurricanes in the 1970s and 1980s, throughout the Caribbean (Gladfelter 1982, Bythell and Shepard 1993, Bruckner 2002). It is likely that disease killed most of the elkhorn coral on Crab Cove reef as there were primarily dead, intact colonies and little evidence of old, broken branches such as those found on reefs hit by storms. Many reefs in the British Virgin Islands (BVI) and US Virgin Islands (USVI) still have large amounts of upright elkhorn colonies, in growth position but completely dead. In the last 5 years, there has been evidence of an increase in living elkhorn coral colonies on many of these reefs. Although elkhorn coral grows relatively quickly (about 5 – 10 cm/yr, Gladfelter et al. 1978), diseases, predation, damage from boats and careless snorkelers and storms continue to cause partial and in some cases total mortality of elkhorn corals, and it is not clear if this species is making a comeback. Elkhorn and staghorn coral (*Acropora cervicornis*) are now being proposed for listing as threatened under the U.S. Endangered Species Act because of their scarcity throughout most if not all of their former range (Bruckner 2002).

Methods

The US Geological Survey and National Park Service have been doing research on elkhorn colonies in three national parks using a protocol that involves mapping elkhorn colonies using GPS technology. Data recorded include colony size; presence of disease; predation by snails (*Coralliophila abbreviata*), fireworms (*Hermodice carunculata*), and damselfishes (primarily *Stegastes planifrons*); and physical breakage (Devine et al.2002, Rogers et al. 2004). We used this basic protocol to survey Crab Cove reef and grouped the colonies into 4 size categories based on the maximum dimension (see below).

Size Category	Maximum dimension in centimeters
Small (S)	< 10
Medium (M)	11 – 50

Large (L)	51-100
Huge (H)	> 100

Elkhorn coral grows on the patch reef in Crab Cove (directly off the base of the hiking trail) and along the shoreline extending southwest towards the Iguana Head. We decided to focus on the patch reef and started our survey along the fairly well-defined southern edge of this reef. Using a Garmin LX, we took a GPS waypoint at the beginning and end of our survey and traced the perimeter of the reef area that we examined. In 3 hours of observation, we were not able to completely finish surveying this patch reef. However, it will be possible for us to survey the same area in the future, using the boundaries delineated by the GPS. We took an individual GPS waypoint for each living elkhorn colony, or in some cases for several colonies if they were growing very close together. We photographed each colony, estimated size, recorded presence of disease and snails, estimated the percent of the colony that was dead, and wrote descriptive comments. In an effort to survey every living elkhorn coral on the reef, we swam compass bearings back and forth, roughly perpendicular to shore. However, it was difficult to navigate with the compass while looking for corals and hard to keep track of which corals had already been surveyed. We ended up making only limited use of the compass to keep track of our position as we made observations.

We also took GPS waypoints for three distinctive corals (2 *Siderastrea siderea* and 1 *Dendrogyra cylindrus*) to make it easier to re-locate particular areas of the reef. We do not know if we will be able to monitor the fate of individual corals over time. However, we have been successful in using laminated photographs and the GPS unit to navigate back to specific colonies at two reefs on St. John. The GPS has an estimated accuracy of $\pm$3 m. Our objective is not to return to each elkhorn colony every year but rather to get an estimate of the density (number of colonies per square meter) and approximate size distribution to see how these change over time.

There are many challenges when trying to quantify the abundance of living elkhorn coral on a reef. The abundance is a reflection not only of the number of colonies but also their size and their condition. For example, if a large colony suffers partial mortality from predation or disease, and dies leaving only a few remnant patches, each patch is genetically the same as the initial larger colony but physiologically a separate colony. In this case, the number of colonies has increased while the amount of living coral tissue has actually decreased. When we encountered several patches (often non-branching crusts) on an old dead colony, we used the designation "ss" for "same structure". In this way, we could differentiate these from several small colonies that had developed separately from each other and were not remnants. It is usually not possible to tell if corals grew from sexual recruits, unless they are found on bare rock substrate. We have observed small, remnant crusts of elkhorn that went on to develop branches and which looked identical to sexual recruits. We know that some reefs in the USVI contain several genotypes of elkhorn coral (I. Baums, pers. comm.). However, genetic clones are abundant because elkhorn colonies can reproduce asexually through fragmentation. At

times we observed very complex colonies, with several patches of live elkhorn that were difficult to identify as separate from each other. We attempted to describe these but could not always get a definite count. When colonies are fairly abundant, it can also be difficult to keep track of exactly which colonies have already been surveyed. Surveying shallow elkhorn reefs during rough sea conditions can be very challenging. We were fortunate that it was very calm the day we did our survey.

Results

We observed a total of about 256 colonies of living elkhorn at 140 GPS waypoints. Most of the colonies fell into the "Medium" size category with a total mortality (old and recent) of 10 % or less. Many colonies had old lesions that looked like those from white pox disease. Active white pox (indicated by irregular patches of recently exposed white skeleton) was seen on 11 colonies (Fig. 4). Although white pox may have been affecting elkhorn for over 30 years (Rogers et al. 2005), it was first reported in 1996 (Holden 1996) and has been recently observed affecting elkhorn reefs in the Florida Keys (Patterson et al. 2002). No white band disease was seen. We saw only six broken-off branches ("fragments") (Fig. 5). We were not able to search very thoroughly for snails but found a total of only four *C. abbreviata* on two colonies (Fig. 6). Damselfish territories were seen on seven colonies, one of which was almost completely covered by the distinctive "chimneys" that form when damselfish bite the live coral (Fig. 7). Some colonies had more than one factor causing mortality, for example disease and damselfish predation. Only seven colonies were greater than 1 meter in maximum dimension, and some of these may have been encrusting over old dead skeleton (referred to as "re-sheeting") rather than having developed from sexual recruits. We saw very few small colonies that could represent new recruits (Fig. 8).

We obtained a geo-referenced aerial photograph of the Crab Cove reef from the Survey Department of the BVI government. We plotted the GPS waypoints on this photo to map the spatial distribution of the colonies, their condition (with or without disease), and size classes (Figs. 9 and 10). No clear spatial patterns emerged, although there was a tendency for largest colonies to be on the seaward edge of the reef, and there was possible grouping of smaller colonies in the mid-portion of the reef. Colonies with active white pox were distributed throughout the zone that we studied.

Active white pox disease was found affecting only 4% of the total elkhorn colonies that we surveyed in Crab Cove. Disease and colony size appeared to be correlated. As size increased, the percent of diseased colonies increased as well (Fig. 11). We did not notice any colonies 10 cm or less with white pox, and only 3% of the most abundant size class (11-50 cm) displayed signs of disease. However the two larger size classes, 51-100 cm and >1m, exhibited a disease prevalence of 19 and 22% respectively. Although white pox lesions (disease "patches") can heal, it appeared that many colonies in Crab Cove had old lesions as well as new ones.

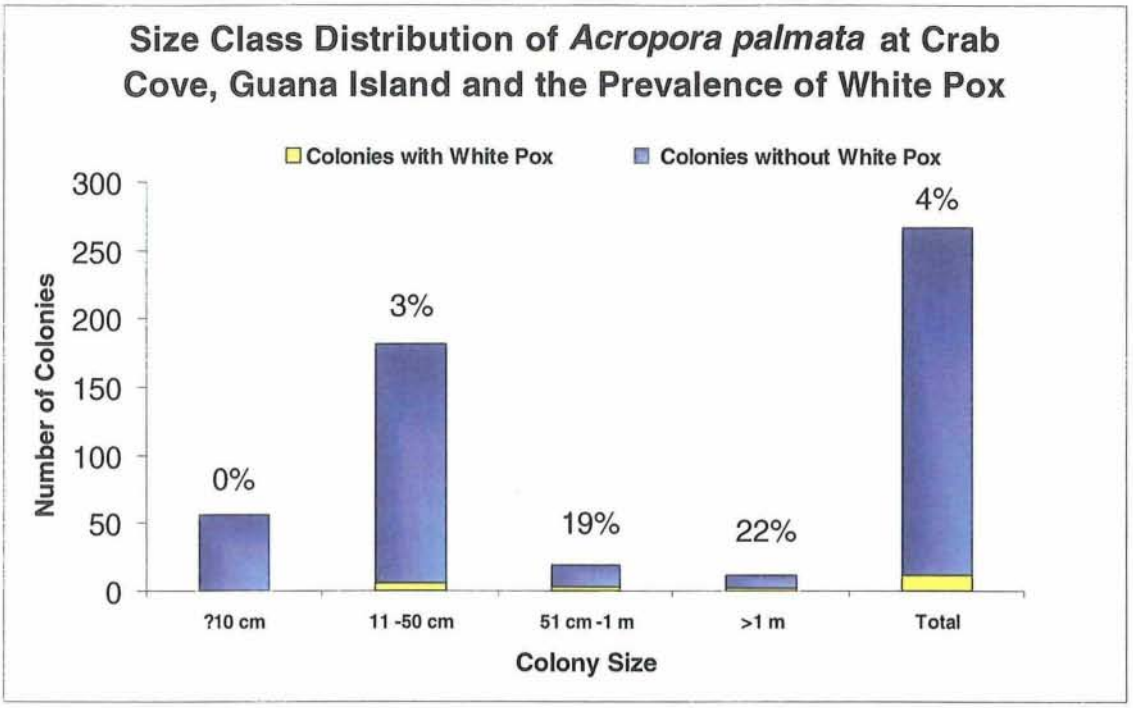

Fig. 11. Relationship of size class and presence of white pox disease.

Conclusions/Future Work

We are confident that we could repeat our survey in 1-2 years and obtain useful information on the status of the Crab Cove reef, specifically on whether or not the elkhorn population is showing signs of recovery. Our research on elkhorn reefs around St. John has shown that monthly (or more frequent) surveys are often necessary to attribute the cause of coral mortality. Annual surveys can provide data on overall abundance and overall condition, but cause of partial or total colony mortality may not be possible to determine.

One of us (CR) first saw Crab Cove reef and other elkhorn reefs around Guana Island a year ago, in August 2003. It was thought that these reefs might be in better condition (less disease, more large elkhorn colonies) than those around St. John, because Guana has no sedimentation or sewage problems that could adversely affect the reefs, and less visitation by snorkelers. However, we have confirmed that these shallow elkhorn reefs around Guana have suffered considerable loss from diseases and other unknown stressors and are not in conspicuously better condition than reefs in St. John.

Diseases (white pox and other uncharacterized diseases) are currently the biggest threat to elkhorn corals in the western Atlantic/Caribbean. Recent hurricanes have also caused a great deal of mortality. The living elkhorn on many reefs in the BVI and USVI appears to be less than 5% of the levels of abundance in the 1970s, although some signs of recovery are evident. We believe that the research we did at Crab Cove provides a useful model for studies of other elkhorn reefs and a baseline for assessing future trends.

Acknowledgments

We are grateful to Dr. Lianna Jarecki for encouraging us to do this research and for going to so much trouble to accommodate us at Guana Island. Many thanks to Dr. Henry Jarecki for his great generosity in making it possible for us to learn more about this special island. We also appreciate the help we received from Roger Miller and others who made our stay so enjoyable.

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Figures (Rogers & Muller 2005)

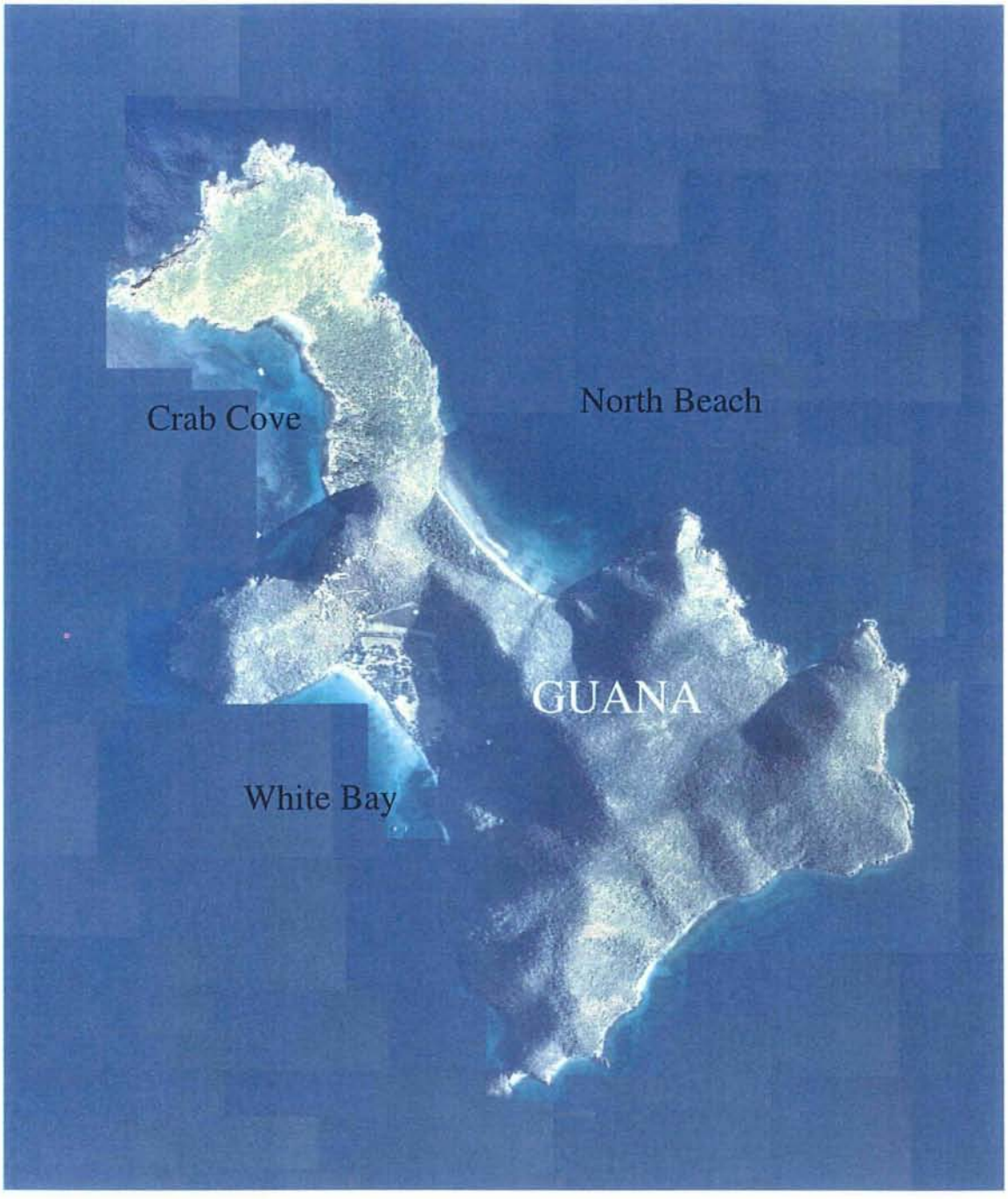

Figure 1. Guana Island and location of Crab Cove

Fig 2. Crab Cove reef has a complex physical structure comprised mostly of living and dead elkhorn coral.

Fig. 3. White band disease on an elkhorn coral from Buck Island, St. Croix.

Fig. 4. White pox on an elkhorn coral at Crab Cove.

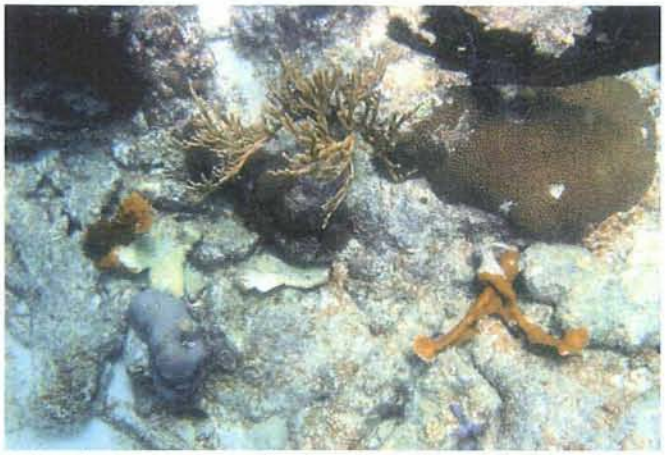

Fig. 5. Fragments (broken branches) from an elkhorn coral.

Fig. 6. Coral-eating snails (*Coralliophila abbreviata*) leave predation scars (bright white areas) on elkhorn coral.

Fig. 7. The threespot damsel (*Stegastes planifrons*) bites live coral, resulting in "chimneys" with turf algae.

Fig. 8. An elkhorn coral recruit (from sexual reproduction) growing in Crab Cove. This small colony is less than 10 cm across.

Paul C. Sikkel, Ph.D., Hatfield Marine Science Center, Newport, OR and Department of Biology, Murray State University, Murray, KY

Summary of activities: Guana Island Marine Science Program 2004-2005

During July of 2004, I visited Guana Island for two days to survey sites in preparation for a study on the relationship between habitat and parasite loads (primarily gnathiid isopods) in Caribbean reef fishes (primarily damselfishes). I surveyed reefs in Muskmellon Bay and White Bay and found the sites suitable for the proposed study. Due to a job change, I was unable to return to Guana in 2005. Also, in Spring of 2005, I received a grant from U.S. Virgin Islands NSF EPSCoR to conduct a study similar to the one I originally proposed for Guana Island. However, this study was broader in focus than the original study. In particular, it focused on other, more exploited/managed fish species, and included among reef comparisons. It also included parasitic monogeneans that are believed to be influenced by water/habitat quality.

As a first step in assessing among-site variation in parasite infestation, we quantified monogenean loads (Fig. 1) in Ocean Surgeonfish (*Acanthurus bahianus*) and Blue Tang (*Acanthurus coeruleus, Fig. 2*) among four sites in the U.S and British Virgin Islands. These sites included Lameshur Bay, St. John, Brewers Bay, St. Thomas, Flat Cay, St. Thomas, and Muskmelon Bay, Guana Island. These sites were chosen for their abundance of surgeonfish, variation in gross habitat features, and our ability to easily access them. All sampling was conducted from July through October 2005. In August of 2005, my collaborators, Dr. Donna Nemeth and Ms. Amber McCammon visited Guana Island for three days to collect surgeonfishes from Muskmelon Bay. Because of the difficulty of collecting fish during the day, most fish were collected from their nocturnal shelter holes. All fish were collected with nets. In the laboratory, fish were anesthetized using clove oil, "dipped" in freshwater for 5 min, placed in fresh seawater for recovery, and later released to the site of capture. We filtered the water through plankton mesh, rinsed the filtrate into petrie dishes, and counted monogeneans under a dissecting scope. Counting was facilitated by adding red food coloring that was absorbed by the monogeneans.

Among all sites, of the 63 *A. bahianus* collected, only two had parasitic monogeneans (one each). In contrast, 47 of 80 *A. coeruleus* were infected (1-28 each). These species represent a new host record for the parasite *Neobenedenia melleni* (E. Williams, Pers. Comm.). This highly significant difference between species (chi-square, $p < 0.001$) was not due to the larger body size of the latter as *A. bahianus* averaged over 2 cm longer ($t = 3.70$, $df = 141$, $p = 0.001$). The difference also maintains even where comparisons are limited to the two sites where both species were common and where most *A. coeruleus* were infected (Lameshur and Brewers bays). Because of the highly significant between-species difference (chi-square, $p < 0.01$), we limited among-site comparison to *A.*

coeruleus. We found significant variation among sites (Kruskal-Wallis test = 31.83, $p < 0.001$). Parasite loads averaged highest at the Guana Island site (Fig. 3), with 15 of 20 fish infected (median = 5 parasites per fish, range = 1-28). At the other extreme, only three of 35 fish collected at Flat Cay were infected by 1-2 parasites each. While fish size differed significantly among sites ($F = 13.20$, df = 3, 139, $p < 0.001$), fish from Guana Island ,where parasite loads were highest, averaged 3 cm smaller than fish from other sites ($p < 0.01$, Fishers LSD). In comparison, fish from the site with the lowest loads (Flat Cay) were smallest, although not significantly so ($p > 0.07$). Thus, differences in parasite load among *A. coeruleus* were not an artifact of differences in body size.

These results indicate strong among-site variation in loads of parasitic monogeneans. By sampling a broader range of sites and quantifying local habitat characteristics, we aim to identify the habitat variables that best predict parasite loads in *A. coeruleus*. We hypothesize that between-species differences are attributable to differences in habitat use within reefs. We aim to test this, along with the alternative hypotheses that differences are attributable to differences in susceptibility to parasites and/or differences in the amount of interaction with cleaners.

Figures

Figure 1. A parasitic monogenean *Neobenedenia sp.*

Figure 2. Blue tang schooling over reef.

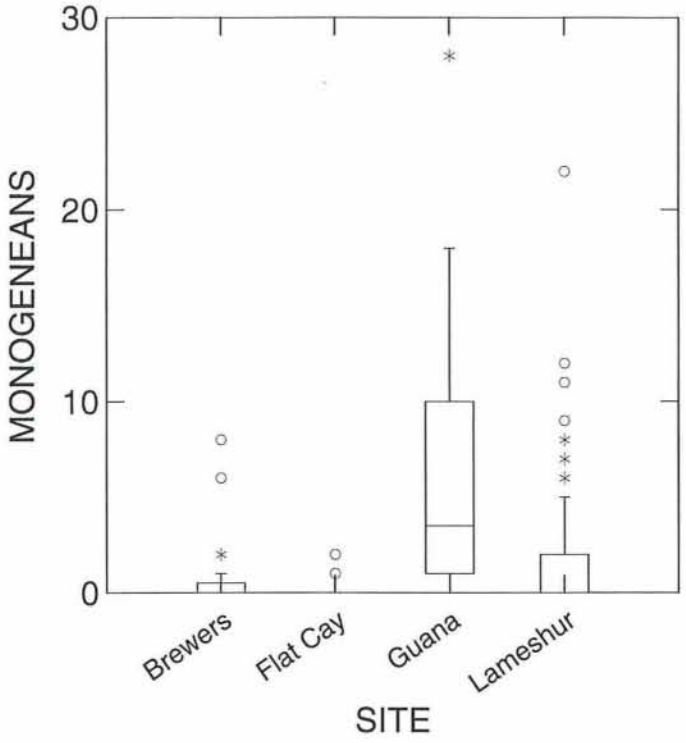

Figure 3. Box and whisker plot of the number of parasitic monogeneans per *A. coeruleus* collected from four sites in the U.S. and British Virgin Islands.

Large Boat Damages Coral Reef in BVI Fisheries Protected Area

Written by Lianna Jarecki, July 11, 2004

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Massive damage was inflicted upon one of the BVI's most beautiful coral reefs in a quiet incident last month. Sadly such incidents will continue to occur with even greater cumulative impact as the BVI invites mega-yachts to our waters.

On July 7th the Holo Kai, a ship of 165 feet in length, sailed into Muskmelon Bay on Guana Island, where it dropped two very large bow anchors in a small patch of sand, then backed up as it layed out 150 feet of heavy chain across the coral reef and finally placed a third, stern anchor with heavy chain lying across a deeper part of the coral reef. Muskmelon Bay, because of its healthy coral reef and abundant fish populations, is a recently-declared Fishing Priority Area, a popular dive site, and, as a result of the new Fisheries Regulations, a site where any anchoring is illegal. Smaller boats anchoring and causing damage in Muskmelon Bay and other protected marine areas mostly go unreported, but marine scientists on Guana observed the Holo Kai's position the morning after it anchored in Muskmelon Bay and reported the event to the Conservation and Fisheries Department. These scientists had surveyed coral reefs in this same area only two days earlier and knew that the ship was positioned over a healthy and diverse community of large corals. In fact, the coral reef at this site has been monitored annually since 1992, providing abundant evidence for its long-term vitality. This pre-existing information on the state of the reef provided a rare opportunity to precisely determine the destructive effects of anchoring.

Though the ship departed early on the morning of July 8th, the marine scientists had located its position in the water and found a correspondingly large area of severely damaged coral reef. Brain corals the size of large boulders were severely scraped; medium-sized and smaller coral heads were broken or completely overturned; sea fans and soft corals were flattened. The most visually dramatic damage was evident in broken columns of pillar coral that stood three feet high the day before. Photographs of this damage can be viewed on the internet at http://www.irf.org/guanareefdamage.html.

The area of continuous damage caused by the Holo Kai anchors measured 30,000 square feet, more than 2/3 of an acre. Broken corals and sea fans covered roughly one-third of the bottom within this area. Virtually all of the large coral colonies were overturned or broken and are now dying. The incidence of damaged corals inside this area was more than ten times that recorded in nearby areas, where some coral damage may be due to the anchors and chains of charter yachts, representing another point of concern. The potential for recovery from such damage is low because corals grow extremely slowly (most grow less than ½ inch per year).

Several countries have begun to fine boats for causing damage to reefs. The Belize government, for example, charged a fine of $75,000 for each of two incidents of anchor damage in 1997, one caused by a 282-foot sailing ship and the other by a 185-foot schooner. The Windspirit paid $350,000 to the US government in a court settlement after

it dragged anchor in 1988 across a reef in the Virgin Islands National Park, St. John. Fourteen years later, biologists from the VI National Park found no sign of corals recovering or re-growing in the damaged area..

The damage at Muskmelon Bay occurred despite the fact that the captain of the Holo Kai thought that he had taken all precautions to avoid environmental damage--checked the charts for any indication of a no-anchoring zone or protected area (these are not currently indicated on any BVI charts); used sonar to determine the nature of the bottom, which showed up as "rock" (much of the hard ground in BVI waters is the living "rock" produced by corals, but he and many other visiting captains are not aware of this); placed his two bow anchors in sand (but did not concern himself with the heavy chain between the anchors and his ship). The mission of the Holo Kai in the BVI was one of conservation and research, and the captain was very concerned about the damage he had caused when notified.

In the case of the Holo Kai, a simple indication of the location of Fisheries Priority Areas and Marine Protected Areas on charts or other documents given to captains when they check in at Customs would have prevented this particular tragic incident. But, at present there is no organised effort to disseminate such information effectively. Furthermore, there are many sensitive reef and seagrass areas in the BVI that are not afforded special protective designations. Large boat anchors can destroy these areas in a few hours.

We are therefore faced with an impending tragedy as the BVI makes plans to accommodate mega-yachts. Mega-yachts are too large to use the existing moorings system, which accommodates boats up to 60 feet in length. The incident of the Holo Kai should be taken as an urgent warning that the BVI must find the means to control boat anchoring and limit anchoring to large sandy areas where the potential damage to our marine resources is low. Healthy coral reefs, such as Muskmelon Bay, are too valuable to the BVI to allow their destruction by uncontrolled visitation.