Survey and Impact Assessment of Derelict Fish Traps in St. Thomas and St. John, U.S. Virgin Islands

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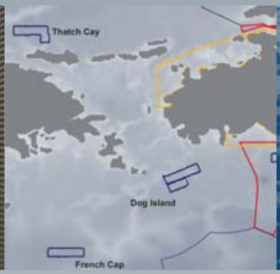

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ABOUT THIS DOCUMENT

Marine debris is a growing problem in the marine environment with impacts to many user groups. To fully understand the causes and impacts of marine debris requires collaboration from all stakeholders. This project represents a unique partnership, funded by NOAA's Marine Debris Program, to address the economic and ecological impacts of marine debris. Project partners consisted of federal and territorial agencies, academia, and local commercial fishermen, whose contributions added greatly to the knowledge of derelict fish traps in the U.S. Virgin Islands (USVI) and the overall success of the project.

The purpose of the collaboration was threefold: 1) to assess the causes and potential impacts of lost fish traps in the USVI; 2) develop experiments to evaluate potential impacts from ghost fishing, assess trap fouling as a indicator of time-at-sea and quantify trap movement and impacts to benthic communities due to storms; and 3) to assess the efficiency of autonomous underwater vehicles (AUVs) as a tool to detect and verify derelict traps in a coral reef ecosystem. Information regarding fishing effort and specific areas of trap loss provided by the St. Thomas Fisherman's Association (STFA) were instrumental in understanding the spatial scope of the problem and to provide a baseline to direct AUV surveys. The expertise of the U.S. Navy's Naval Surface Warfare Center's AUV operations was a valuable asset in searching for derelict traps; and novel field experiments conducted at the the University of the Virgin Islands increased our understanding of the ecological implications of derelict traps.

Products from this project include: a text report, a masters thesis, a database on fishing effort and trap loss in the USVI, and a database on the abundance and distribution of derelict traps that were identified during this project. The results of this project are available via hard copy report and from the project website: http://ccma.nos.noaa.gov/ecosystems/coastalocean/derelictfishtraps.aspx. For more information on this project direct questions and comments to:

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Executive Summary

Since 2001, NOAA National Centers for Coastal Ocean Science (NCCOS), Center for Coastal Monitoring and Assessment's (CCMA) Biogeography Branch (BB) has been working with federal and territorial partners to characterize, monitor, and assess the status of the marine environment across the U.S. Virgin Islands (USVI). At the request of the St. Thomas Fisherman's Association (STFA) and NOAA Marine Debris Program, CCMA BB developed new partnerships and novel technologies to scientifically assess the threat from derelict fish traps (DFTs).

Traps are the predominant gear used for finfish and lobster harvesting in St. Thomas and St. John. Natural phenomena (ground swells, hurricanes) and increasing competition for space by numerous user groups have generated concern about increasing trap loss and the possible ecological, as well as economic, ramifications. Prior to this study, there was a general lack of knowledge regarding derelict fish traps in the Caribbean. No spatially explicit information existed regarding fishing effort, abundance and distribution of derelict traps, the rate at which active traps become derelict, or areas that are prone to dereliction. Furthermore, there was only limited information regarding the impacts of derelict traps on natural resources including ghost fishing.

This research identified two groups of fishing communities in the region: commercial fishing that is most active in deeper waters (30 m and greater) and an unknown number of unlicensed subsistence and or commercial fishers that fish closer to shore in shallower waters (30 m and less). In the commercial fishery there are an estimated 6,500 active traps (fish and lobster combined). Of those traps, nearly 8% (514) were reported lost during the 2008-2010 period. Causes of loss/dereliction include: movement of the traps or loss of trap markers due to entanglement of lines by passing vessels; theft; severe weather events (storms, large ground swells); intentional disposal by fishermen; traps becoming caught on various bottom structures (natural substrates, wrecks, etc.); and human error.

Autonomous underwater vehicles (AUVs) were successfully used in this study to identify and quantify traps in control areas; however success was limited to areas with reduced ridges (rugosity; <15 degree slope). In controlled test sites, AUV's had a 94% success rate detecting traps over sand and rhodolith (a coral-like red algae) seafloor, and a 42% success rate detecting traps over high-relief coral reef habitats. To assess the quantity of DFTs, approximately nine km^2 of seafloor was surveyed by AUVs at six separate locations receiving fishing effort. The AUV surveys identified 122 targets as traps and another 43 targets as non-trap/objects of interest (i.e., man-made objects). Verification of a subset of the total targets was conducted with a remotely operated vehicle and 25% (N=22) were determined to be derelict traps (43 total traps), while seven additional DFTs were discovered. The amount of area surveyed is not sufficient to quantify the overall abundance of DFTs in the region. Continued AUV surveys are recommended to provide a more comprehensive quantification of DFTs.

Fish mortality in experimental DFTs (escape panels closed) was low (5%) and lower than previous STFA observations (9%). In contrast, mortality was rare in traps with escape panels open, allowing fish to move freely in and out. Experiments were conducted at depths less than approximately 18 m (60 ft) but should also be investigated at deeper depths.

One objective of the experimental design was to determine if derelict traps could be aged based on their fouling communities. Traps were colonized by a variety of organisms and at rates that were highly variable depending on depth and location. After a year, experimental traps remained structurally robust; and throughout the experiment, some traps, particularly those in nearshore seagrasses provided structure that supported a large number of juvenile fishes, thus providing some nursery function.

Previous studies have shown that trap fishing and derelict traps may have negative impacts to benthic habitats. While our study did not quantify impacts, we observed that trap movement was minimal during the study period. Trap movement was only observed as Hurricane Earl displaced traps up to 150 m. Through the duration of this study, we observed several derelict traps that had been incorporated into the surrounding habitat.

There are several potential management actions that may help reduce any negative impacts associated with derelict fish traps. The majority of derelict traps are the result of unintentional loss; however, traps have been discarded at sea by fishermen when their gear is no longer suitable to fish with. One potential solution would be to establish adequate land-based trap disposal facilities to reduce this issue. In addition, our study revealed that traps are set in areas that are also used by commercial shipping. Shipping lanes in the region are virtually non-existent which increases the risk of collision between ships and trap lines causing entanglement and ultimately movement and loss of traps. It is recommended that shipping lanes, especially for the cruise ship industry, be established and designed to minimize overlap with commercial fishing and other user groups.

Overall, this project has provided significant insights on some of the important concerns regarding DFTs in the region. While preliminary findings may suggest that impacts from DFTs appear to be minimal, more information needs to be generated, specifically for waters less than 20 m deep and greater than 40 m deep. Our AUV surveys suggested that the amount of derelict traps in the region was greater than expected. Further AUV surveys are recommended to better assess the abundance and location of traps especially in relation to marine protected areas or other sensitive habitats. Lastly, a look at possible solutions to reduce the number of derelict traps, best management practices to deal with traps once they become derelict, and ways in which enforcement agencies may assist in providing solutions and/or support are all issues that need further study.

Chapter 1: Introduction

Randy Clark[1]

1.1. Introduction and Background

Derelict fishing gear is included within the US working definition of marine debris (Marine Debris Research, Prevention and Reduction Act, 2006) and is typically associated with areas of concentrated fishing effort (Hess et al., 1999). Traps are common fishing gear that passively capture organisms, and can become derelict as a result of: intentional abandonment of old and damaged traps; movement of traps and or loss of trap markers commonly caused by the entanglement of trap lines with boat propellers or other gears, such as trawls or hook and line gear; fouling on the benthos (e.g., traps and ropes become caught on rocky substrates); and human error and inclement weather (Laist, 1995; Figure 1.1).

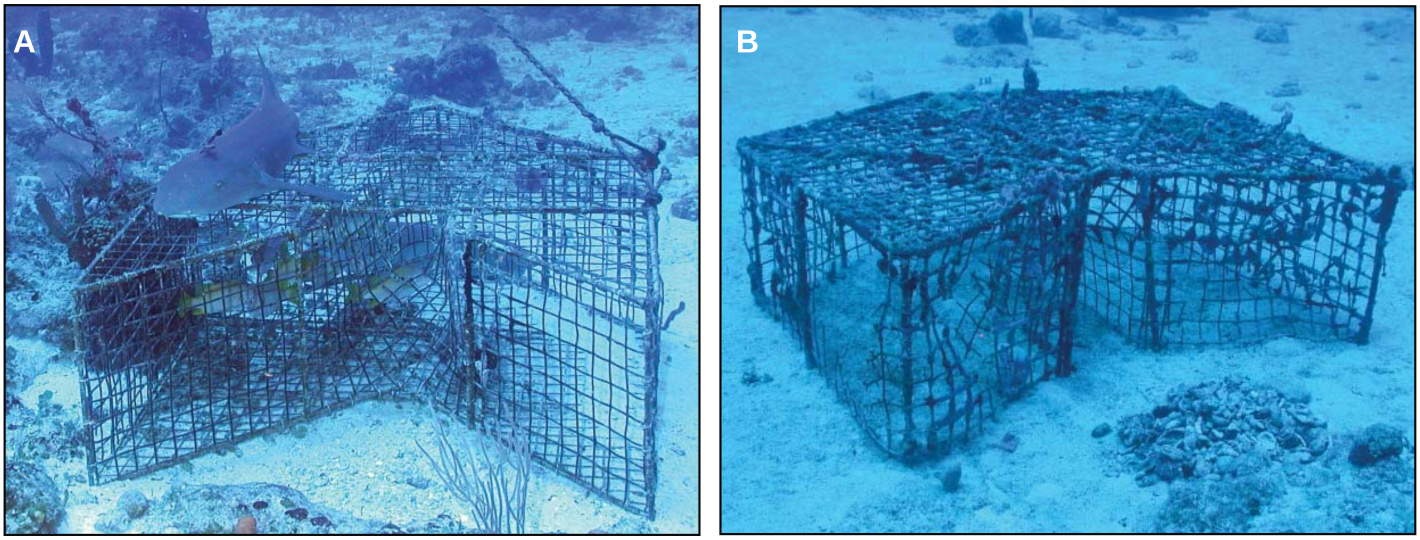

Figure 1.1. A) Actively fishing trap. Note the lines extending up to surface buoys or to other traps. B) Derelict fish trap. Photos: NOAA/NCCOS/CCMA Biogeography Branch.

Negative impacts, such as "ghost fishing" (a trap continuing to catch fish even after it has become derelict) and habitat damage, associated with derelict fish traps (DFTs) in coral reef environments have been examined (Chiappone et al., 2005; Uhrin et al., 2005; Marshak et al., 2008; Lewis et al., 2009). These impacts may impede habitat structure and function, including those designated as Essential Fish Habitat (EFH). Traps may move as much as several kilometers, by major storms and anomalous weather phenomenon (Olsen, pers. comm.), and up to tens of meters by strong wind events (Uhrin et al., 2005). Such movement can result in accumulations of large piles of traps. Derelict fishing gear has been known to impact biological resources as unattended gear may continue to cause unnecessary mortality (Chiappone et al., 2005; Matsuoka et al. 2005), but this has not been quantified in U.S. Virgin Island (USVI) reef ecosystems. Derelict gear also can cause navigation and safety issues (Macfadyen et al., 2009).

Traps are used widely throughout the Caribbean to catch finfish and crustaceans (Recksiek et al., 1991), and DFTs likely comprise a large portion of the submerged marine debris (Macfadyen et al., 2009). Traps are the predominant fishing gear used to capture fish and lobster in the territorial and federal waters of St. Thomas and St. John, USVI and support a strong commercial fishery (Agar et al., 2005; Figure 1.2).

[1] Center for Coastal Monitoring and Assessment, National Centers for Coastal Ocean Science, National Ocean Service, National Oceanic and Atmospheric Administration

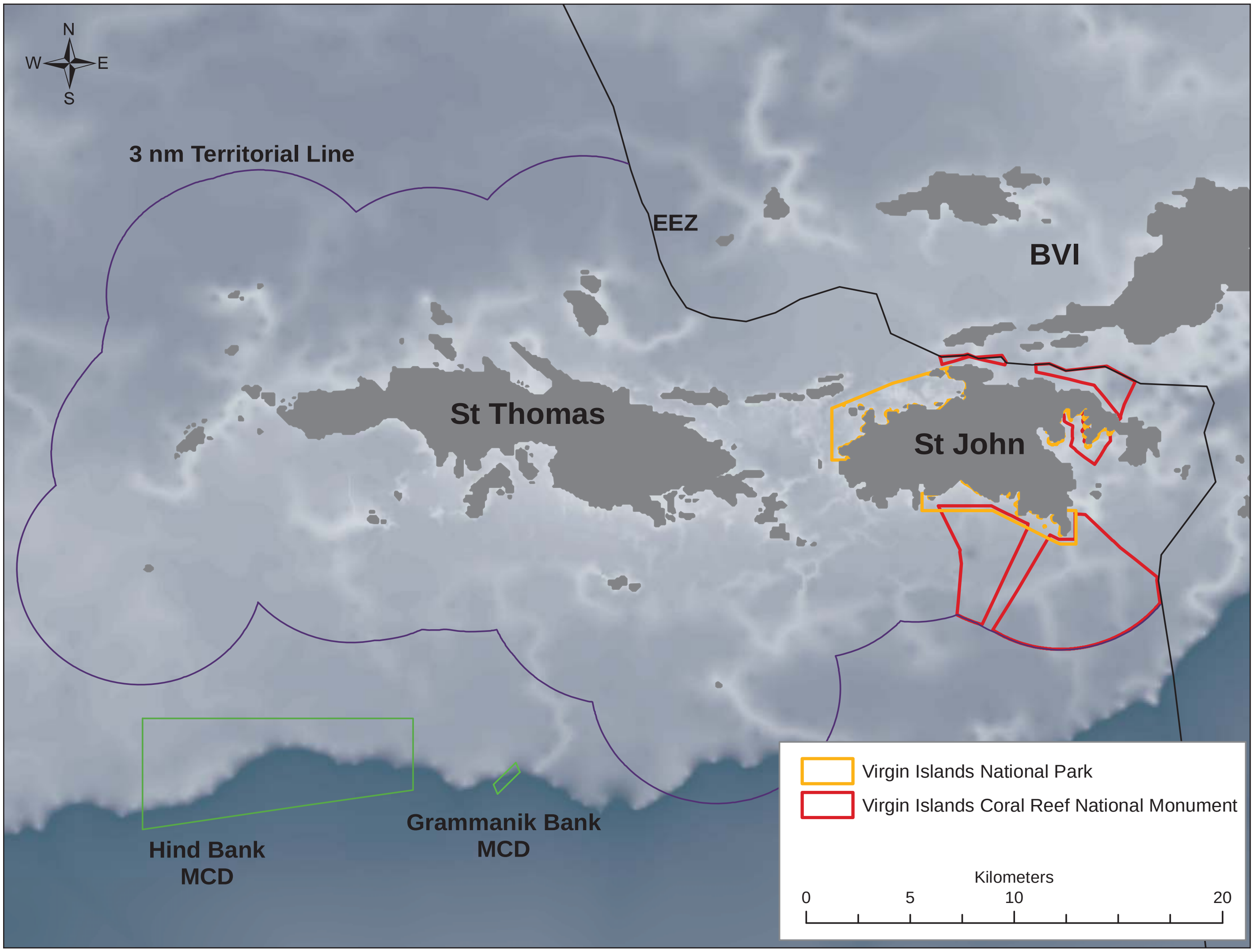

Figure 1.2. Federal and territorial waters and selected marine protected areas of St. Thomas and St. John, USVI.

Currently there is a general lack of knowledge about commercial/recreational fishing effort and associated derelict fish traps in the Caribbean, including the waters around St. Thomas and St. John, USVI. Prior to this study, no information existed regarding the abundance and distribution of derelict traps, the rate at which active traps become derelict, or areas that promote dereliction. Furthermore, little information existed on the fate of traps in the water, such as the impact on marine organisms and the colonization of derelict fish traps by fouling organisms and how their condition changes over time.

Image 1.1. Commercial fish species caught in traps in the U.S. Virgin Islands (USVI). Photos: St. Thomas Fishermen's Association (STFA).

This study provides the first effort to quantify fishing effort and trap loss in the waters of St. Thomas and St. John and subsequent economic impacts to the fishery (Image 1.1). The study also provides the first experimental assessment of derelict trap behavior by examining fish and invertebrate mortality, fouling communities on derelict traps, potential for trap movement, and the potential for derelict traps to integrate with the natural benthic community. Lastly, we tested the efficacy of using autonomous underwater vehicles (AUVs) to help locate derelict traps. In a controlled experiment, we deployed AUVs fitted with sidescan sonar and digital cameras to map and photograph the seafloor at specific locations where traps had been strategically placed and to explore locations where traps were thought to have been lost by fishermen in the past. AUVs had been used to detect objects of interest to the military in other marine systems with success, but had not previously been used to detect derelict traps in areas with complex benthic communities (i.e., coral reef ecosystems; Figure 1.3). This presented a new technical challenge and novel suite of environmental conditions for the equipment and operators.

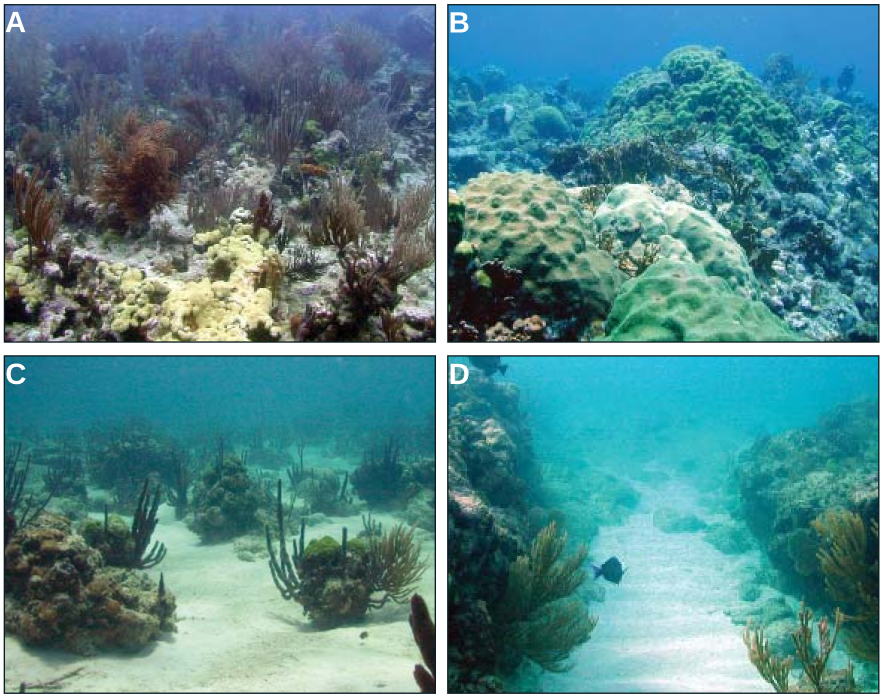

Figure 1.3. Examples of benthic habitats with complex reef communities: A) colonized pavement, B) linear reef, C) aggregate patch reefs and D) spur and groove reefs. Photos: NOAA/NCCOS/CCMA.

The impetus for this study emerged from the need for a greater scientific understanding of derelict fish traps and how they function within coral reef ecosystems. To address this need, a partnership was formed among the St. Thomas Fishermen's Association (STFA), NOAA's Marine Debris Division, NOAA's National Centers for Coastal Ocean Science (NCCOS) Center for Coastal Monitoring and Assessment (CCMA) Biogeography Branch, NOAA's National Marine Fisheries Service (NMFS), National Park Service (NPS), the University of the Virgin Islands (UVI), and the U.S. Navy's Naval Surface Warfare Center Panama City Division (NSWC PCD) with the following objectives:

1. Spatially quantify fishing effort and trap loss in the waters of St. Thomas and St. John.
2. Test the use of AUVs to detect traps in a complex coral reef ecosystem and to quantify DFT abundance and distribution around St. Thomas and St. John.
3. Examine derelict trap characteristics and impacts:
    a. Resource mortality
    b. Fouling communities
    c. Trap movement

1.2. REFERENCES

Agar, J.J., M. Shivlani, J.R. Waters, M. Valdés-Pizzini, T. Murray, J. Kirkley, and D. Suman. 2005. U.S. Caribbean Fish Trap Fishery Costs and Earnings Study. NOAA Technical Memorandum NMFS SEFSC 534. Miami, FL. 127 pp.

Chiappone, M, H. Dienes, D.W. Swanson, and S.L. Miller. 2005. Impacts of lost fishing gear on coral reef sessile invertebrates in the Florida Keys National Marine Sanctuary. Biological Conservation 121:221-230.

Hess, N.A., Ribic, C.A.and I. Vining. 1999. Benthic marine debris, with an emphasis on fishery-related items, surrounding Kodiak Island, Alaska, 1994-1996. Marine Pollution Bulletin 38:885-890.

Laist, D.W. 1995. Marine debris entanglement and ghost fishing: a cryptic and significant type of bycatch. In: Proceedings of the Solving Bycatch Workshop, September 25-27, 1995, Seattle, Washington, pp. 33-40.

Lewis, C.F., S.L. Slade, K.E. Maxwell, and T.R. Matthews. 2009. Lobster trap impact on coral reefs: effects of wind driven trap movement. New Zealand of Marine and Freshwater Research 43:271-282.

Macfadyen, G., T. Huntington, and R. Cappell. 2009. Abandoned, lost or otherwise discarded fishing gear. UNEP Regional Seas Reports and Studies, No. 185; FAO Fisheries and Aquaculture Technical Paper, No. 523. Rome, UNEP/FAO. 115 pp.

Marine Debris Research, Prevention, and Reduction Act. 2006. United States Statutes at Large, 109th Congress, 2nd Session. Volume 120, Stat. 3333, Public Law 109-449.

Marshak, A.R., R.L. Hill, P. Sheridan, M.T. Schärer, and R.S. Appeldoorn. 2008. In-Situ Observations of Antillean Fish Trap Contents in Southwest Puerto Rico: Relating Catch to Habitat and Damage Potential Proceedings of the Gulf and Caribbean Fisheries Institute 60:447-453.

Matsuoka, T., T. Nakashima, and N. Nagasawa. 2005. A review of ghost fishing: scientific approaches to evaluation and solutions. Fisheries Science 71(4):691-702.

Olsen, D. St. Thomas Fishermen's Association. St. Thomas, U.S. Virgin Islands. Personal communication.

Recksiek, C.W., R.S. Appeldoorn, and R.G. Turingan. 1991. Studies of fish traps as stock assessment devices on a shallow reef in south-western Puerto Rico. Fisheries Research. 10:177-197.

Uhrin, A.V., M.S. Fonseca, and G.P. DiDomenico. 2005. Effect of Caribbean spiny lobster traps on seagrass beds of the Florida Keys National Marine Sanctuary: Damage assessment and evaluation of recovery. American Fisheries Society Symposium 41:579-588.

Chapter 2: St. Thomas and St. John Trap Fishery: Present Status and Trap Loss

David Olsen[1] and Ronald L. Hill[2]

2.1. INTRODUCTION AND BACKGROUND

Fisheries of the U.S. Virgin Islands (USVI), as with most other Caribbean islands, tend to have sectors that can be categorized loosely as commercial, recreational, and subsistence. Typically, commercial fishing is an economic endeavor where landed species are sold for profit, and subsistence fishing provides food solely for the fisherman and his family. Recreational fishing, on the other hand, is a sport activity, including tournament fishing and charter boat or headboat operations, where catches may be kept or released at the discretion of the fisherman or because of fishing regulations. USVI fishing regulations currently exist almost exclusively for the management of commercial fishing and fishing gear.

In the U.S. Caribbean, fish and lobster traps are defined and regulated as commercial fishing gear (Figure 2.1). Any USVI fisherman using traps is required to possess a commercial fishing license and the traps must be inspected by the Department of Planning and Natural Resources (DPNR) and uniquely labeled prior to deployment. In practice, there are unlicensed subsistence fishermen using traps that are not properly regulated. These traps tend to be fished closer to shore and in shallower waters than most of the commercial fleet (R. Hill, unpubl. data). The catch from these traps are not quantified in the collection of fishery data (Swingle et al., 1970) and the impacts of this sector of the fishery remain unrknown.

Figure 2.1. Image of traps used by fishermen in the U.S. Virgin islands (USVI). Photo: St. Thomas Fishermen's Association (STFA).

2.2. PRESENT STATUS OF THE FISHERY

Trap fishing occurs in both territorial waters (within 5.6 km or 3 nautical miles [nm] of the coastline) and federal waters (5.6-370 km or 3-200 nm) of the USVI (see Chapter 1, Figure 1.1). There are directed fisheries for both Caribbean spiny lobster (*Panulirus argus*) and various reef fishes (Garrison et al., 1998; Sheridan et al., 2006). The USVI DPNR, Division of Fish and Wildlife (DFW) is responsible for managing fishing within territorial waters while the Caribbean Fisheries Management Council (CFMC), in conjunction with NOAA's National Marine Fisheries Service (NMFS), under the authority of the Secretary of Commerce, manage fisheries within federal waters. The two management entities collaborate and generally develop compatible regulations to manage the fishery.

[1] St. Thomas Fishermen's Association
[2] Southeast Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration

2.2.1. Trap Designs and Costs

Fish traps, also known locally as “pots,” have traditionally been the most widely used gear in the multi-gear, multi-species fisheries of the U.S. Caribbean (Fielder and Jarvis, 1932). Three reasons have been proposed for their wide popularity (Austin, 1988):

1. Traps are the most effective small boat gear in the strong trade winds that buffet the USVI because they can be left to fish for days and then hauled in the morning when it is often reasonably calm.
2. Other fishing techniques can be used to augment catches while the traps fish.
3. Traps effectively catch a wide diversity of reef fish that are not harvestable by other means.

According to Kojis and Quinn (2006) trap design has changed very little from the earliest records in the USVI, but the materials used to construct traps have changed.

> *“In the 1930’s, arrowhead traps made from mats of split vines woven into 2.5-5.0 cm hexagonal mesh and braced with a framework of wood were common (Fiedler and Jarvis, 1932). Because it lasted longer, St. Croix fishers favored making traps from 0.3 cm diameter marine cable, which fishers found discarded. In 1968, arrowhead traps were still the most widely used gear and the principal method of harvesting food (Swingle et al., 1970). The traps were using new materials such as plastic coated or galvanized welded mesh chicken wire. Occasionally, the frame was made of reinforced steel instead of wood and a zinc anode was added to prevent electrolysis. Mesh size ranged from 1.8 to 5.0 cm. Traps were individually buoyed. The importance of traps to the fishers persisted until at least 1981, when >80% of the fishermen use[d] only traps (Olsen and LaPlace, 1981)."*

Trap design today is more varied than in 1967. The traditional arrowhead, or chevron traps, are still popular but many fishermen build square or rectangular traps (Figure 2.2A) and sometimes Z or S shaped traps. Rectangular traps are more easily stacked on commercial fishing boats. Most traps are built of reinforced steel (i.e., "rebar") and covered with plastic or vinyl-coated galvanized mesh, though some fishermen, especially on St. Croix, still build the trap frame from wood. According to Title 12, VIRR [Virgin Island Rules and Regulations], "the minimum mesh size is 1.5 in (3.75 cm) hexagonal in St. Croix and 2 in^2 (5 cm^2) in the northern Virgin Islands (Kojis and Quinn, 2006).” Lobsters are sometimes caught in fish traps although dedicated lobster traps made of wood or plastic slats are the norm (Figure 2.2B).

Figure 2.2. A) Fisherman about to deploy a rectangular trap ; and B) a plastic lobster trap. Photos: NOAA/NCCOS/CCMA Biogeography Branch and STFA.

In addition to meeting mesh size requirements, traps must, by regulation, have escape panels that are tied shut with a biodegradable twine or “rot cord.” The twine is expected to rot within a reasonable length of time in order to open the panel on any traps that are lost to reduce ghost fishing. Degree of effectiveness of this measure is unknown, with limited studies on “time to opening” or level of compliance. Traps in the USVI are fished either singly, with each trap attached to a surface buoy, or in “strings” or “trawls” in which each trap is tied to the next in line, with surface buoys only at each end of the string (Figure 2.2B; Sheridan et al., 2006). Surface buoys must be identifiable as to owner, either through the use of a color coding system or engraving of owner numbers. In the USVI, fishermen tend to use polypropylene line between traps so that the line floats above the bottom reducing the possibility of entanglement with the benthos and providing a target for grappling, if needed to recover the trap string. Trap strings average 13 traps per string, ranging from 4-25 (Sheridan et al., 2006). In recent years there has been more of a tendency to reduce the use of surface buoys in order to reduce theft and interaction with surface vessels, the goal being to reduce economic loss. If traps are set blind, without surface buoys, fishermen use triangulation or, more recently, global positioning system (GPS) to locate the site and a grappling hook is pulled through the water to snag the floating line between traps in order to retrieve them (Kojis and Quinn, 2006; Sheridan et al., 2006).

In the present study participating members of the St. Thomas Fishermen's Association (STFA) provided information on trap design and construction. They confirm that commercial fishermen target either reef fishes, spiny lobster, or both and trap designs vary accordingly. The arrowhead and rectangular designs are most common for reef fishes. Construction of new traps is mainly correlated with an individual's loss rate from the prior year and traps are not commonly made every year. Most fishermen build their own fish traps. The common plastic lobster traps are usually purchased from Florida (D. Olsen, pers. comm.); however, some lobster traps are built by local fishermen.

Considerable variability was observed when considering costs to build traps. In general, lobster traps were cheaper to make than fish traps, but some fishermen spend well over $400 for construction of each trap. Some variability might be expected since individual fishermen construct traps of slightly different sizes and complexities. The traps built for this project (see Chapter 4) cost $200/trap. A lot of traps were constructed during 2008-2009 as significant ground swell occurred during 2007-2008 in the region and destroyed or moved many traps.

2.2.2. Number of Traps and Fishermen

In order to understand the magnitude of the potential problem of derelict traps, an estimate of the number of traps fished and lost each year is needed. A variety of sources are available to examine both the numbers of traps that are fished in USVI waters and the numbers of commercial fishermen who fish them. From a commercial fishermen census, Kojis and Quinn (2006) estimated that the number of traps fished for the entire USVI had increased from about 1,882 in 1930 to 3,296 in 1968 and 10,409 in 2003. In late 2001, Sheridan et al. (2006) reported an estimate from DPNR of 1,500 traps around St. Croix and 7,000 around St. Thomas. Their related analysis of commercial catch reports from 2000-2001 produced an estimate that 766 traps were reported fished around St. Croix and 4,087 traps were fished around St. Thomas/St. John, a total of 4,853 for the entire USVI. Their speculation was that total traps owned did not necessarily equal the number of traps in the water (fished). Regardless, the trap fishery is responsible for the majority of landings reported to the USVI territory (Figure 2.3).

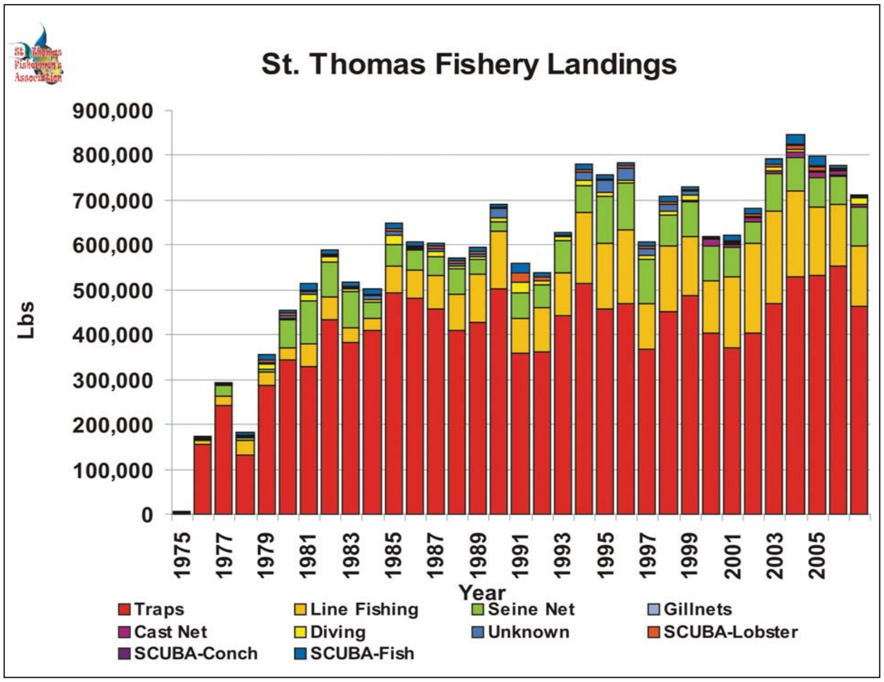

Figure 2.3. Fishery landings by gear type in St. Thomas, USVI, 1975-2007. Source: D. Olsen.

For this project, the STFA created a database to encapsulate various data regarding the commercial trap fishery (Table 2.1). Prior to this effort, STFA estimated that approximately 5,000 fish traps and 3,000 lobster traps were in operation during 2005-2009 by STFA members on the St. Thomas/St. John shelf. In 2011, the STFA conducted a thorough review of inventory establishing a current baseline of 3,899 fish traps and 2,632 lobster traps (D. Olsen, unpub. data).

Table 2.1. Results of St. Thomas Fishermen's Association (STFA) informational database, 2010.

Response	Average	Minimum	Maximum	Respondents
Boat Length	30.1	22	36	14
Fish Traps Fished Per Year	150.7	60	300	14
Lobster Traps Fished Per Year	213.9	0	605	14
Fish Traps Normally Made Per Year	3.2	0	30	14
Lobster Traps Normally Made Per Year	1.4	0	20	9
2008 Lost Traps Per Fisherman	15.8	0	40	14
2007 Lost Traps Per Fisherman	19.1	0	75	8
2006 Lost Traps Per Fisherman	4.9	0	20	3
2005 Lost Traps Per Fisherman	1.2	0	4	2
Fish Trap Cost	$271.07	$60.00	$600.00	14
Lobster Trap Cost	$144.09	$60.00	$400.00	14
Fish Traps Made Last Year	46.6	0	114	14
Lobster Traps Made Last Year	5.7	0	60	14

The precise number of fishermen, even those that are registered as commercial fishermen, has been difficult to assess over the years due to lack of reporting and variations in the definitions that have been used to distinguish full-time fishermen. Recent changes to the local fishing regulations requiring consistent catch reporting have improved agencies' abilities to account for this sector. Holt and Uwate (2004) analyzed available records from 1974 to 2003 and provided annual estimates for the numbers of licensed commercial fishermen across the three-decade time period. They estimated a mean of 231 (± 16.8 standard error, SE) for St. Thomas/St. John and they calculated that the current fishery had 171 commercial fishermen. During 2006-07, STFA (unpub. data), estimated approximately 160 licensed fishermen on St. Thomas, with only about 64 of them being classified as "active."

In the 2006-07 fishing year, STFA's 56 members made 83% of the fish trap hauls and 97% of the lobster trap hauls reported to the local government. Participants in the current study made 50% of the reported fish trap hauls and 90% of the reported lobster trap hauls so their information should provide a solid representation of conditions within the larger trap fishery of St. Thomas/St. John.

2.2.3. Spatial Distribution of Trap Fishing

As part of the conditions for obtaining a commercial license, commercial fishermen are required to report catch data on a monthly basis and assign catch to a spatial quadrant (Figure 2.4). Although reporting forms and reporting requirements have changed over the last three decades, an effort has been made to quantify the spatial distribution of fishing effort. Data forms, from as early as the 1970s, included indications of fishing locations. As a compromise between managers, who wanted to be able to analyze fishing distributions, and fishermen who wanted to maintain secrecy of their fishing locations, fairly large "statistical areas" were established within which fishermen should indicate their locations. As an example, the most recent maps (Figure 2.4) contain four statistical areas for St. Thomas, three for St. John, one for the British Virgin Islands (BVI), and six for St. Croix.

Sheridan et al. (2006) reported on two complementary attempts to assess spatial distribution of trap fishing around St. Thomas/St. John: analysis of submitted catch reports (from fishing year 2000-2001) and queries of fishermen by DPNR in late 2001. One complication, according to the authors, was that a fisherman could report fishing in more than one area for each allotted landing total. When this occurred, they divided the total number of traps fished evenly among the areas cited, a known inaccurate compromise. Although the modal number of areas fished by fishermen queried was two, seven fishermen reported operating in three or four areas, and one fisherman operated in six areas. Through the analysis (Sheridan et al., 2006), the commercial catch reports (CCR) and queries both identified St. Thomas southwest (TSW) as the most fished area (32.5% CCR, 38.1% queries). The CCR identified St. Thomas northwest (TNW; 29.8%), St. Thomas southeast (TSE; 15.2%) and St. John southeast (JSE; 10.5%) as the next most commonly fished areas (see Figure 2.4). These numbers varied slightly from the data of the randomly queried fishermen, which ranked St. Thomas southeast (TSE; 27.0%), St. John southeast (JSE; 11.6%) and St. John southwest (JSW; 10.5%) in decreasing abundance after TSW. Both sampling methods generally agreed that the fewest traps were placed in St. Thomas northeast (TNE; 4.1% CCR, 2.2% queries), BVI waters (2.5% CCR, 3.4% queries), and St. John north (JN; 0.2% CCR, 0.8% queries). It is unknown why few commercial traps are reported fished in TNE, but JN encompasses Virgin Islands National Park (VIIS) where commercial fishing is restricted and the available area between land and the international border of the BVI is limited.

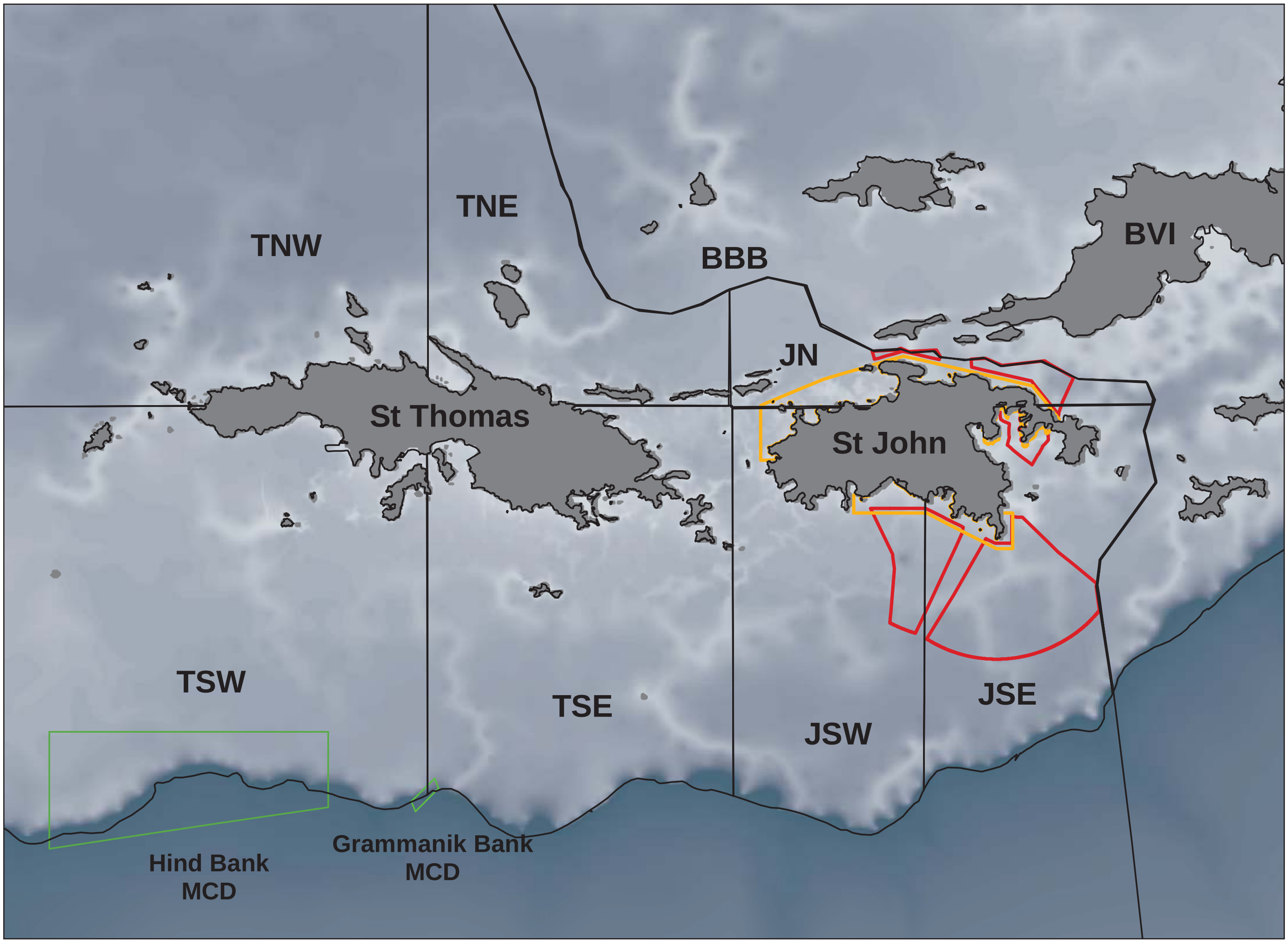

Figure 2.4. Federal and territorial catch reporting zones for St. Thomas and St. John, USVI. Four marine protected areas (MPAs) are displayed: Virgin Islands National Park (VIIS-orange line) and Virgin Islands Coral Reef National Monument (VICR-red line) in St. John and Red Hind and Grammanik Bank Marine Conservation Districts (MCD), green lines, south of St. Thomas. EEZ=Exclusive Economic Zone; BVI=British Virgin Islands.

In the field component of their study, Sheridan et al. (2005) and more recently Hill et al. (upub. ms.) report on efforts to develop more fine-scale spatial information on trap fishing effort, collected cooperatively by NMFS and DPNR in 2002, 2003, 2005 and 2006 (Figure 2.5). Boat-based surveys were conducted, primarily around St. Thomas, the western end of St. John in the northern Virgin Islands (VI), and around St. Croix (southern VI) to record GPS locations of surface buoys, attributing depth and habitat information to those locations where possible. A subset of traps, where depths were within recreational SCUBA limits, were further inspected by divers to assess detailed habitat information, trap contents, and any damage to habitat components attributable to trap fishing (Sheridan et al., 2005; Hill et al., unpub ms.).

As seen in Figure 2.5, trap buoys were found in all areas around St. Thomas, including JN (1.9% [annual average], 1.8% [standard deviation (SD)]), with surprisingly high numbers in TNE (24.5%, 16.4% SD) and JN, both of which were lightly reported in the CCR. An abundance of traps were located in TNW (25.9%, 20.5%SD) and TSW (22.3%, 23.8%SD; Hill et al., unpub. ms.), as reported in CCR or queries. Whether these trap placements are representative of certain fishing communities is currently unknown. Hull Bay, Red Hook, and Frenchtown (Figure 2.5) are the most frequently used commercial fishing harbors; Hull Bay is one of the few harbors on the north side of St. Thomas and is less populated than either Frenchtown or Red Hook. When compared to the dominant fishing center locations, the majority of trap buoys recorded were found closest to Hull Bay (Figure 2.6), while nearly a third of the buoys were close to Red Hook. This may indicate the origination of fishers, but more

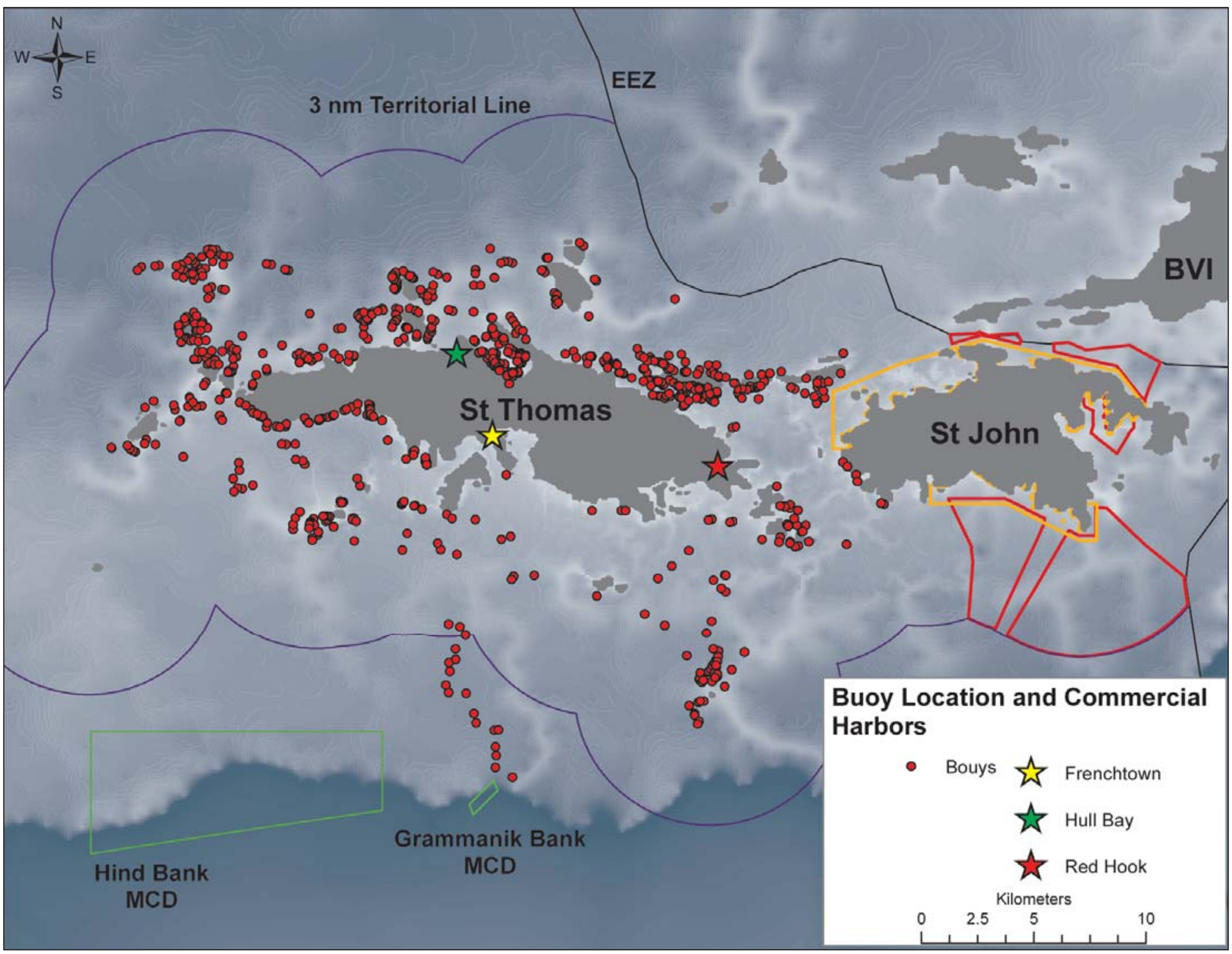

Figure 2.5. Position of trap surface buoys collected by USVI Department of Planning and Natural Resources (DPNR), 2002-2006.

research needs to be conducted to assess fishing patterns. Data from one of the authors (Hill, unpub. data) and information from DPNR could be used to confirm the origin of the fishermen in future analyses, although we are aware that traps in TNW are frequently placed by boats out of Red Hook (W. Ledee, pers. comm.).

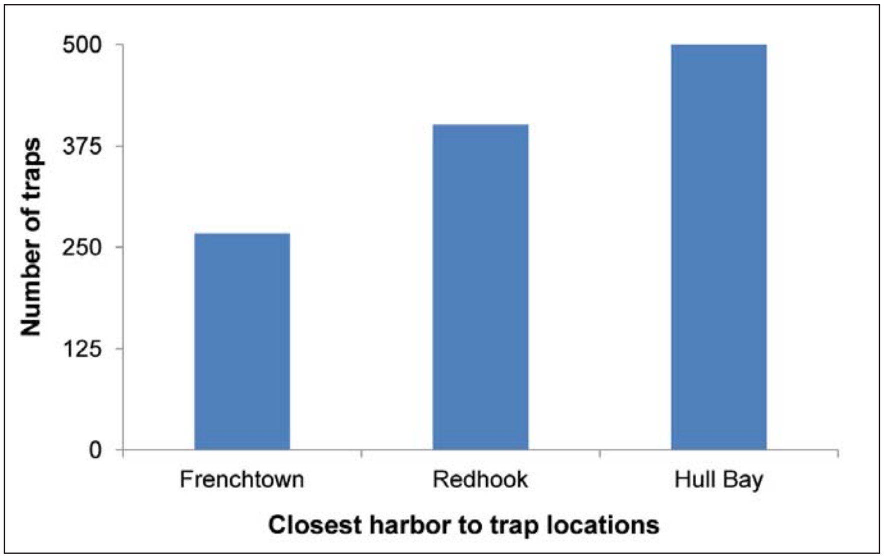

Figure 2.6. Closest harbor to trap position, 2002-2006.

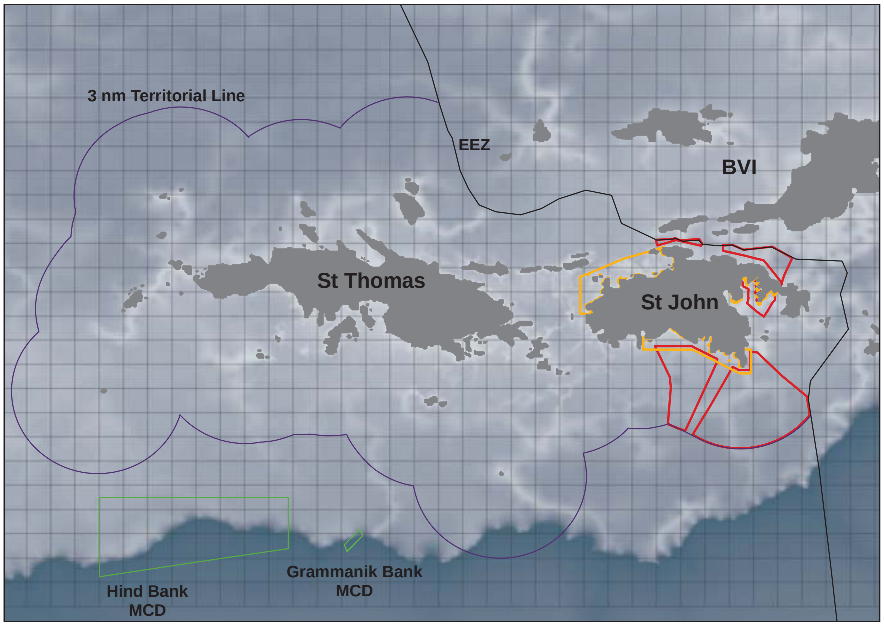

Figure 2.7. Sampling frame used by St. Thomas Fishermen's Association (STFA) members to quantify fishing effort in the region. Grid cells =1.5 km^2.

In the present study, fishing data from 2006-2009 provided from the most active (N=14) STFA members were used to populate a spatial database of fishing effort attributed to 1.5 x 1.5 km^2 grid cells (Figure 2.7). This grid cell resolution is significantly greater than the current system of statistical areas for catch reporting but not so great that it reveals specific fishing locations. Grids were generated from nearshore waters to the Exclusive Economic Zone (EEZ) boundary. Fishermen, primarily using personal logbooks, identified locations on charts where they commonly, or currently, set fish and lobster traps and detailed the numbers of traps they fished in that area. Data were entered into a GIS database. Numbers of traps were distributed across the grid cells that made up identified fishing locations. Total numbers of traps by grid cell were analyzed to generate color graduated displays of trap distributions (Figure 2.8a-c). STFA participants further identified specific locations (latitude/longitude) where they had lost traps in recent years.

Analysis of participating STFA fishermen's data indicated that their fishing locations occur, primarily, in 372 of the possible 1,700 grid cells (Figure 2.8a-c). The majority of fished grid cells (62%) are in the southern or southeastern portion of the region and the majority of effort is implemented in territorial waters. Most lobster and fish trapping is conducted in waters south of St. Thomas and St. John. irected lobster trapping is primarily occurring south of St. Thomas and St. John while highest numbers of fish traps are set predominantly south of St. Thomas. Effort in the north is limited for both fish and lobster traps and is concentrated at the shelf edge in water with depths of approximately 61 m (200 ft). The majority of fishing effort appears to be closest to Frenchtown and Red Hook Harbors (Figure 2.9), although the port of origination was not included in the database.

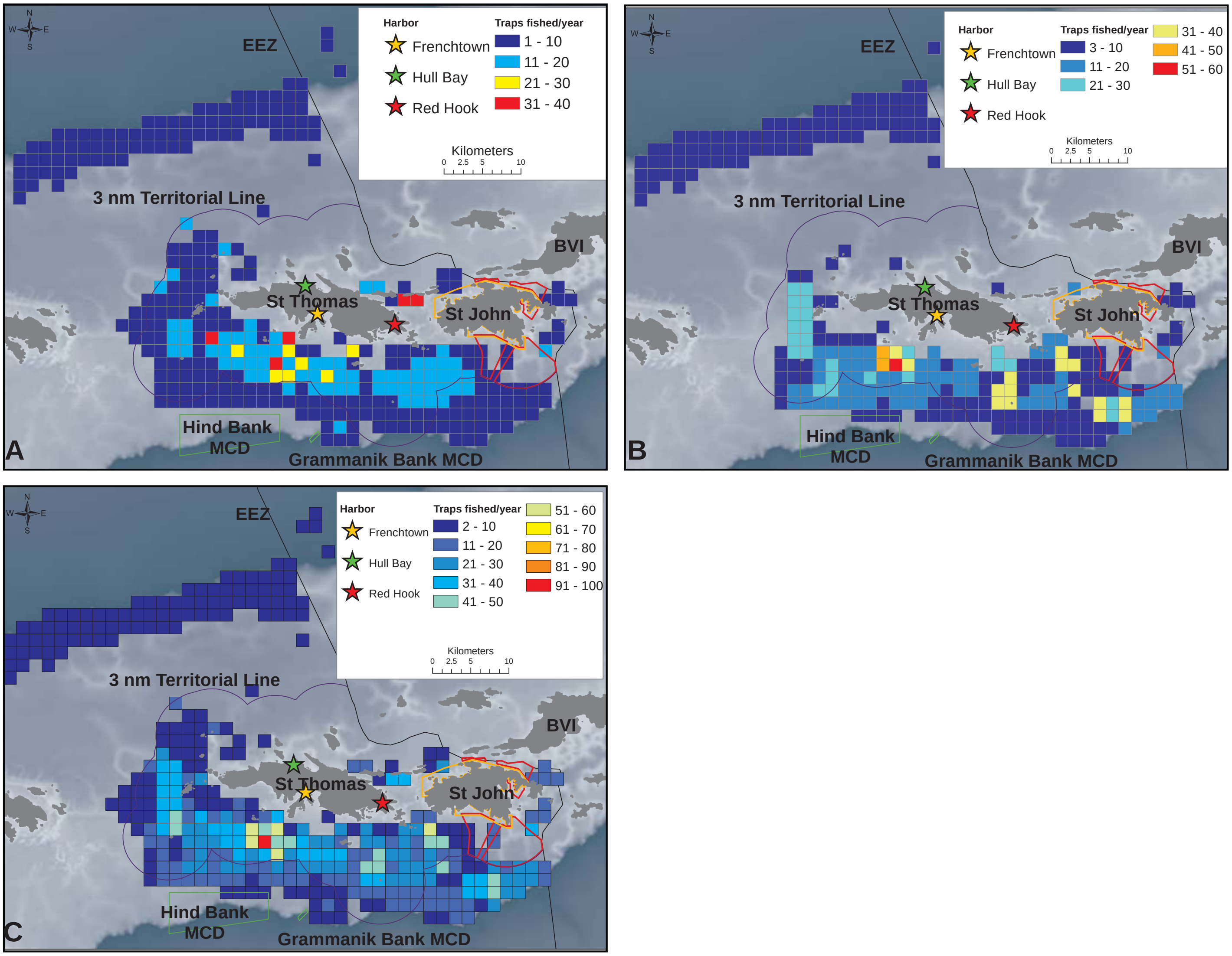

Figure 2.8. Fishing effort in federal and territorial waters of St. Thomas and St. John using A) fish traps, B) lobster traps and C) all traps.

Depth may be a contributing factor to both trap placement and trap loss (Lewis et al., 2009). In the queries reported by Sheridan et al. (2006), St. Thomas fishermen reported mean fishing depths of 47.5 m (range 18.3-183 m). In the boat based surveys (Hill et al., unpub. ms.), mean depth was calculated as 21.8 m, ranging from 2.1 to 50 m. The distribution shows 95% of the surveyed traps were located in waters less than 40 m deep, while 80% were found at depths of 30 m or less (Figure 2.10). Depths of diver surveys followed a similar distribution pattern shifted to shallower sites for diver safety. From the STFA data collected for this project, mean depth (m) was calculated for each grid cell with effort recorded; effort was predominantly targeted at depths 30-50 m but ranged from less than 10 m to

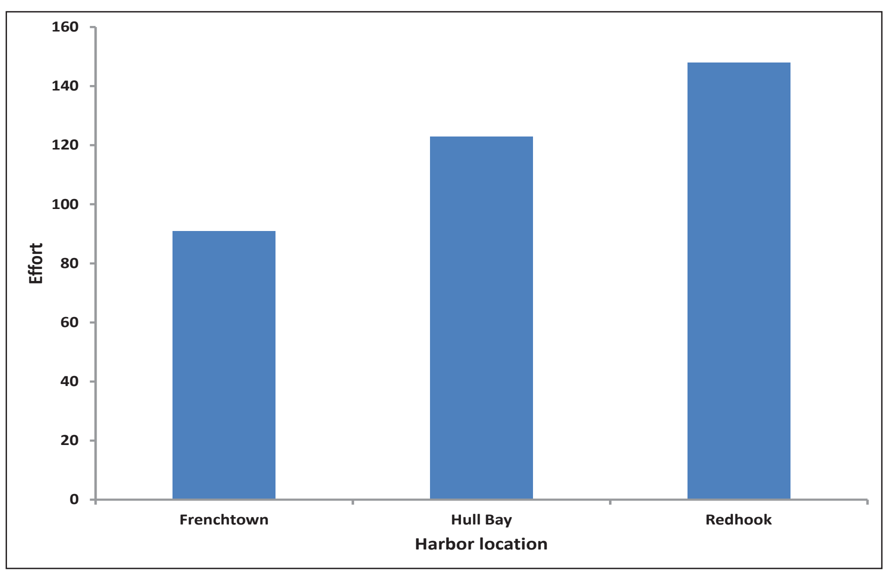

Figure 2.9. Closest harbor to STFA fishing effort grids.

greater than 100 m (Figure 2.11A). Both lobster and fish trap effort followed the same pattern (Figure 2.11B).

Almost all fishermen queried by DPNR (Sheridan et al., 2006) reported that they moved their traps on a seasonal basis. Shifts in weather patterns (tide/currents, ground swell, hurricane season, etc.) and changes in fish target species were the most often cited reasons traps were moved. Recent fishery research data may indicate that seasonal peaks of unmarketable species (parrotfish or boxfish) also play a role in trap relocations (D. Olsen, pers.comm.).

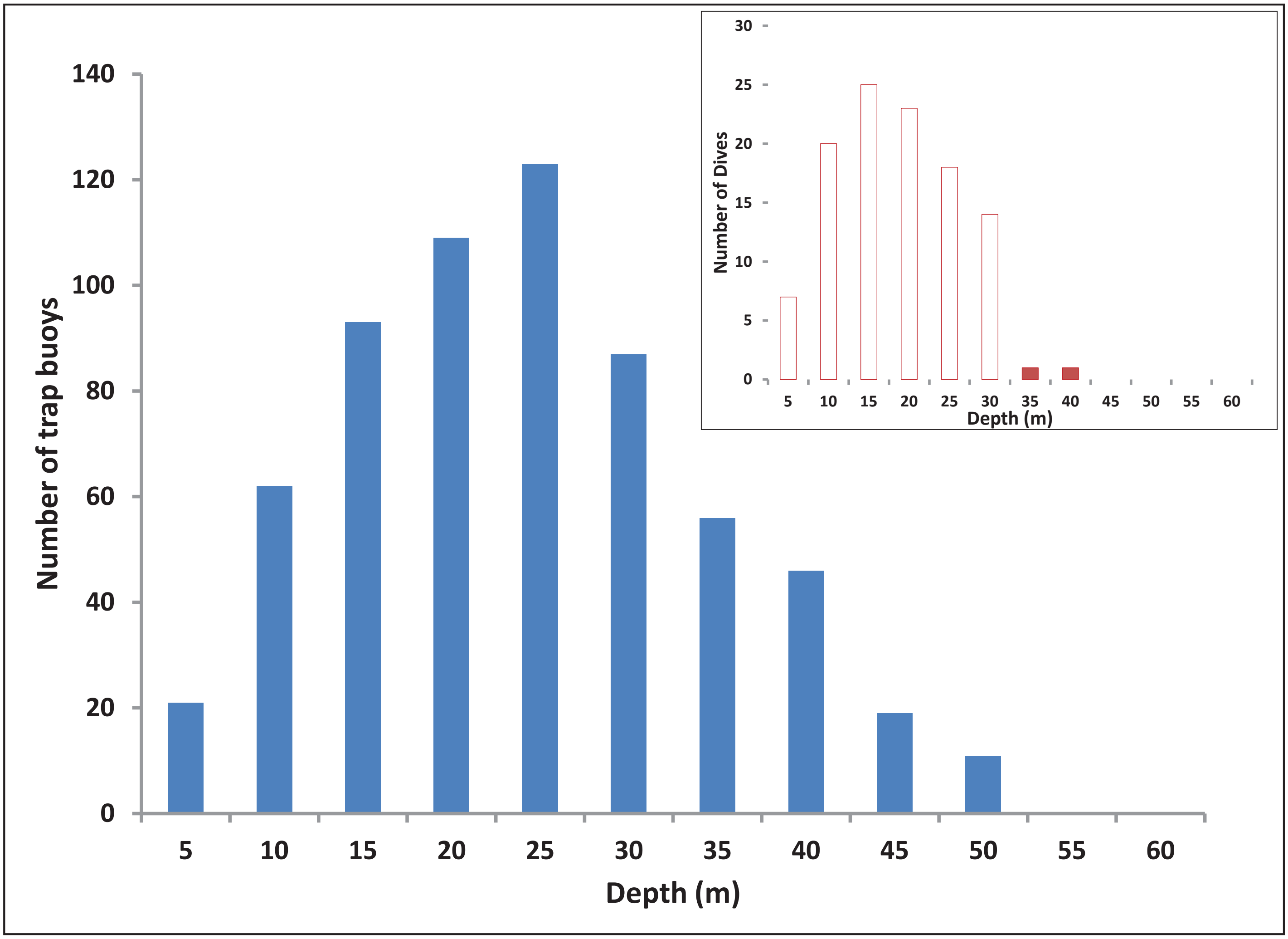

Figure 2.10. Depth of water under the trap surface buoys, 2002-2006. Inset: Depth of traps from diver surveys.

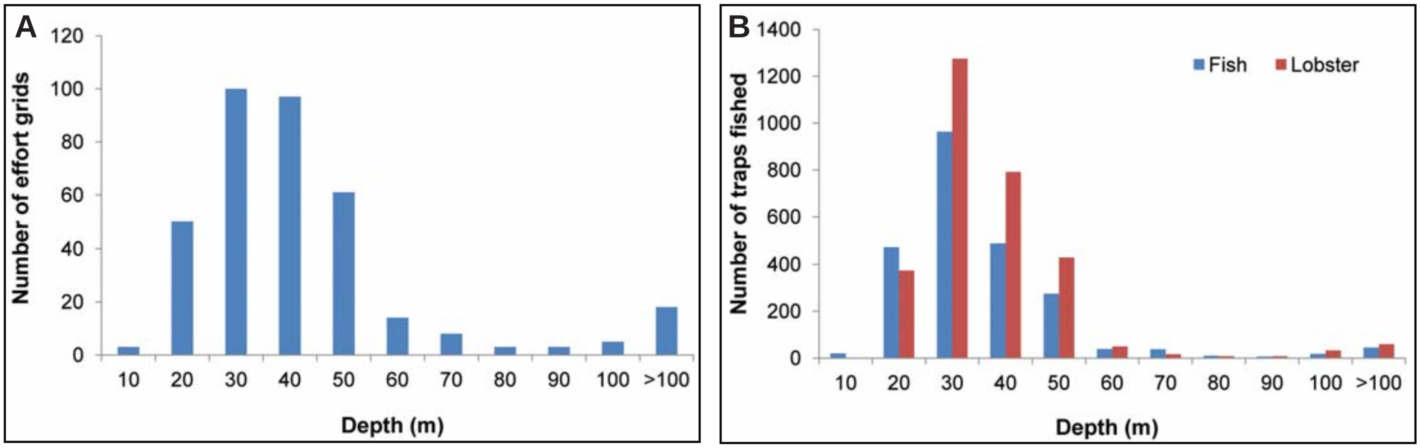

Figure 2.11. A) Mean depth (m) of fishing grids that were reported to be fished by STFA members. B) Number and depth (m) of fish and lobster traps deployed by STFA members.

Other than the recent studies of trap distributions (Sheridan et al., 2005; Hill et al., unpub. ms.), few research efforts have identified habitats targeted by trap fishers. Traps that were observed *in situ* by SCUBA surveys were limited to depths less than 38 m (122 ft) for safety reasons. These traps were fished in coral habitat (14%), sponge/ gorgonian hardbottom (29%), bare substrate (32%), seagrass (13%), or macroalgae (11%) (Sheridan et al., 2005). Approximately 43% of the traps observed were set on hardbottom or reef. Mapping of buoy positions on existing habitat maps suggests as much as 58% of traps may be placed on some type of hardbottom, either pavement/soft coral or reef habitats (Figure 2.12).

2.2.4. Trap Loss

In addition to quantifying fishing effort, STFA fishermen generated a database of locations where traps were lost, either to theft or weather (Table 2.1). These data were input as specific locations (latitude/longitude). Creation of the database highlighted the limitation of participants in remembering their activities. When queried about how many traps had been lost in years prior to 2008, most fishermen were unsure. This uncertainty might have been even more dramatic had the groundswell event in 2008-2009 been less severe.

Additional derelict trap location data came from dive shops, other NOAA data, National Park Service (NPS), and the University of the Virgin Islands (UVI). The complement of derelict traps provides an estimate of 604 traps in the region (Figure 2.13); the majority (N=514) were those reported lost by STFA members. There is a slight probability that traps

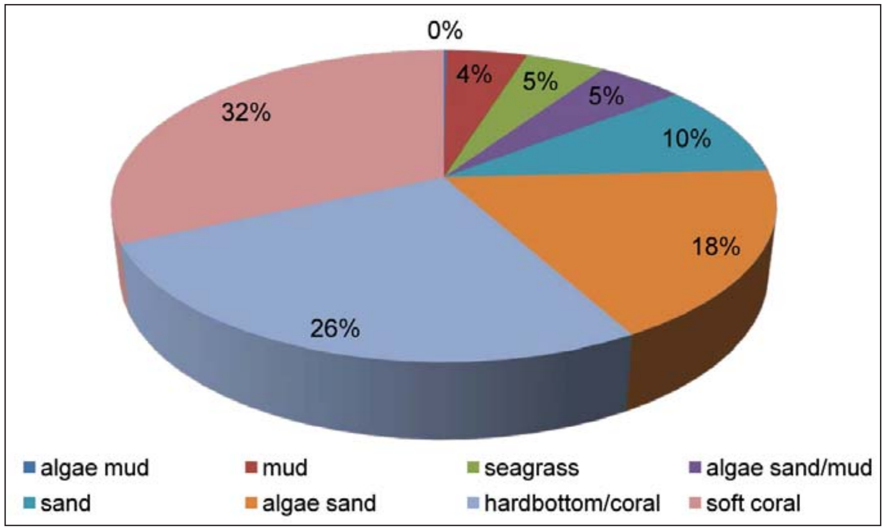

Figure 2.12. Habitat type at buoy locations based on benthic habitat maps, USVI DPNR, 2002-2006.

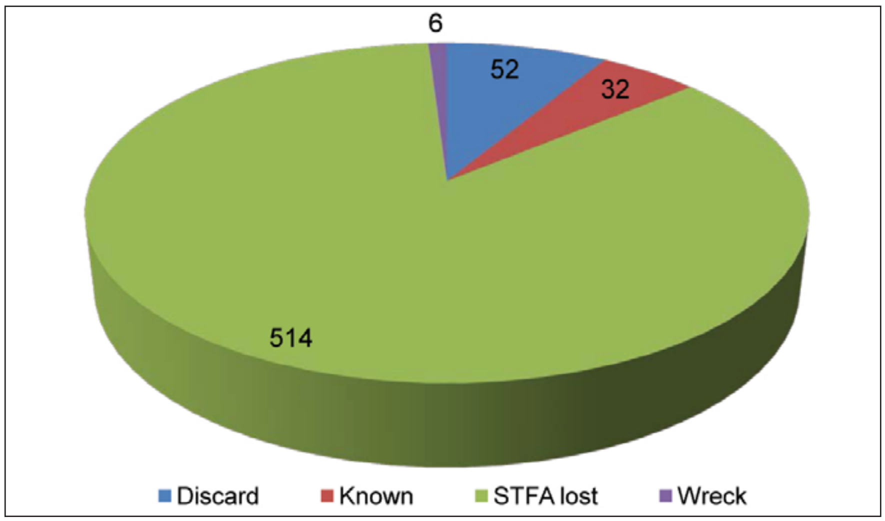

Figure 2.13. Current abundance of derelict traps as reported by STFA and other local entities.

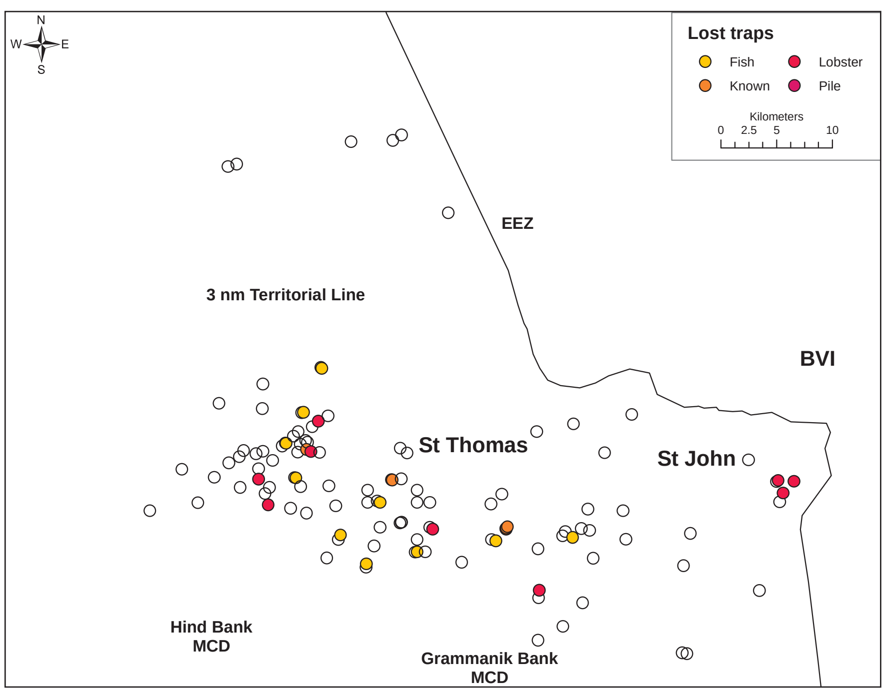

Figure 2.14. Spatial distribution of derelict traps in the study area.

reported by other non-STFA sources may in fact be traps also reported by STFA. Other sources documented 38 lost traps at various locations, including adjacent to wrecks that are common recreational dive sites. Intentionall discarded traps (all were fish traps) accounted for 52 or approximately 9% of the total derelict traps. These traps, while still debris, are not considered ghost fishing since they are typically discarded with escape panels open. The total number of derelict traps is likely underreported as there were eight trap piles (all west of St. Thomas), apparently where derelict traps accumulate, that contain an unknown, but large numbers of traps (Figure 2.14).

Overall, STFA trap loss was estimated to be about 10% per year (D. Olsen, pers. com). The majority of traps lost were fish traps (N=293); but comparable numbers of lobster traps were lost as well (N=221). Trap loss most commonly occurred for both types of traps at depths ranging between 20-40 m (Figure 2.15). Fish trap loss was generally related to the STFA fishing effort data, but the pattern was not very strong. In contrast, lobster trap loss was highly correlated with effort (Figure 2.16).

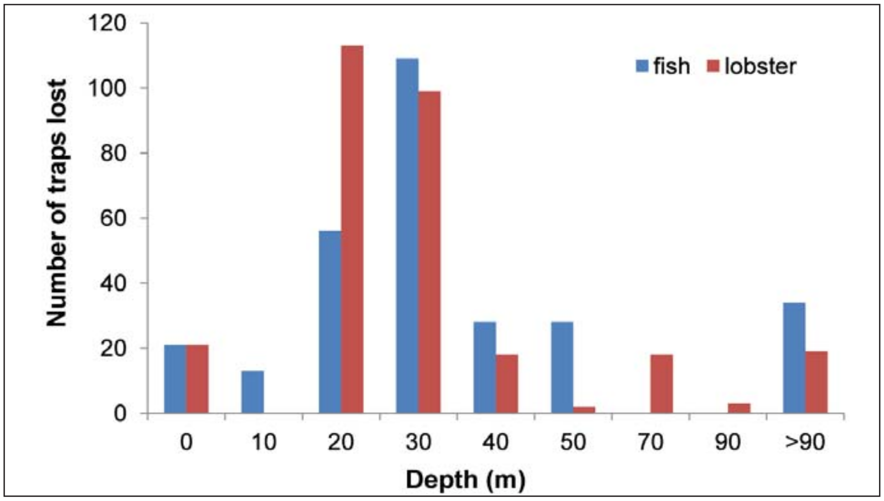

Figure 2.15. Depth of traps reported lost by STFA.

Trap loss has often been attributed to theft, as well as to interactions with surface vessel; cruise ships in particular (Sheridan et al., 2006; D. Olsen, pers. comm.). All the study participants noted that theft of both catch and traps was a problem.

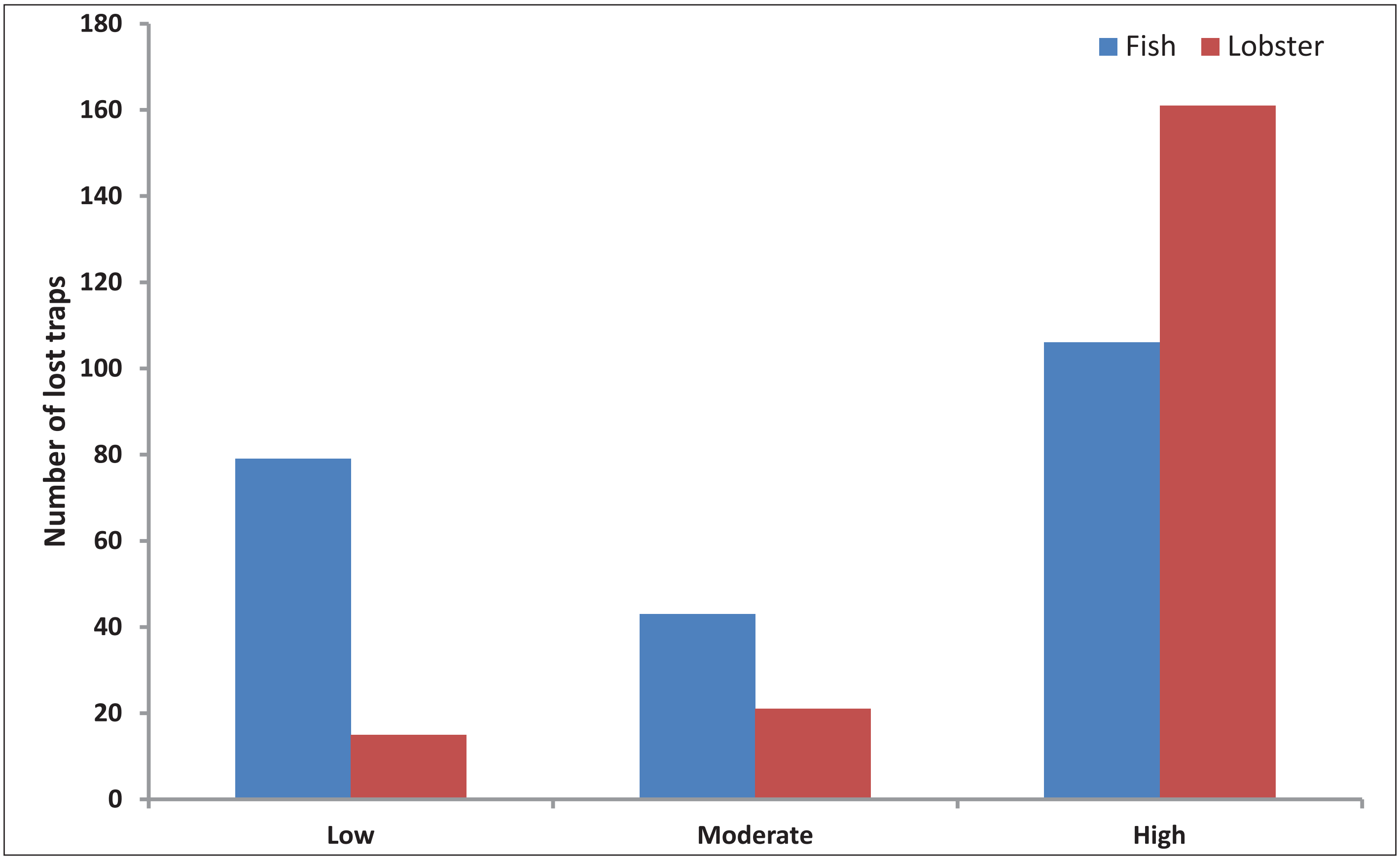

Figure 2.16. Correlation of STFA fishing effort and number of traps lost.

2.3. CONCLUSIONS

The effort information gathered here portrays two different fishing regimes. A deep water (greater than 40 m) commercial operation, primarily conducted by STFA, and a shallow water contingent (less than 40 m), presumably comprised of a host of different user groups (i.e., commercial and subsistence). These two depth based regimes should be investigated further in order to clarify fishing effort in the region and their association with derelict traps.

The increased resolution of fishing effort around St. Thomas was a huge advantage in investigating and quantifying the problem of derelict fish traps. Based on these data, a survey design using autonomous underwater vehicles (AUVs; See Chapter 3) was developed to quantify derelict fish traps (DFTs) in the region instead of surveying blindly. The relationship between commercial fishermen and scientific agencies is often tenuous; however this project serves as an overwhelming success and should serve as a model for future collaborations.

The number of traps lost per year is concerning; however, the rate of loss is lower than that observed in the Chesapeake Bay trap fishery (30%; Havens et al., 2008). Trap loss in St. Thomas/St. John is within the lower range (10-20%) of annual trap loss among the coral habitats in the Florida Keys (Lewis et al., 2009). Trap loss for the unknown shallow water portion of the fishery needs to be quantified. Additionally, it is alarming to know that 10% or more of trap loss around St. Thomas is attributed to fishermen discarding old traps. This practice needs to be investigated and an efficient alternative developed. Perhaps a means for dockside collection of these obsolete traps could be devised and implemented with assistance from local agencies or non-governmental organizations (NGOs). Trap theft is also an issue but is highly dependent on engaging local police and fisheries enforcement. In the past this has been voiced as a major concern for local fishermen.

Perhaps the greatest concern is the loss of traps due to interactions with vessel traffic. The information gathered here provides a “hot spot” portrayal of fishing effort and can serve as preliminary data for a marine spatial planning process in the region. Knowledge of these prime fishing locations could be used to define traffic corridors or alternative shipping/boating lanes to minimize use conflicts. This approach may be the most direct way to reduce trap loss and reduce the generation of derelict traps around the USVI, although it also requires increased enforcement or significant compliance and cooperation.

2.4. REFERENCES

Austin, C.B. 1988. Part One: Pot Fishing in the USVI. Report to Caribbean Fisheries Management Council and Division of Fish and Wildlife, Department of Planning and Natural Resources. University of Miami, FL.

Fiedler, R.H. and N.D. Jarvis. 1932. Fisheries of the Virgin Islands of the United States. U.S. Deptartment of Commerce, Bureau of Fish Investigations Report 14. 32 pp.

Garrison, V.H., C.S. Rogers, and J. Beets. 1998. Of reef fishes, overfishing, and *in situ* observations of fish traps in St. John, U.S. Virgin Islands. Revista de Biologia Tropical 46(5):41-59.

Havens, K.J., D.M. Bilkovic, D. Stanhope, K. Angstadt, and C. Hershner. 2008. The effects of derelict blue crab traps on marine organisms in the lower York River, Virginia. North American Journal of Fisheries Management 28:1194-1200.

Holt, M. and K.R. Uwate. 2004. Estimates of the number of licensed commercial fishers per year in the U.S. Virgin Islands, 1974/75 to 2003/04. Division of Fish and Wildlife, U.S. Virgin Islands Department of Planning and Natural Resources. 20 pp.

Kojis, B.L. and N.J. Quinn. 2006. A census of US Virgin Islands commercial fishers at the start of the 21st century. pp. 1326-1334. In: Y. Suzuki, T. Nakamori, M. Hidaka, H. Kayanne, B.E. Casareto, K. Nadaoka, H. Yamano, and M. Tsuchiya (eds.). Proceedings of the 10th International Coral Reef Symposium. Okinawa, Japan. 1950 pp.

Ledee, W. St. Thomas fisherman, Caribbean Fisheries Managment Council member. St. Thomas, U.S. Virgin Islands. Personal communication.

Lewis, C.F., S.L. Slade, K.E. Maxwell, and T.R. Matthews. 2009. Lobster trap impact on coral reefs: effects of wind driven trap movement. New Zealand of Marine and Freshwater Research 43:271-282.

Olsen, D. St. Thomas Fishermen's Association. St. Thomas, U.S. Virgin Islands. Personal communication.

Olsen D.A. and J.A. LaPlace. 1981. Demonstration of advances in Virgin Islands small boat fishing techniques. Marine Fisheries Review 43(11):11-15.

Sheridan, P., R. Hill, G. Matthews, R. Appeldoorn, B. Kojis, and T. Matthews. 2005. Does Trap Fishing Impact Coral Reef Ecosystems? An Update. Proceedings of the Gulf and Caribbean Fisheries Institute 56:511-519.

Sheridan, P., R. Hill and B. Kojis. 2006. Trap fishing in the U.S. Virgin Islands: how and where effort is exerted. Proceedings of the Gulf and Caribbean Fisheries Institute 57:175-188.

Swingle, W.E., A.E. Dammann and J.A. Yntema. 1970. Survey of the Commercial Fishery of the Virgin Islands of the United States. Proceedings of the Gulf and Caribbean Fisheries Institute 22:110-121.

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Chapter 3: Detecting and Mapping the Distribution of Derelict Traps

Timothy A. Battista[1], Randy Clark[1] and Peter Murphy[2]

3.1. INTRODUCTION AND BACKGROUND

Autonomous underwater vehicles (AUVs) equipped with side-scan sonar have been used to successfully locate and identify derelict fish trap (DFT) abundance and distribution (Havens et al., 2008; NRC, 2006; Figure 3.1). These efforts have been conducted in relatively homogeneous habitats consisting of low rugosity and soft bottom sediments. No attempts have been made to evaluate DFT detection efficiency of side-scan sonar in coral reef ecosystems that have a variety of different habitats (seagrass, sand/mud, reefs) and a considerable range of rugosity.

Figure 3.1. Autonomous underwater vehicle (AUV). Credit: Kongsberg Maritime.

Fishing effort information provided by St. Thomas Fishermen's Association (STFA) and the U.S. Virgin Islands' (USVI) Department of Planning and Natural Resources (DPNR) provided information that we could use to focus AUV surveys to assess the abundance and distribution of DFTs in the region. To do so, the project team collaborated with the U.S. Navy's Naval Surface Warfare Center (NSWC), Panama City Division, to conduct a small scale assessment of the efficiency of side-scan sonar to detect DFTs in complex habitats and implement surveys to identify the distribution and abundance of DFTs in select areas on the St. Thomas and St. John shelf. Here we present the results of the controlled efficiency test and initial findings of AUV DFT surveys.

3.2. METHODS

3.2.1. AUV Specifications

Two Hydroid (Kongsberg Company) Remus 100 Autonomous Underwater Vehicles owned by the U.S. Navy's NSWC, Panama City, FL were used to conduct operations. The vehicles were outfitted with standard sensor and payload as well as some additional features (Table 3.1). These vehicles are rated to 100 m water depth, have a diameter of 19 cm, length of 160-170 cm (depending on modules utilized) and weight in air of approximately 37 kg. While the vehicles' published endurance is 10 hrs at 2.3 m/s survey speed, we experienced typical mission endurance of approximately 5 hrs at 1.5 m/s. The endurance is primarily a function of operating environment (tidal or oceanographic current) and battery charge, which degrades over their lifespan (typically three years). In addition, one vehicle utilized a customized digital camera that recorded vehicle position, altitude, bearing, and depth for each picture frame in a text file, and

Table 3.1. Common hardware and software components contained on the Remus 100 autonomous underwater vehicle (AUV).

Components
Teledyne RDI 1200kHz ADCP with up/down transducers (SN242)
Teledyne RDI 600kHz DVL with bottom facing transducers (SN263)
YSI CT model 600 XL sensor
Marine Sonic Technologies Limited dual frequency (900/1800kHz) sidescan sonar
WHOI Micro-modem AComms system
Kearfott T16 Inertial Navigation System
Iridium communication capabilities (vehicle to Iridium base stations)
Wet Labs BB2F Scattering and Fluorometer ECO Sensor-470, 650, chlorophyll
CADCAC (computer aided detection computer aided classification) capable

[1] Center for Coastal Monitoring and Assessment, National Centers for Coastal Ocean Science, National Ocean Service, National Oceanic and Atmospheric Administration
[2] Marine Debris Program, Office of Response and Restoration, National Ocean Service, National Oceanic and Atmospheric Administration

each frame file name was time and event stamped. Vehicle performance was monitored real-time using the Hydroid RECON Software (Remote Control Capability), where the vehicle can be queried for its status and location using a REMUS Ranger and acoustic transponder. On the surface, communication with the vehicles was conducted using WiFi.

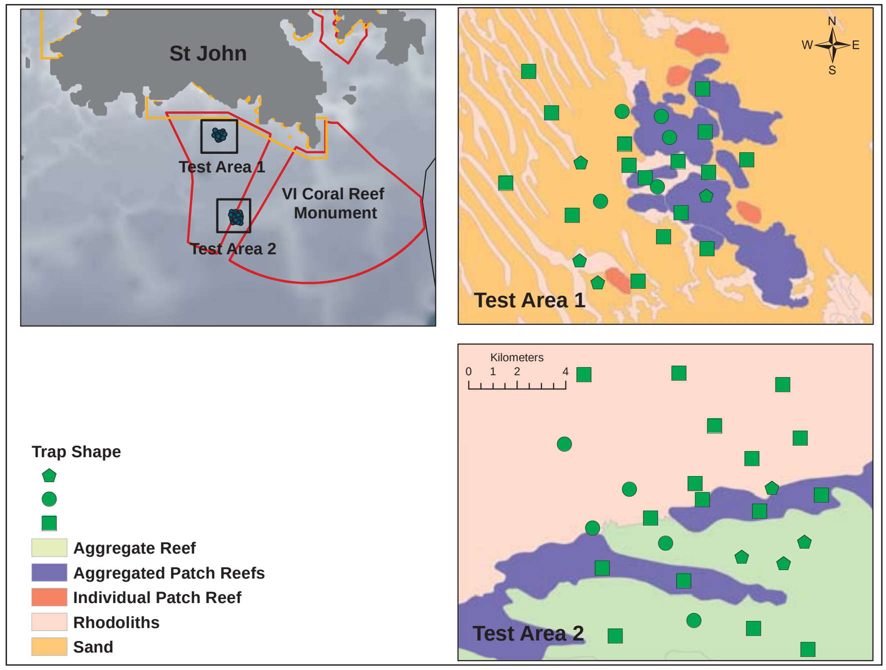

Figure 3.2. Location of test areas and trap placement among benthic habitat types on the south shore of St. John.

Survey planning was conducted using the Vehicle Interface Program (VIP) to define boundary extents, survey line spacing, vehicle altitude off the seafloor, survey speed, survey direction, sidescan frequency and range settings. Two test areas were selected to calculate the probability of trap detection in controlled locations (Figure 3.2). Additionally, survey areas outside the test areas were selected based on seafloor slope, prevailing wind and wave conditions, and STFA and DNR fishing effort (Figure 3.3).

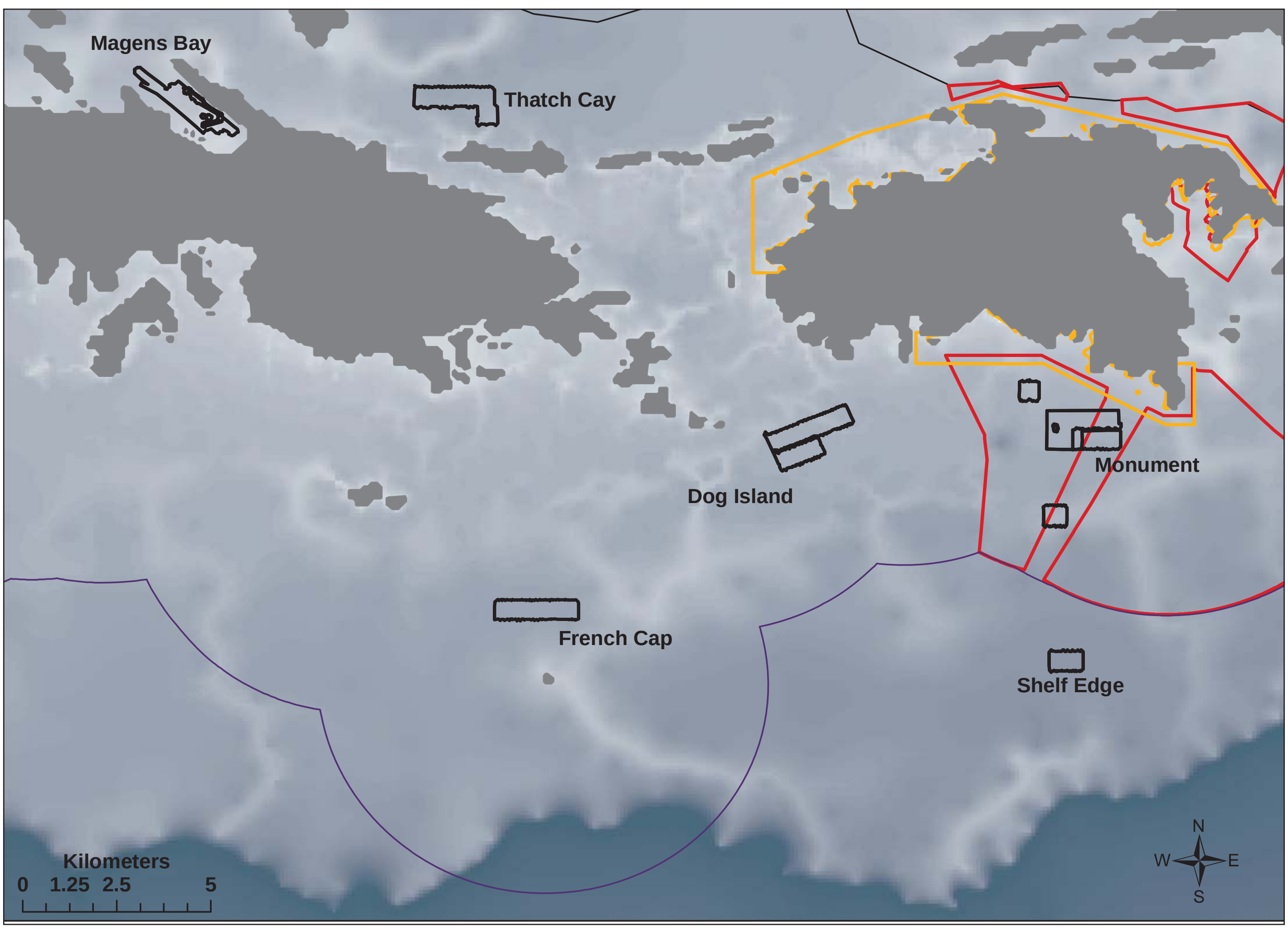

Figure 3.3. Outer boundaries of AUV tracklines for test areas and derelict trap surveys during October 2011.

3.2.2. Efficiency Test

Test areas were chosen as representative of typical habitats in the region and traps were deployed in a controlled experiment to assess target detection capabilities. These areas were previously mapped by the Biogeographic Branch in 2009 to provide highly accurate, detailed seafloor maps that could be used to select trap placement areas (see Kendall et al. [2001] for methods and definitions of habitat types). Three variables were considered important to quantify prior to conducting surveys. These were: 1) ability to detect traps over differing habitat structure and topographic complexity; 2) ability to detect and identify different trap types commonly used in the U.S. Caribbean (rectangular, arrow/chevron, and lobster traps); and 3) test various survey acquisition parameters (vehicle height off the bottom, range scale, survey direction, slant angle).

Trap doors were tied open and lines and floats attached. Typically, derelict traps do not have lines with buoys attached, but we didn't want to contribute our experimental traps to the derelict population. Deployed traps were of various sizes: four chevron (1.2 x 1.0 x 0.5 m), 16 square (1.25 x 1.15 x 0.5 m) and five rectangular lobster traps (0.83 x 0.63 x 0.45 m), were deployed in Areas 1 and 2 and exact global position system (GPS) coordinates were recorded. Traps were manually deployed off a STFA vessel with a ¼" twisted polypropylene line and float attached for retrieval. In advance of the surveys, the AUV assessment team was provided the bounding coordinates of the areas, but no further information regarding the number of traps, types of traps, locations, or bottom composition. The AUV surveys were conducted for Area 1 on October 4, 2010 and Area 2 October 10, 2010. Area 1 is completely contained within the St. John Virgin Islands Coral Reef National Monument (VICR) and Area 2 partially contained. Both test areas represent typical habitats used by fishermen and are in close proximity to commonly fished areas (see Figure 2.5.). Area 1 was selected to encompass areas including low habitat and topographic complexity (sand and rhodoliths), and moderate to high habitat and topographic complexity (aggregate and individual patch reefs). Rhodoliths are crustose benthic marine red algae of various sizes that can from extensive aggregations (Foster, 2001). Area 2 was selected to encompass more homogeneous areas, including rhodoliths and low to moderate habitat and topographic complexity (aggregate patch reef and aggregate reef). Using pot haulers, STFA fishermen moved traps from Area 1 to Area 2 and retrieved the traps after the survey.

The AUV's were transported to the sites and hand deployed over the side of the vessel. The National Park Service (NPS) 25 ft Whaler (*R/V Haulover*) was used for mission support (Figure 3.4). Upon deployment of the vehicles, WiFi signal was established and command relayed to commence operations (Figure 3.5). Once submerged, communication, vehicle status, and vehicle range was integrated using the Hydroid Ranger and Transponder. Upon deployment, the AUV will determine its position relative to the survey extent, navigate to the appropriate start point,

*Figure 3.4. National Park Service (NPS) Virgin Islands Coral Reef National Monument (VICR) research vessel (*R/V*)* Haulover*. Photo: NOAA/NCCOS/CCMA Biogeography Branch.*

and begin survey until successful completion. The vehicle will surface for retrieval at the completion of the project area, intermittently as needed to acquire GPS signal to improve its navigation ability, if the mission is aborted by the operator, or if the vehicle experiences mechanical or technical errors. The operator can monitor vehicle warnings through VIP and integration with the Ranger. If the system status indicates a fault (e.g., water intrusion, unintended contact with the bottom, navigation issues) the operator can send a command to abort the mission for vehicle retrieval.

Figure 3.5. Tracking equipment and computer set up for AUV deployment. Photo: NOAA/NCCOS/CCMA Biogeography Branch.

Upon completion of a daily mission, vehicles were transported to the base camp for battery charging, vehicle maintenance, and data downloading. These activities are conducted using the Hydroid Power/ Data Interface Module providing a high speed data download, and recharging of the Lithium-ion batteries.

Side-scan survey swath widths were qualitatively compared to examine the optimal range to detect traps. The AUV side-scan could operate using swath widths of 30, 40 and 50 m. The first AUV test survey was conducted using a 50 m swath, followed by 40 m and 30 m. It was decided that a 50 m swath width was notably more difficult to identify targets than at 40 or 30 m. This could potentially be due to reduced acoustic intensity as a function of increased slant range and reduced across-track resolution with increased slant range. There was little difference observed between the 30 m and 40 m swath widths, so in order to increase survey efficiency we conducted the remaining test surveys and real surveys using the 40 m swath.

Sidescan imagery was analyzed by observers who did not know the shapes, numbers and locations of traps deployed. The observers visually interpreted the imagery with the task of identifying features, or targets they thought were traps or objects of interest. If a target was classified as a trap, interpreters would rank their classification on a scale of 1-4 based on their confidence of the signal in the imagery. Targets ranked 4 were highly confident and ranks of 1 were low in confidence. Items of interest were deemed not a trap and no confidence ranking was assigned.

Test area identifications were evaluated by comparing the number of positive identifications and by also comparing the relationship of confidence scores with relation to habitat types, habitat slope and depth.

3.2.3. Derelict Fish Trap (DFT) Surveys

Survey areas were selected based on STFA or DPNR fishing effort information, specific trap loss locations provided by STFA, or protected areas that allowed surveys to be conducted when weather was bad (see Chapter 2, Figure 2.5). Survey areas were designed as 1 km^2 boxes, where both AUVs were deployed simultaneously (Figure 3.6). Upon completion, AUVs were retrieved and brought back to base camp as previously mentioned. Data were downloaded and imagery processed using Chesapeake Technologies Sonar Wiz to visually identify targets. Specific GPS coordinates, depth, target height and width, sidescan snapshot, and confidence were recorded for each target in a survey target report. During inclement weather conditions, AUV surveys were conducted in protected waters where fishing effort was assumed low.

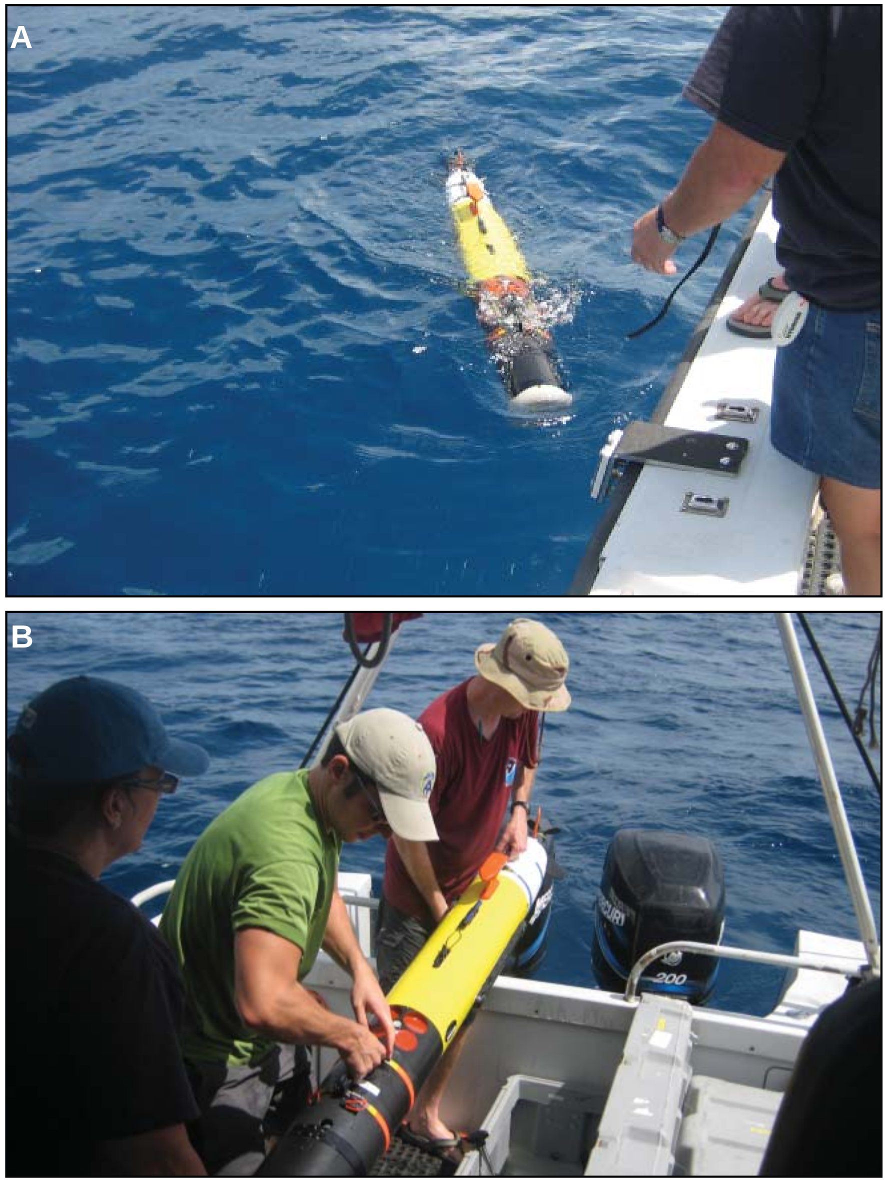

Figure 3.6. A) AUV deployment and B) AUV retrieval from deployment. Photos: NOAA/NCCOS/CCMA Biogeography Branch.

Initially we had intended to conduct verification of every target using the AUV digital camera, but were not given permission to use it beyond the test phase. To accomplish target verification, a remotely operated vehicle was deployed on a separate mapping mission in April 2011. A Phantom II remotely operated vehicle (ROV) operated by NOAA's Undersea Research Center (University of Wilmington at North Carolina) was deployed from the NOAA ship *R/V Nancy Foster* to provide video and digital still verification of potential trap targets. Given the navigational depth constraints of the NOAA ship *R/V Nancy Foster*, the location and depths of targets in Megan's Bay and Thatch Cay precluded target verification with the ROV. All other AUV survey sites were verified by the ROV mission. Using the ROV, we were able to navigate to the coordinates we had accumulated during the AUV surveys. Due to time constraints, we limited our verification to targets that were classified as traps, unless there was a cluster of targets close together that included traps and objects of interest. Verified targets were recorded on video and subsequently classified as one of three choices: not a trap, a DFT or an actively fishing trap. We classified targets as actively fishing by re-examining the side-scan imagery and if we felt confident that the target in the imagery was a trap but not found by the ROV, then the target was classified as an actively fishing trap.

3.3. RESULTS

3.3.1. Test Area Survey

Overall, 50 traps were randomly placed in two 0.5 km x 0.5 km test areas, wherein 37 traps were detected by visual interpretation of sidescan imagery (Table 3.2). Trap detection on sand and rhodolith habitats was highly successful, 100% and 89%, respectively. Overall, traps placed on aggregate patch reef or reef habitats had lower detection rates: 48% on patch reefs and 38% on aggregate reefs. Trap size or type was not a factor in the probability of detection; however, trap location in reference to habitat type was significantly correlated to detection (Figure 3.7). Chevron traps were detected 100% of the time on sand and rhodoliths, but not detected at all on reef habitats. When chevron traps were

Table 3.2. Trap type and number deployed by habitat type. Pooled numbers from both test areas.

Trap type	Overall traps detected/ traps deployed	Number of traps detected/traps deployed by habitat			
		Sand	Rhodoliths	Aggregated Patch reef	Aggregated Reef
chevron	4/8	2/2	2/2	0/1	0/3
lobster	8/10	2/2	4/4	1/2	1/2
square	25/32	9/9	10/12	4/8	2/3
Total	37/50	13/13	16/18	5/11	3/8

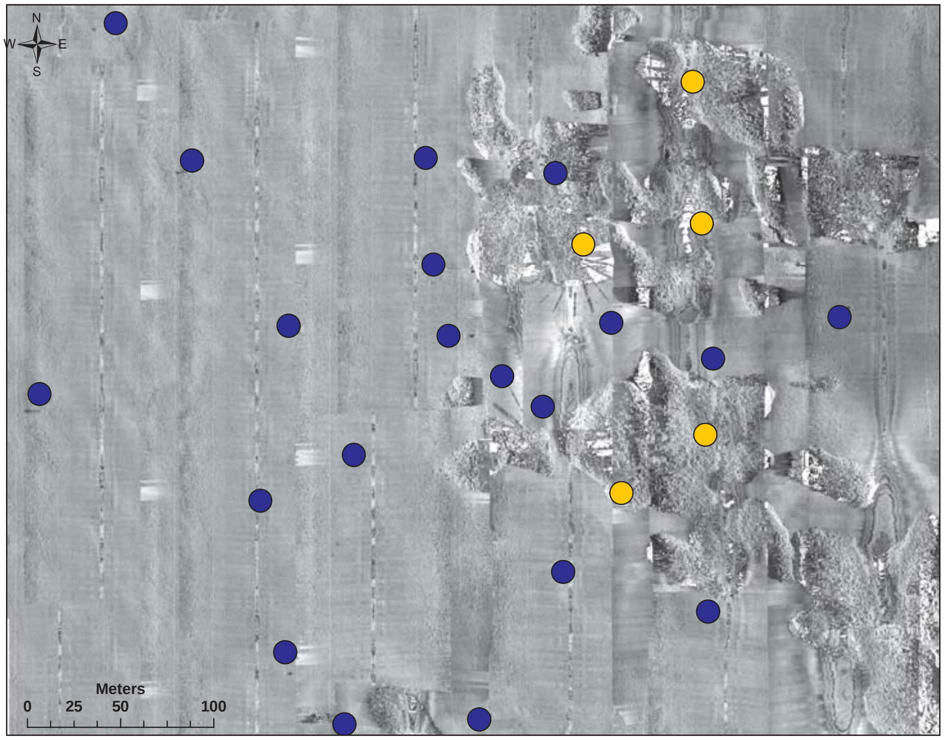

Figure 3.7. Mosaic of sidescan imagery of test area 1; note the flat homogeneous substrate on the left and high relief reef on the right. Blue circles indicate traps that were successfully identified and orange circles indicate traps that were not detected.

identified, their unique shape enabled them to be confidently classified as a trap (Figure 3.8A). In contrast, lobster traps are one third smaller than the chevron and square traps, but had 50% detection rates on hard bottom. Square traps were detected with a 66% success rate on reef habitats.

Of the 37 successful trap detections, 28 were classified as trap and 9 as non-trap/objects of interest. Two-thirds (14 of 21) of the targets on sand and rhodolith habitats were classified as traps with high confidence (Table 3.3). Square traps comprised the majority of the low confidence detections on sand and rhodolith habitats; one was the result of a trap that must have landed on its side (Figure 3.8B). The majority of lobster trap detections were classified as traps, but with low confidence or non-trap/ objects of interest (Figure 3.8C). In fact, three of the five detections were targets on reef habitat (Table 3.3). It was difficult to see trap outlines on broad complex reef habitats (Figure 3.8D), but depending on the placement, the trap would be evident (Figure 3.8E).

Benthic relief or complexity appeared to be a complicating factor where areas that had a slope of 5% or greater had low detection rates. The AUV could not navigate on habitats with slope greater than 15%.

Table 3.3. Matrix of trap detections and classificiation made through the visual interpretation of sidescan imagery from both test areas. Confidence 4 (Conf. 4) is the highest amount of certainty. Conf.=Confidence; OI=non-trap/object of interest; ND=trap not detected.

Habitat type	Trap Type	Conf. 1	Conf. 2	Conf. 3	Conf. 4	OI	ND
Sand	chevron				2		
	lobster					2	
	square	2	1	1	2	3	
Rhodolith	chevron				2		
	lobster	1				3	
	square	1	2	2	5		2
Aggregated patch reef	chevron						1
	lobster	1					1
	square		1		2	1	4
Aggregated reef	chevron						3
	lobster		1				1
	square	1		1			1

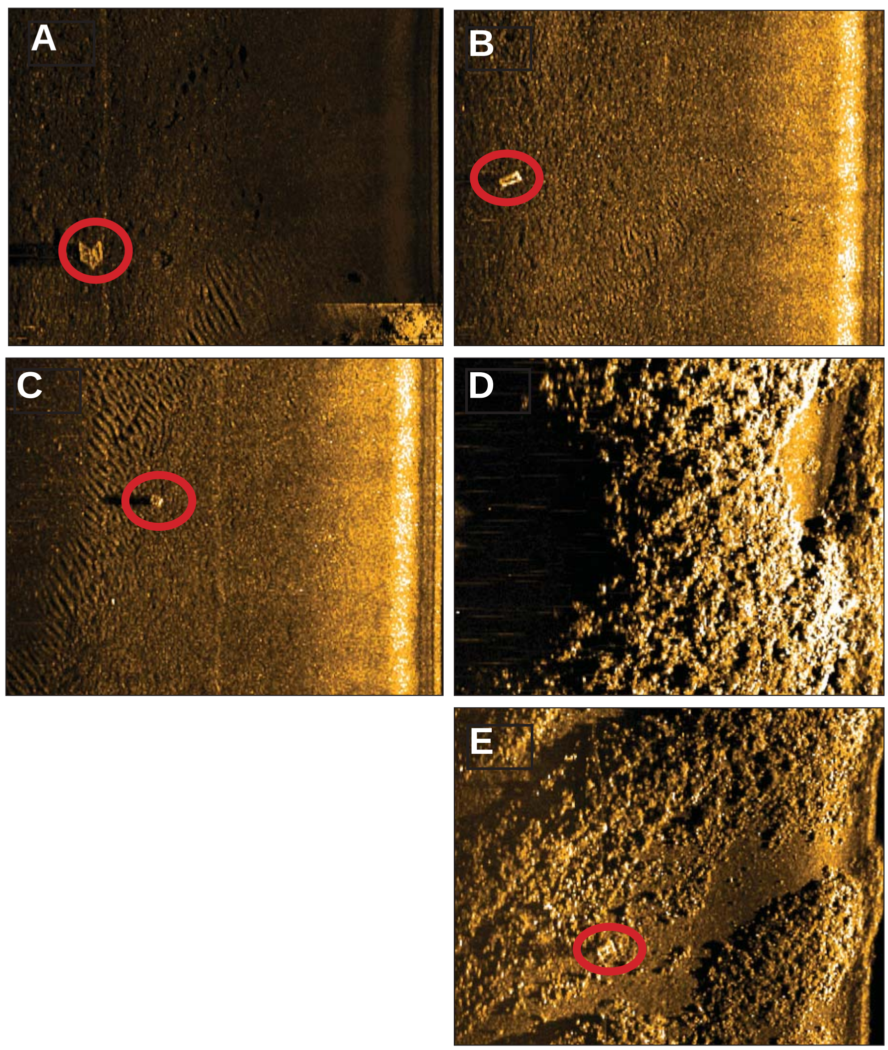

Figure 3.8. A) Chevron trap on sand, B) square trap on its side on sand habitat, C) lobster trap on rhodolith habitat, D) complex aggregate patch reef (no trap present), and E) square trap on aggregate patch reef. Trap in each figure is indicated with red circle.

3.3.2. Derelict Fish Trap Surveys

Six areas were surveyed with the AUV (Table 3.4) that encompassed an area of 8.62 km^2. After reviewing the imagery, 165 targets were identified as either a trap or object of interest. Most targets, 74%, were considered traps with a mean abundance of 14.15 traps/km^2. Objects of interest were less frequently encountered with a mean abundance of 4.9 objects/km^2.

Table 3.4. Derelict fish trap (DFT) target information, verification and target trap type (live, DFT or non trap). OI=non trap/object of interest. Non trap is a target identified as a trap in the sidescan imagery but verification revealed that the target was not a trap.

Survey Location	Area Surveyed (km^2)	Targets		# targets verified		Trap ID		
		OI	Traps	OI	Trap	Derelict	Live	Non trap
Dog Island	1.82	11	48	0	25	0	12	13
French Cap	1.12	5	20	3	20	3	7	10
Magens Bay	1.59	6	7	0	0	0	0	0
VICR	2.27	14	39	14	39	18	8*	17
St. John Shelf	0.47	1	3	1	3	1	1	1
Thatch Cay	1.35	6	5	0	0	0	0	0
Totals	**8.62**	**43**	**122**	**18**	**87**	**22**	**20**	**41**

** indicates live traps were encountered at verification that were not present at the time of AUV survey.*

Target verification took place approximately 5 months after the AUV survey. No significant storms had come through the region during this time. Verification was accomplished using an ROV launched from *R/V Nancy Foster*. Nine survey lines were conducted that recorded 15.5 km of video searching for the pre-identified trap targets. Only trap targets were actively searched for with ROVs, which included approximately 70% (87 trap targets and 18 objects of interest) of the targets. More than 50% of the trap targets were verified as traps and two items of interest turned out to be a trap. Each target did not necessarily reflect one trap. For example, one of the non-trap/objects of interest was in fact a congregation of nine lobster traps. Similarly, three single trap targets in the sidescan imagery were determined to be a cluster of multiple traps. Overall, 43 derelict traps and 23 live traps were verified from the 87 targets, a subset of the total targets identified. Interestingly, seven derelict traps and one live trap were found in the ROV survey that was not identified in the AUV imagery. Forty targets initially classified as trap targets were verified as non-trap objects or just features within the substrate. Verified trap abundance based on survey area amounted to 7.6 total traps/km^2 and 4.9 derelict traps/km^2.

The majority of trap targets were observed around Dog Island (Figure 3.9A) and the VICR (Figure 3.10A). Depths at the Dog Island survey site ranged from 28-32 m and the habitats were mostly

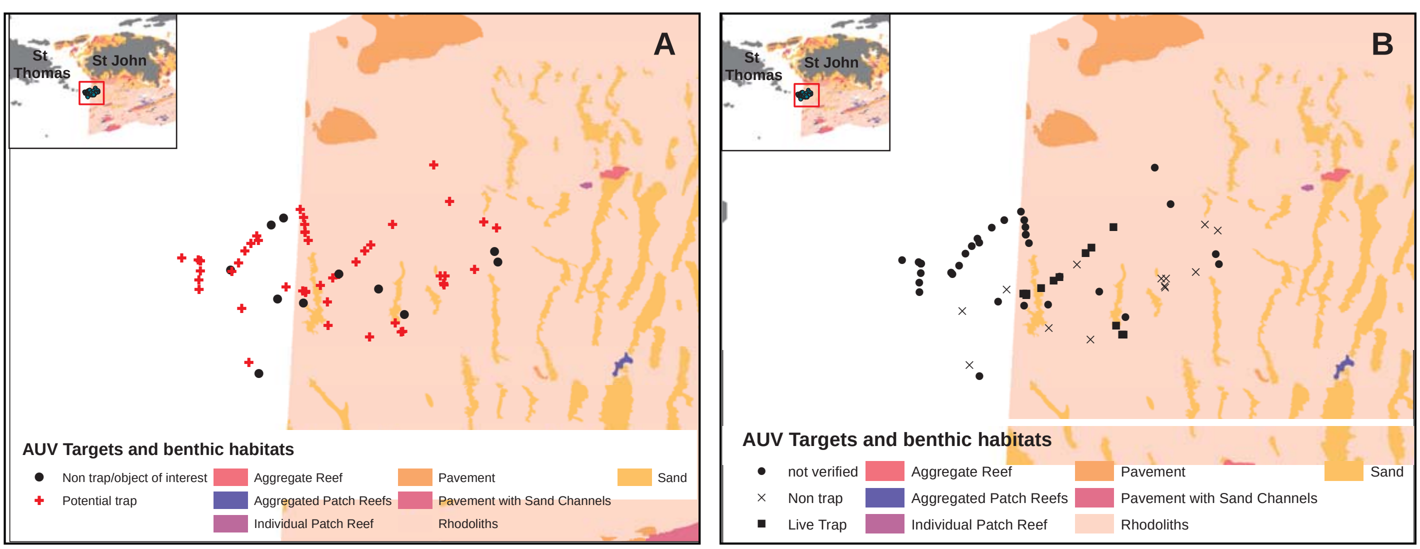

Figure 3.9. A) Trap targets identified at Dog Island survey site, October 2010. B) Trap verification, March/April 2011. Extent of benthic habitat map extends to the middle of the survey site.

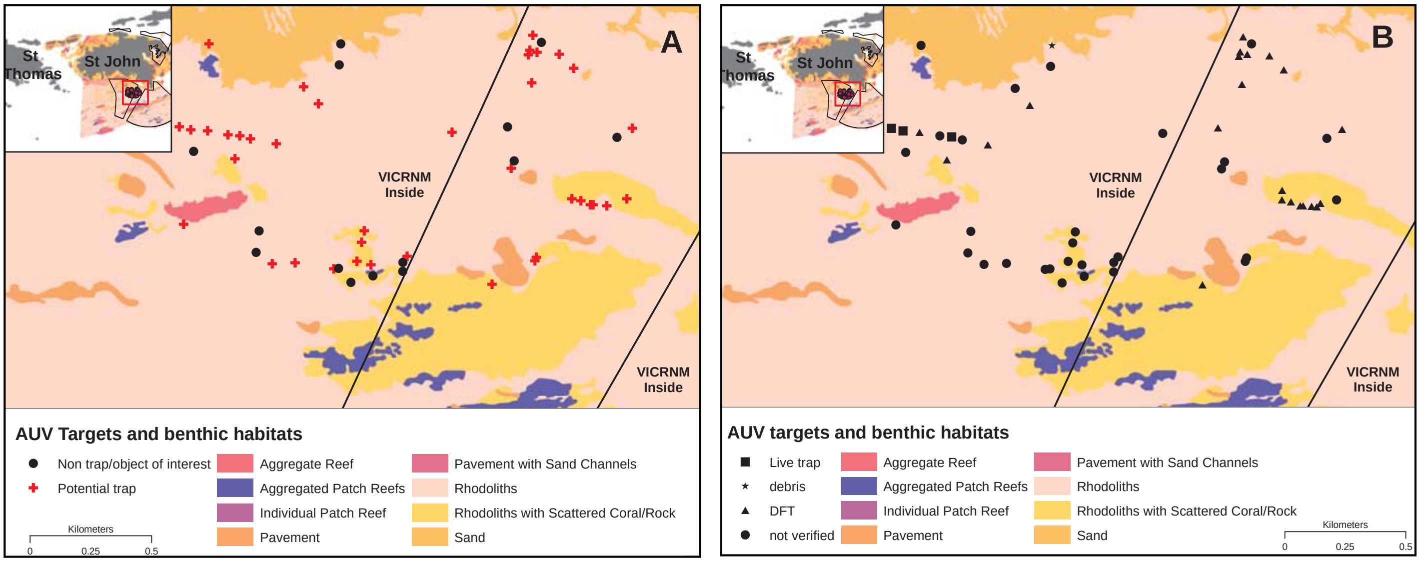

Figure 3.10. A) Trap targets identified at the VICRNM survey site, October 2010. B) Trap verification, March/April 2011.

homogenous sand and rhodolith, although the habitat map did not cover the western portion of the survey area. Areas of higher relief are in close proximity to west and north of the survey area. Overall, 48 targets were detected that were potential traps and 11 that were classified as non-trap. Tweny-five of the trap targets at the Dog Island site were verified and none were labeled as DFT (Figure 3.9B). Twelve targets were classified as active traps as there was no trap(s) present when verification was conducted. One can presume that these traps were actively fished during AUV surveys and there was sufficient time for them to be removed by the time verification was conducted. Five of the targets were classed as Confidence 1, one at Confidence 2, and six at Confidence 3.

There were 53 total targets at the VICRNM site where 14 were classified as non-trap/objects of interest and 39 classified as trap (Table 3.4). Twelve of the non-trap targets were natural benthic features. One was unidentified as marine debris and the other was a pile of nine lobster traps. Of the 39 trap targets, 17 were verified as non-trap and 18 were confirmed DFT (Figures 3.10A and B). Most of the trap targets that were verified as non-trap had low confidence, either 1 or 2. Most trap targets that were either active or DFT were classified with confidence 2 or greater. Seven active traps were observed, four of which were found inside the VICR, a no-take marine protected area (MPA). Four targets contained more than one trap. Of the 20 verified targets that were traps and one non-trap (pile of lobster traps), we observed a total of 33 total traps.

Other survey sites yielded three DFT at French Cap and one on the deep shelf of St. John. In addition, eight live traps were observed among those two sites. No target verification surveys were done at Magen's Bay or Thatch Cay.

Although survey areas were limited in correlation with STFA effort information, results indicate that most DFTs were found in areas that were reported as low fishing effort (the area in the wedge of the VICR). Overall the amount of survey area was not sufficient to get a confident estimate of the correlation between DFT abundance and STFA or DPNR fishing effort for either fish or lobster.

3.4. CONCLUSIONS

Sidescan imagery has been used to detect fish traps on homogeneous, flat substrates in a variety of marine ecosystems. The methods and results described here provide the first attempt to use sidescan sonar to detect traps (both DFT and active) in a coral reef ecosystem. In a controlled environment with known trap locations on low and high relief substrates, traps were detected and confirmed at a high success rate on low to moderate areas of relief. Trap detection was less successful on areas with high relief. During target verification, most, if not all, DFT sightings were on low and moderate relief habitats, either sand, colonized pavement or low relief aggregate reef. Among the 62.5 km of ROV surveys conducted by NOAA's National Ocean Service (NOS) in April 2011, no DFTs or live traps were observed on high relief substrate. This corroborates with the STFA as they claim that they deploy traps on flat substrates near reef structures. One can assume that most DFTs will be found on low relief habitats, with a high success rate of detection by the sidescan, and that DFTs will likely not be present on complex reef structures. Using a 40 m swath width for future survey designs and eliminating high relief areas is recommended. However, soft bottom areas adjacent to areas of high relief can be a hotspot of trap accumulation as identified by STFA. Improved AUV navigation systems with better bottom tracking and avoidance capabilities would allow surveys to be conducted over habitats with higher relief.

Target verification also provided insight into DFT condition. There was significant variability in the amount of colonization and the taxa comprising the colonization. There was also variability in the level of trap degradation (Figure 3.11A-D). Traps ghost fishing were not observed, although it was difficult to see the status of most traps and to verify that the escape panels were open or closed. However, it was apparent that traps that were intact were not capturing substantial amounts of biota. Overall, the vast majority of traps observed with the ROV were either severely degraded or the escape panels were open.

Figure 3.11. Traps identified in remotely operated vehicle (ROV) verification, March/April 2011. A) Active fish trap with snapper inside. B) Derelict trap with little fouling. C) Derelict trap with heavy sponge and algal fouling. D) Severely deteriorated chevron trap. Photos: NOAA/NCCOS/CCMA Biogeography Branch.

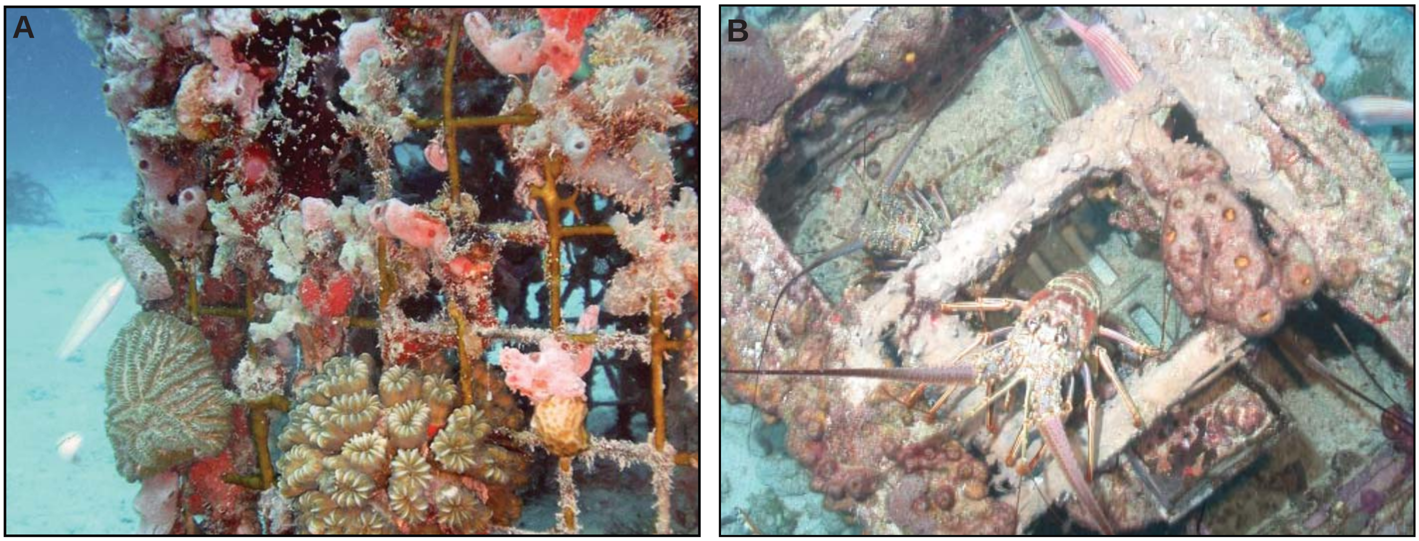

Figure 3.12. Biological communities associated with derelict traps. A) Colorful coral and sponge communities. B) Lobster, grunt, and squirrelfish have found refuge in derelict lobster trap. Photos: NOAA/NCCOS/CCMA Biogeography Branch.

This visualization also provided insight into trap impacts. Very few traps were found on reefs and nearly all were found on low relief hardbottom or sand. The impacts of DFT's in coral reef ecosystems may include the following:

- continued catch of target and non-target species,
- interactions with threatened/endangered species,
- physical impacts on the benthos,
- a role as a vector for invasive species, and
- introduction of synthetic material into the marine food web.

One can suspect that there is an abundance of derelict traps in the region based on the anecdotal information from photographs that show how derelict traps interact with surrounding habitat. Fouling and invertebrate communities on DFTs can enhance diversity by providing structure and/or food for other biota (Figure 3.12a-b). This has been observed in the Chesapeake Bay (Havens et al., 2008); Georgia (Manley et al., 2009); and, North Carolina (Uhrin and Schellinger, 2011). In fact, some agencies are using crab pots as substrate for oyster reef restoration (Brumbaugh et al., 2009; Manley et al., 2009; Kreutzer, 2010). During this project, ROV surveys discovered DFTs acting as surrogate reef habitat for lobsters, fish, and lionfish. One of the experimental traps (see Chapter 4) placed in seagrass was recruited by several fish species (Figure 3.13). Further investigations should be conducted to see how DFTs contribute to habitat quality and assessing trap removal priorities. Obviously, traps that have become part of the surrounding habitat shouldn't be removed, but standards should be developed to assess a removal process.

Figure 3.13. Juvenile grunt and goatfish recruiting to experimental trap in Flat Cay, St. Thomas, USVI. Photo: NOAA/NCCOS/CCMA Biogeography Branch.

The DFT surveys highlight a potentially significant management issue. Four DFTs and four active fish traps were identified in the VICR, a marine protected area with no take regulations. Currents or wave action could have relocated the DFTs into the Monument and their origins are uncertain, but active fishing in the MPA is concerning. Because of its diverse reef structure, fishermen have fished the “wedge”, a gap in the monument that extends southeast of St. John (Boulon et al., 2008). Targeted fishing effort in the wedge could be a source of DFT in the Monument.

Overall, 8-9 km^2 were surveyed with the AUV. Based on this limited sample area, DFT per unit area was significantly lower in the St. Thomas/St. John survey areas (4.98/km^2) than those in the Chesapeake Bay, 20/km^2 (Havens et al., 2008) or Port Susan, WA, 35/km^2 (NRC, 2006). In the wider Caribbean, DFTs have not been quantified and rates of loss or density are unknown. The only documentation of DFT loss comes from the Guadaloupe trap fishery, where it is estimated that 20,000 traps are lost annually (Burke and Maidens, 2004). The DFT density estimates reported here may be low suggesting that more surveys need be conducted, especially in areas of high fishing effort.

Next steps should include continued AUV surveys to get a better assessment of DFT abundance in the region. Further investigation on the impact of removing DFTs is also needed. The fishermen and the local coastal zone managers need to discuss the options for trap disposal and develop a feasible alternative besides dumping them at sea.

3.5. REFERENCES

Boulon, R.H., M.E. Monaco, A.M. Friedlander, C. Caldow, J. Christensen, C. Rogers, J. Beets, J. Miller, and S.D. Hile. 2008. An ecological correction to marine reserve boundaries in the US Virgin Islands. pp. 1074- 1077. In: B.M. Riegl and R.E. Dodge (eds.). Proceedings of the 11th International Coral Reef Symposium, Vol. 2. Ft. Lauderdale, Florida, USA. 1425 pp.

Brumbaugh, R.D. and L.D. Coen. 2009. Contemporary approaches for small-scale oyster reef restoration to address substrate versus recruitment limitation: a review, comments relevant for the Olympia oyster, *Ostrea lurida* Carpenter 1864. Journal of Shellfish Research 28(1):147–161.

Burke, L. and J. Maidens (eds.). 2004. Reefs at Risk in the Caribbean. World Resources Institute. Washington, DC. 80 pp. Online: http://pdf.wri.org/reefs_caribbean_full.pdf (Accessed 9 April 2012)

Foster, M.S. 2001. Rhodoliths: Between rocks and hard places. Journal of Phycology 37:659-667.

Havens, K.J., D.M. Bilkovic, D. Stanhope, K. Angstadt, and C. Hershner. 2008. The effects of derelict blue crab traps on marine organisms in the lower York River, Virginia. North American Journal of Fisheries Management 28:1194-1200.

Kendall, M.S., C.R. Kruer, K.R. Buja, J.D. Christensen, M. Finkbeiner, and M.E. Monacao. 2001. Methods used to map the benthic habitats of Puerto Rico and the U.S. Virgin Islands. NOAA/NOS Biogeography Tech Report. Silver Spring, MD. 45 pp.

Kreutzer, A.D. 2010. The role of crab traps in oyster restoration. M.S. Thesis, College of Charleston, Charleston, South Carolina. 60 pp.

Manley, J., A.J. Power, R. Walker, D. Hurley, C. Belcher, and M. Gilligan. 2009. Evaluation of Eastern Oysters, *Crassostrea virginica* (Gmelin, 1791), restoration techniques for use in intertidal southeastern United States habitats characterized by heavy siltation rates. Occasional Papers of the University of Georgia Marine Extension Service Vol 9. 32 pp. Online: http://georgiaseagrant.uga.edu/images/uploads/media/Oyster_Vol9.pdf (Accessed 9 April 2012)

Natural Resources Consultants, Inc. (NRC). 2006. Derelict fishing gear: Identification and removal project, Port Susan: Final Report. Seattle, WA. 14 pp.

Uhrin, A. V. and J. Schellinger. 2011. Marine debris impacts to a tidal fringing-marsh in North Carolina. Marine Pollution Bulletin 62:2605-2610.

Chapter 4: Ecological Impact of Derelict Traps

Gabrielle Renchen[1], Simon J. Pittman[1,2,3] and Ronald L. Hill[4]

4.1. BACKGROUND AND INTRODUCTION

It has been assumed that lost or derelict traps continue to trap fish causing unintended mortality (i.e., ghost fishing) and may also damage benthic habitat through snagging and wave driven movements. However, there have been no quantitative studies and remarkably few surveys to estimate the prevalence of ghost fishing in the Caribbean and to assess the impact on marine fish communities. Where studies on derelict fishing gear have occurred, they mainly focused on nets rather than traps and results are highly variable. In instances where traps were studied, crabs and lobster were the focus rather than finfish (Matsuoka et al., 2005). In the Gulf of Oman, a study of finfish and traps estimated ghost fishing mortality at 1.34 kg/trap per day, decreasing over time with an estimated cost due to lost fish valued at $145 per trap over a three month period and $168 per trap over six months (Al-Masroori et al., 2004). In Japan, Matsuoka et al. (2005) found that fewer organisms were observed in the traps that were largely deformed due to breakage of frames, buried in sediment, and covered by accumulated fouling organisms. No equivalent research has been conducted for any of the Caribbean trap fisheries.

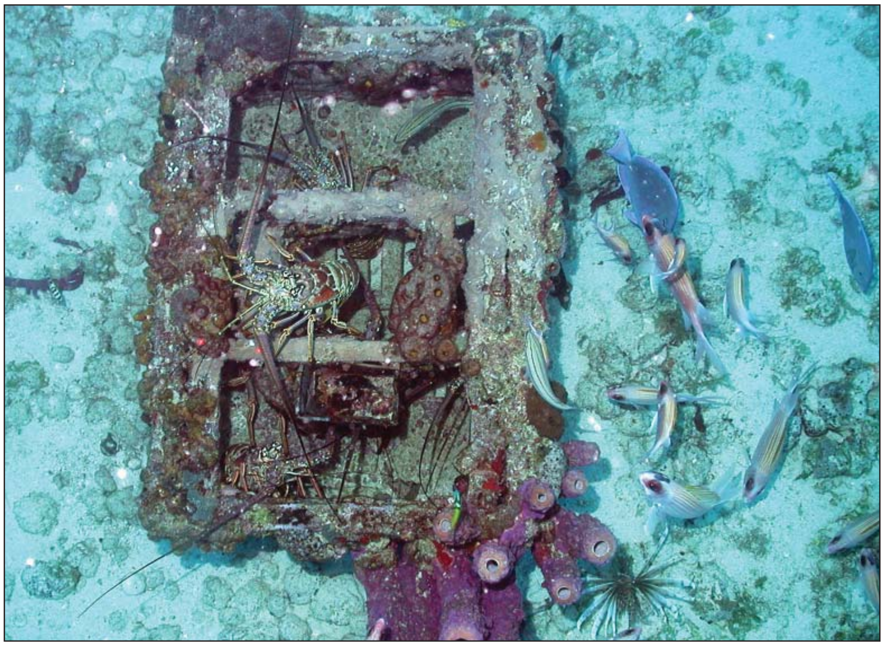

Figure 4.1. Heavy colonization and recruitment to a derelict lobster trap. Photo: NOAA/NCCOS/CCMA Biogeography Branch.

In addition, very little is known about the benthic communities that colonize derelict fish traps (DFTs; (Figure 4.1). The development of marine fouling communities is a natural process resulting from the settlement and subsequent growth of algae and invertebrates on submerged materials (Evans, 1981). Fouling community structure may vary temporally and spatially along different gradients, such as latitudinal, inshore to offshore or on a finer scale between habitats based on breeding times and larval transport, as well as with changes in water quality (Holmes et al., 1997). The structure of fouling communities can be analyzed spatially or temporally for sequential patterns of community change, also known as seriation (Clarke et al., 1993). An understanding of the community composition of fouling communities and the sequence of colonization may offer an opportunity to estimate the age of DFTs. Breakdowns in seriation may indicate the lack of stability in fouling communities and modification by disturbances (Warwick et al., 2002; Clarke et al., 1993).

Our study provides the first experimental, long-term assessment of derelict traps in the Caribbean. The impetus for this study emerged from mutual needs expressed by fishermen and managers

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for a greater scientific understanding of derelict fish traps. We implemented field experiments that simulated the processes associated with derelict traps and then made extensive short and long term field observations on the ecological patterns and processes affecting derelict traps, including fish capture and escapement, colonization and degradation rates, trap mobility, and other interactions with marine fauna. This study quantifies rates of ghost fishing and determines if it is advisable to locate and remove derelict traps to reduce further impacts or if it is better to leave them in place to be incorporated into the substrate. The data will support and complement other studies that are aimed at reducing by-catch from fish traps (i.e., St. Thomas Fishermen's Association [STFA] and NOAA's National Marine Fisheries Service [NMFS] cooperative research into the effectiveness of escape vents). Fishermen may also benefit from underwater observations on how fish interact with traps during different times of the day and night and in different habitat types and across shelf locations.

Several studies have investigated the impacts of the trap fishery on bottom habitats in the US Caribbean. Appeldoorn et al. (2000) noted damage from traps during setting, fishing, and hauling in southwestern Puerto Rico. They studied traps set in hardbottom or reef habitats to examine potential for damage to corals and other reef organisms. They found that all traps (N=10) caused some damage to either stony corals, gorgonians, or sponges, with an average of 70 cm^2 damage in the 1 m^2 area under the traps. Quandt (1999) conducted a small study of traps set on coral reefs off St. Thomas and reported scrapes and breakage to 5% of all corals observed and tissue damage to 47% of all gorgonians observed; gorgonians were the dominant benthic component in the study site. In an extensive study in Florida, Puerto Rico and U.S. Virgin Islands (USVI), boat surveys recorded trap locations, while surveys were used to examine fine scale habitat characteristics and damage to benthic organisms in coral reefs and associated habitats (with the caveat that diver surveys were limited to <40 m due to safety concerns). Initial estimates from 182 diver-based trap surveys throughout the USVI attributed damage to some structural organism in 50% of the traps surveyed, although sponges and gorgonians were predominantly affected rather than stony corals (Sheridan et al., 2005). Data from St. Thomas demonstrated damage from 53% of the traps observed and identified key scleractinian corals (i.e., *Montastrea* and *Diploria*) and soft corals (i.e., *Pseudopterogorgia* and *Plexaura*) as being the most common benthic organism under traps (ranging from 33-53% cover) followed by three additional octocoral genera (i.e., *Erythropodium*, *Gorgonia*, and *Eunicea* spp.), ranging from 20-30% cover (Hill, unpub. ms.). Surveys conducted in the Florida Keys demonstrated benthic habitat damage also occurred when traps were shifted within a habitat or into other habitats during storm events (Lewis et al., 2009). Displacement of traps by somewhat minor storm events may contribute to higher loss rates and conversion of fishing traps to derelict traps, and this may be exacerbated in the USVI by the common practice of fishing unbuoyed traps (D. Olsen, pers. comm.).

4.2. EXPERIMENTAL DESIGN

Twelve new fish traps were deployed to simulate derelict traps using a stratified sampling design that included near-reef and far-reef samples at both nearshore (~10 m depth) and offshore (~20 m depth) stations (Renchen, 2011). The strata used for each station consisted of: 1) sand immediately adjacent to a coral reef (within 1 m); and 2) sand or seagrass more distant from coral reefs (≥80 m). Benthic habitat strata were defined using NOAA's benthic habitat maps (Kendall et al., 2001). The inshore location, Perseverance Bay (18°20'22.96" N, 64°59'34.73" W), is an open bay on the southern shore of St. Thomas, while the offshore location, Flat Cay (18°19'00.86" N, 64°59'22.96" W) is a small uninhabited island approximately two kilometers south of Perseverance Bay (Figure 4.2). Evidence of historical trap fishing at both locations was supported by sightings of derelict fish traps and actively fished traps.

Among the 12 traps placed, two commonly used trap designs were constructed for the experiments: 1) the chevron or arrowhead trap (Figure 4.3A) and 2) and the more common rectangular trap

(Figure 4.3B). The fish traps were constructed by a local St. Thomas fisherman with a steel rebar frame, horseneck funnel and two inch vinyl coated square mesh. Trap design was not used as an experimental factor, but general comparative observations were made between the two designs. Permanent markers were placed in the substrate at 40 cm from two sides of each trap to monitor potential trap movement.

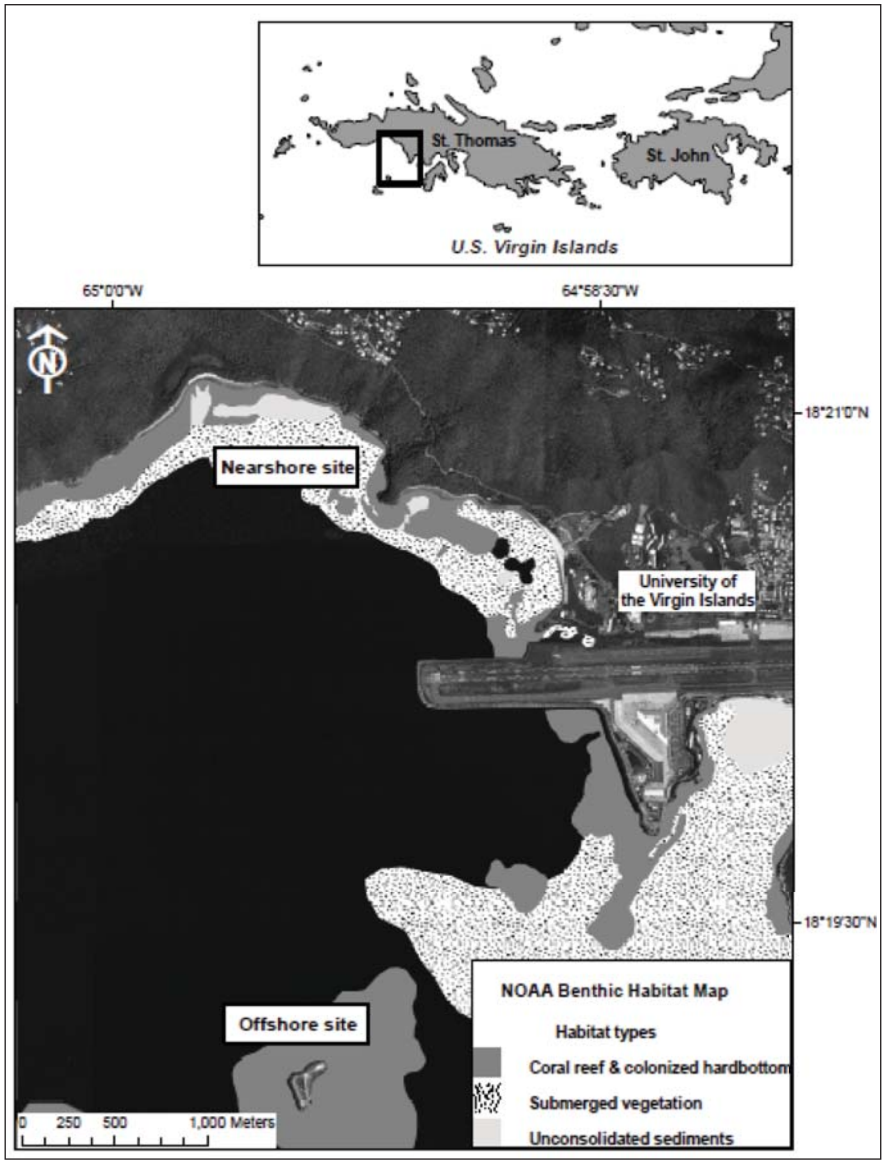

Figure 4.2. Location of nearshore and offshore experimental trap survey areas.

Initially, six of the traps were deployed with the escape panels open to represent either a discarded trap or a trap lost long enough that the panels had opened, and six with the escape panels closed simulating recent trap loss. Traps were not baited as there was no existing information regarding fishermen's preference to bait traps. The design compared mortality rates of traps setup to fish versus those with escape panels opened. Escape panels on fishing traps were tied closed using untreated 0.31 cm (1/8 in) jute twine as required by territorial and federal regulations.

Traps were surveyed during daylight

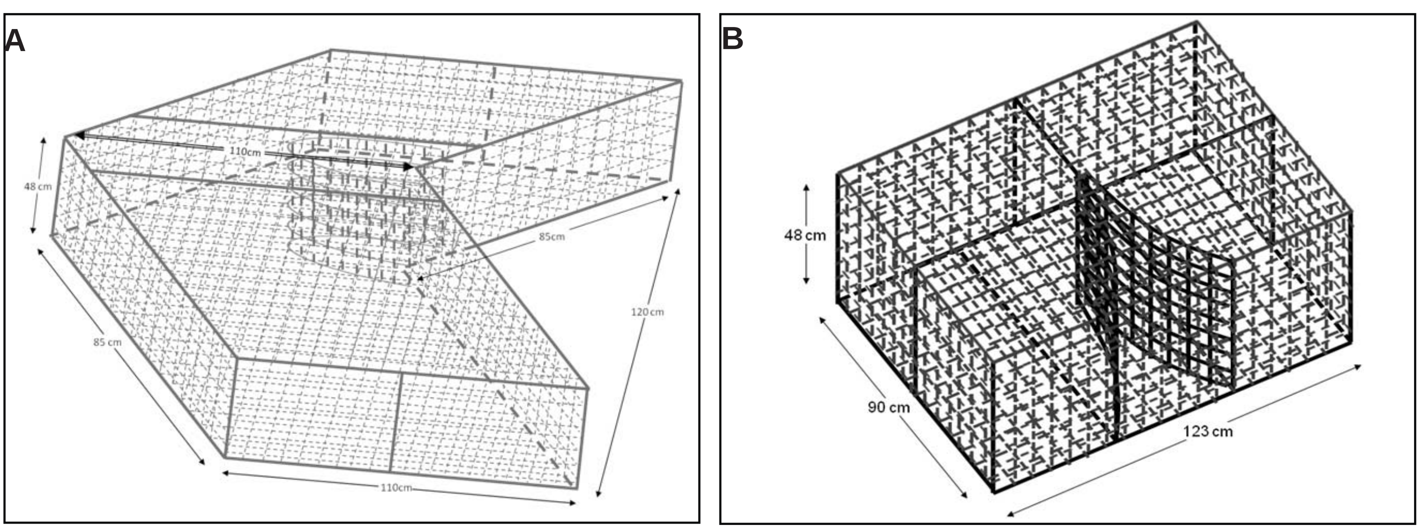

Figure 4.3. Dimensions of traps used for catch and fouling experiments: A) chevron or arrowhead trap and B) rectangular trap. Sketches provided by C. Jeffrey, NOAA/NCCOS/CCMA Biogeography Branch.

hours using SCUBA three days a week for six months (53 total surveys for the offshore traps; 58 surveys for the inshore traps) to quantify fish and macroinvertebrate species abundance, size, mortality, and behavior within traps. Behavioral observations were recorded at a distance of several meters to reduce the effect of the diver's presence on the fish. All species and individual fish were photographed to aid in estimation of residence times in traps, and to assess any physical damage to the fish. In addition, the abundance, size and general behavior of fish within one meter outside of the trap were also recorded.

Diurnal observations were also conducted (Renchen, 2011). A trap was deployed with a video camera set up outside the trap 1 m from the entrance funnel of the trap (Figure 4.4). We used a custom-built underwater video surveillance system to continuously record in high resolution for 24 hours. The entire system was mounted in a polyvinyl chloride (PVC) underwater housing rated to 130 feet in depth (Renchen et al., in press).

Figure 4.4. TrapCam. Photos: NOAA/NCCOS/CCMA Biogeography Branch.

Video samples were analyzed for fish entry and exit times, activity periods, and behavioral interactions of fish both inside and outside the traps.

Economic loss through mortality was determined by calculating the length-weight relationship for each species, quantifying mortality rates, and scaling by current market prices. Jute deterioration parameters on each actively-fishing trap, documented through photography, were recorded as the number of days it took for the cord to completely break and the number of days for the escape panel to open.

To assess the condition and biotic colonization (i.e., seriation) of traps, surveys were conducted using SCUBA once a month from January 2010 to January 2011. Each trap was divided into three substrates for analysis purposes: the open, square areas between the mesh (henceforth referred to as inner mesh area), the wire mesh itself and the rebar frame. The inner mesh areas were systematically enumerated for each side of a trap and 25% of inner mesh areas were randomly chosen to monitor over time. The same process was used for the mesh and rebar substrates. The percent cover of fouling organisms was determined at each randomly selected substrate. Fouling organisms were identified to the taxonomic level of phylum (Rhodophyta, Cholorphyta, etc.), as specimens were not collected to positively identify the species present with a microscope. Because any submerged material becomes covered in a biofilm within hours of submergence, the percent cover of a biofilm was only considered in the percent cover estimates if the substrate (trap materials) could not be seen underneath.

4.3. MORTALITY AND TRAP RESIDENCE TIME

A two-way analysis of variance (ANOVA) was used at both the species and family level to test for between-location differences in the number of consecutive days (log transformed) fish spent in the traps at both inshore and offshore locations and near-reef and far-reef habitats. A Wilcoxon test was used to test for differences in mortality between inshore and offshore locations. We applied a suite of multivariate analyses in PRIMER v6 software (Clarke and Warwick, 2001) to examine differences and similarities in fish assemblages by location. Fish information was pooled by site and each trap was used as a replicate. Non-parametric multi-dimensional scaling (nMDS) was used to visually compare assemblage similarities. Analysis of similarities (ANOSIM) was used to test for significant differences, while similarity percentages (SIMPER) were used to determine the contribution of species to assemblage dissimilarity. Multivariate analyses were conducted only on closed traps and used untransformed presence-absence data rather than abundance to avoid the confounding effect from repeat counts of the same individual over multiple trap surveys.

The value of commercial fish during the study period was obtained from STFA and from the local fish markets. The weight of individual fish species was calculated using the length-weight relationship, $W=a\cdot L^b$, with a and b parameters for each species collected from FishBase (Froese and Pauly, 2011). Weight was used to calculate market prices (US$ per lb) for an estimation of economic loss.

During the six month study period, 453 fish comprising 21 families and 42 species were observed (Table 4.1) within the six experimental traps set with the escape panels closed (henceforth referred to as closed). The most frequently observed families (Figure 4.5A) were: surgeonfish (Acanthuridae, n=131 individuals); snapper (Lutjanidae, n=63); porgy (Sparidae, n=47); angelfish (Pomacanthidae, n=39); and boxfish (Ostraciidae, n=38). The most frequently observed species were: Blue Tang (*Acanthurus coeruleus*; n=67 individuals); Doctorfish (*Acanthurus chirurgus*; n=47); Saucereye Porgy (*Calamus calamus*; n=47); Schoolmaster Snapper (*Lutjanus apodus*, n=40); Gray Angelfish (*Pomacanthus arcuatus*; n=33); and Smooth Trunkfish (*Lactophrys triqueter*; n=32). The minimum and

Table 4.1. Summary catch information for the experimental derelict traps. Subscripts indicate open (o) and closed (c) traps and inshore (i) and offshore (o) location.

Trap	GPS Location	Habitat	Total # fish caught	Live fish	Proportion of Mortality (Mort/total fish caught)	Biomass of Fish Caught (lbs)	Biomass of Mortality (lbs)
1_{co}	N 18.31864 W 064.99024	Coral/Near Reef	105	103	0.03	81.16	6.59
2_{co}	N18.31869 W 064.99023	Coral/Near Reef	101	100	0.01	51.87	4.32
3_{oo}	N 18.31871 W 064.99002	Coral/Near Reef	0	0	0	0	0
4_{oo}	N 18.31825 W 064.98981	Sand/Away from Reef	0	0	0	0	0
5_{co}	N 18.31827 W 064.98965	Sand/Away from Reef	66	63	0.05	54.54	1.76
6_{oo}	N 18.31825 W 064.98969	Sand/Away from Reef	0	0	0	0	0
7_{oi}	N 18.34960 W 064.99310	Coral/Near Reef	0	0	0	0	0
8_{ci}	N 18.34953 W 064.99306	Coral/Near Reef	46	44	0.04	46.26	15.19
9_{ci}	N 18.34957 W 064.99025	Sand/Away from Reef	53	50	0.06	52.71	16.53
10_{oi}	N 18.34826 W 064.98718	Sand/Away from Reef	0	0	0	0	0
11_{oi}	N 18.34818 W 064.98724	Sand/Away from Reef	1	0	1	0.96	0.96
12_{ci}	N 18.34817 W 064.98715	Sand/Away from Reef	82	75	0.09	89.88	6.06

maximum total lengths (TL) recorded for trapped fish were 4 cm and 107 cm, respectively, with a mean size of 21.1 cm (± 5.41 SE). The smallest individual observed was a Peacock Flounder (*Bothus lunatus*; 4 cm TL) and the largest was a Nurse Shark (*Ginglymostoma cirratum*; 107 cm TL). During the study, the average biomass observed in closed traps was 0.385 ± 0.326 kg and ranged from 37 g to 7.76 kg. Over 81% of the total biomass was comprised by commercial species (Figure 4.5B). Snappers contributed the most trap biomass (35%), followed by grunts (11%), porgys (10%), surgeonfish (6%), angelfish and parrotfish each at 4%, grouper and triggerfish each at 3% and jacks at 1%. The rest of the commercial species combined were less than 1% of the total biomass.

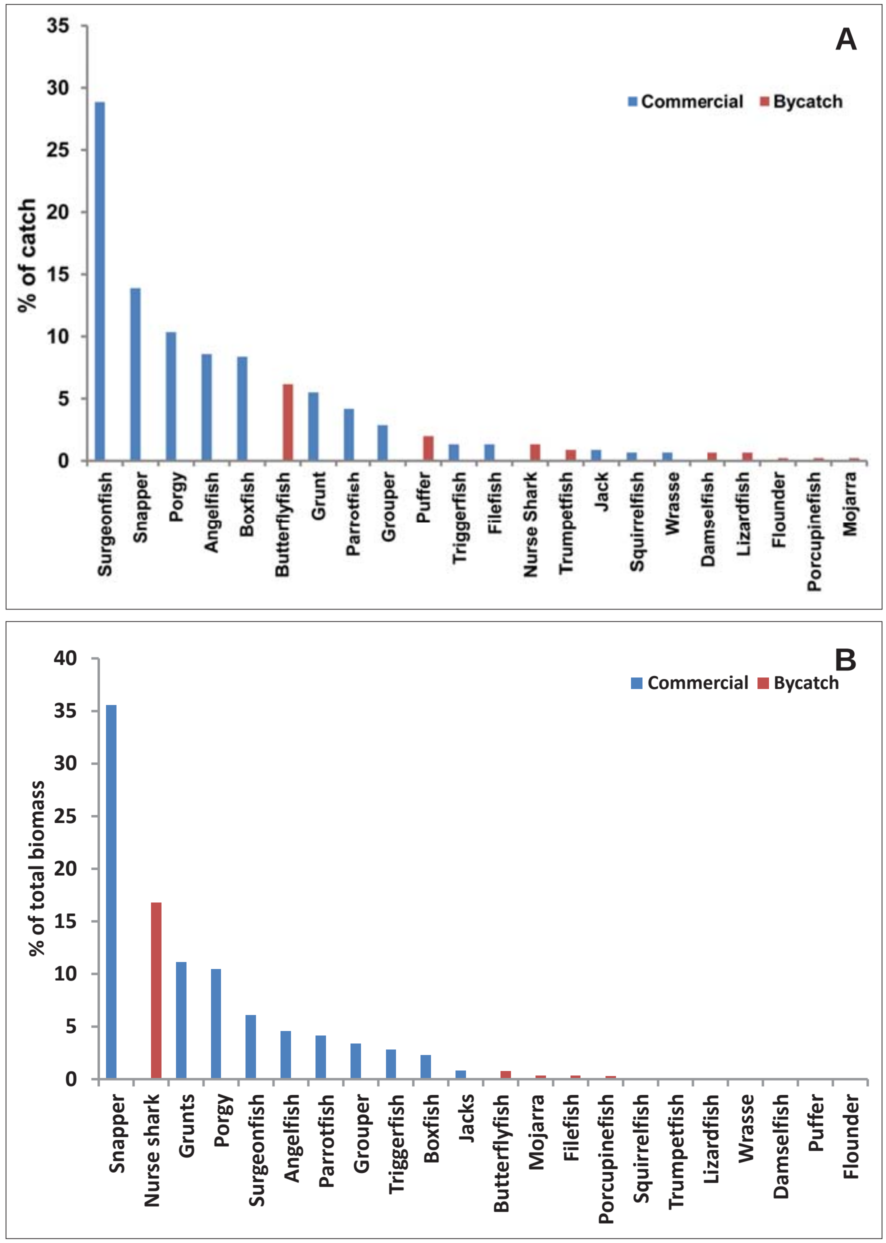

Figure 4.5. A) Percent total abundance and B) percent total biomass of fish families captured in all traps during 2010-2011.

The composition of fish assemblages was significantly different between closed traps set in inshore and offshore (ANOSIM R=0.23, p=0.01) and between those set in near-reef and far-reef habitats (ANOSIM R=0.2, p=0.01). However, the ANOSIM test and nMDS ordination plots (not shown) revealed very high assemblage similarity between locations with no clear separation in assemblage composition. The nMDS plot provided a good two-dimensional representation of the multivariate assemblage data as indicated by a low stress value (0.01) for all plots. Despite high overlap, differences in the prevalence of seven species contributed approximately 60% to the assemblage differences at inshore and offshore locations. Gray Angelfish, Blue Tang and Doctorfish were more prevalent in the offshore traps, contributing 11.8%, 8% and 7.8% to the difference, respectively. Schoolmaster, Smooth Trunkfish, Nurse Shark and Saucereye Porgy were more prevalent in our inshore traps, and contributed 9.9%, 8.9%, 7.6% and 4.6% to the difference, respectively.

Differences in the prevalence of eight species contributed approximately 60% of the difference between assemblages in traps set at near-reef and far reef habitats. Saucereye Porgy, Bluestriped Grunt (*Haemulon sciurus*) and Smooth Trunkfish contributed 9.9%, 8.72% and 5.73% to the difference, respectively, with higher prevalence in our near-reef traps. More prevalent in the far-reef traps, Schoolmaster, Gray Angelfish, Doctorfish, Blue Tang and Nurse Shark contributed 7.7%, 6.6%, 5.7%, 5.3% and 4.9 % to the difference, respectively.

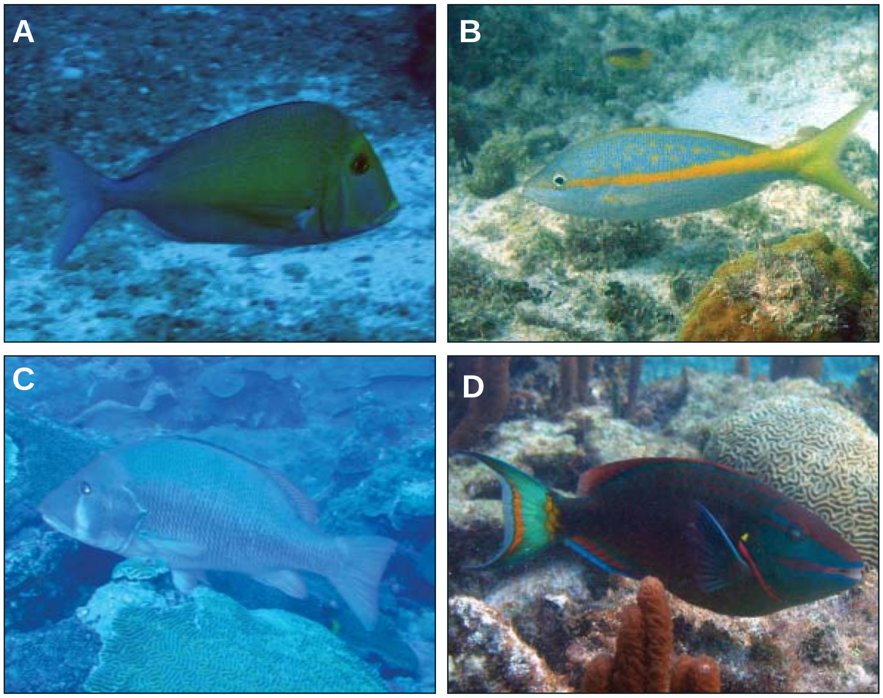

*Figure 4.6. Common fish species observed inside experimental traps A) Saucereye Porgy (*Calamus calamus*), B) Yellowtail Snapper (*Ocyurus chrysurus*), C) Dog Snapper (*Lutjanus jocu*), and D) Stoplight Parrotfish (*Sparisoma viride*). Photos: NOAA/NCCOS/CCMA Biogeography Branch.*

Fish spent an average of 8.2 (±3.4) consecutive days in the closed traps and a median of 5.5 days, with ninety-five percent of fish (433 individuals) able to escape the closed traps. Twenty fish (approximately 5%) perished in the traps. All but one of the mortalities observed were from closed traps, with one expired individual (Great Barracuda, *Sphyraena barracuda*) observed in a trap with the escape panels open. Fish attempting to escape did so by swimming throughout the trap, often colliding into the mesh, looking for an exit, particularly at the corners of the trap. Species most often observed banging into the mesh (Figure 4.6) included Saucereye Porgy, Yellowtail Snapper (*Ocyurus chyrsurus*), Dog Snapper (*Lutjanus jocu*) and Stoplight Parrotfish (*Sparisoma viride*). Approximately 5% of all trapped fish were observed with skin wounds or abrasions, while 20% of those that died had abrasions on their snouts and or foreheads due to repeatedly butting into the mesh while trying to escape (for more details see Renchen, 2011 and Renchen et al., in press). Dead fish, their skeletal remains, and in some cases fleshy remains, were observed in all of the traps that incurred mortalities. Using the price per pound for each species, the 19 mortalities (total 50.5 lbs) from closed traps amounted to an economic loss of US$156.75 over the six month study period. This is equivalent to an average of approximately US$52.25 per trap per year.

A total of 384 fish, comprising three families and six species, were observed grazing on the fouling communities growing on the experimental derelict fish traps. The species observed grazing included; Striped Parrotfish (*Scarus iseri*; n=219), Princess Parrotfish (*Scarus taeniopterus*; n=82), Ocean Surgeonfish (*Acanthurus bahianus*; n=29), Bluehead Wrasse (*Thalassoma bifasciatum*; n=22), Doctorfish (n=17) and Blue Tang (n=15). Both parrotfish species were only present in their juvenile or initial life stages. Grazers were more abundant at the shallow inshore traps (n=281), and overall 80% of the grazers were observed grazing on traps positioned in coral habitats. The average grazer size was 9 ± 0.83 cm.

A total of 100 individuals from 12 families and comprising 18 species were observed on the diurnal video. The most frequently observed fish families and species were similar to those observed by the diver surveys. A wide variety of fish behavioral interactions were observed in the video samples, with 13 behaviors expressed among individual species and families.

Overall, 75% of the observed behaviors were fish attempting to escape, either as an individual or as a school. This behavior was characterized by fish swimming throughout the trap or aggregating in one area of the trap, and typically, banging into the mesh.

Fish were also observed entering and exiting (escaping) the trap through the entrance funnel. A total of 167 entrances and 173 exits were recorded. There were more entrances and exits than the total number of fish caught because several of the fish transiting into and out from the trap were the same individuals observed on multiple occasions. Trap entrances and exits were highest at the hours of 5 am, 8 am and 12 pm (Figure 4.7).

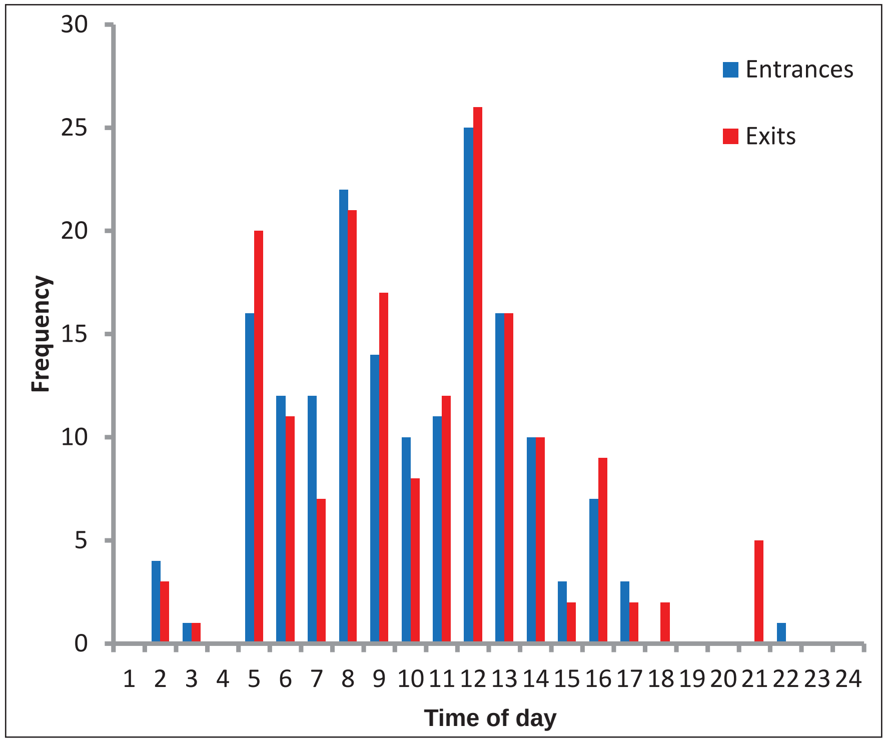

Figure 4.7. Timing of fish entrances and exits at an experimental trap.

Jute twine deterioration took approximately 2.8 ± 0.27 months (82.9 ± 8.14 days) to degrade and approximately 0.79 ± 0.24 months (23.55 ± 7.15 days) more for the escape panel to open after the twine broke. These estimates are based on new traps and new rot cords and therefore represent a worst case scenario of time for doors to open when traps are lost with newly tied cord and may not necessarily reflect the deterioration times for cord that has already been exposed to the elements. Insufficient data were collected in this study to document the range of materials and techniques used to tie escape panels shut in the USVI fishery. The documented results may be fairly representative since fishermen have been observed checking the integrity of their jute twine while working their traps and replacing it if it seems too worn (R. Hill, pers. obs.).

4.4. TRAP SERIATION AND CONDITION

Trap condition was surveyed for 12 months on traps set in the offshore habitats. Since three of the traps set on inshore habitats were swept away by Hurricane Earl in October 2010, only nine months of information were available for analysis in that stratum.

A suite of multivariate analyses in Primer v6 software (Clarke and Warwick, 2001) were applied to examine differences and similarities in fouling communities by location and habitat. nMDS was used to visually compare community similarities. ANOSIM was used to test for significant differences and similarity percentages (SIMPER) to determine the contribution of species to assemblage dissimilarity. The index of multivariate seriation (IMS; Clarke et al., 1993) was also applied to determine the degree to which fouling community change was sequential or linear. Non-metric multi-dimensional scaling ordinations for each pooled trap substrate were again used, however to better visualize the fouling community change in terms of the IMS, the ordinations were overlaid with a trajectory of each month sampled (months 1-12).

During the surveys, biofilm and 12 phyla of fouling organisms were observed growing on the twelve experimental derelict fish traps. The phyla and or film with the highest average percent cover during the surveys for the inner mesh (area between wire mesh squares) were biofilm (0.49%), cyanobacteria (Cyanophyta, 0.30%), hydroids and anemones (Cnidaria, 0.21%), red algae (Rhodophyta, 0.03%) and tunicates (Chordata, subphylum Urochordata, 0.02%). The phyla and or film with the highest average percent cover for the actual mesh were biofilm (63.5%), crustose coralline algae (CCA), (Rhodophyta, 12.24%), Cyanophyta (1%), hydroids and anemones (0.80%) and hydrocoral (0.53%).

Again, the biofilm had the highest average percent cover on the rebar (65.7%) followed by CCA (6.9%), Rhodophyta (3.8%), Mollusca (2.0%) and Cnidaria (0.60%).

After one month all traps were covered with a biofilm. The biofilm and Cnidaria were the only groups present every month of the study on all three substrates. Colonization and disappearance patterns tended to vary with the different trap substrates (Table 4.2); where the inner mesh substrate displayed the most consistency. The percent cover of fouling organisms gradually increased over time, but groups such as Cyanophyta and Rhodophyta tended to fluctuate over the course of the year with no discernible trend. Trends in the appearance of organisms were discernible among the three substrates, but were not attributable to a seasonal pattern. Trends were more evident if only the top five groups were considered at the inshore and offshore locations for the inner mesh area (Figure 4.8), mesh (Figure 4.9) and rebar (Figure 4.10). Calcifying organisms such as Mollusca, Milleporidae and Scleractinia (*Favia fragum*) were only recorded growing on the offshore traps. The percent cover of these organisms steadily increased over the twelve month period and did not fluctuate like the more ephemeral species. Although CCA was likely growing on all the traps, it could only be seen growing on the inshore traps. Ascidians were first recorded after two months, only on the offshore traps.

The composition of fouling communities growing on the inner mesh area, mesh and rebar were all significantly different between inshore and offshore locations (Table 4.3). The nMDS ordination plots (not shown) provided good representations of the data and revealed clear separation in fouling community composition on the inner mesh area, mesh and rebar (stress=0.13, 0.01 and 0.08 respectively).

*Table 4.2. Presence and absence of fouling groups observed growing on the experimental derelict traps from January 23, 2010 to January 23, 2011. Symbols represent the three trap substrates; Innermesh (**x**), Mesh (●), Rebar (■).*

	Month											
Group	**1**	**2**	3	4	5	6	7	8	9	10	11	12
Biofilm	x ● ■	x ● ■	x ● ■	x ● ■	x ● ■	x ● ■	x ● ■	x ● ■	x ● ■	x ● ■	x ● ■	x ● ■
Rhodophyta	-	■	x ●■	x ●■	x ●■	x ●■	x ●■	x ●■	x ●■	●■	●■	●■
Cholorphyta	-	-	x ■	x ■	x ■	x ■	x ■	x ■	x ■	x ■	x ■	x ■
Phaeophyta	-	-		x ●	x ●	x ●	x ●■	x ●■	x ●■	x ●■	●■	●■
Chordata	-	x ■	x ●■	x ●■	x ●■	x ●■	x ■	x ■	x ■	x ●■	x ●■	x ●■
CCA	●■	●■	●■	●■	●■	●■	●■	●■	●	-	-	-
Mollusca	-	■	x ■	x ■	x ■	x ■	x ●■	x ●■	x ●■	x ●■	x ●■	x ●■
Porifera	-	-	■	■	■	x ■	x ●	x	●	●	●	●■
Milleporida	-	-	-	-	-	-	●	●	●■	x ●■	x ●■	x ●■
Scleractinia	-	-	-	-	-	-	-	■	●■	●■	●■	x ●■
Tube Worms	-	-	■	x ■	x ■	■	■	●■	x ■	x ■	x ■	x ■
Cyanophyta	-	x	x ●■	x ●■	x ●■	x ●■	x ●■	x ●■	x ●■	x ●■	x ●■	x ●■

The differences in the percent cover of the film and three phyla contributed approximately 90% to the community differences growing on the inner mesh at inshore and offshore locations. The biofilm and oysters (Mollusca) were more abundant on offshore traps contributing 56.4% and 4.8% to the difference respectively. Cnidaria and Cyanophyta were more abundant on inshore traps contributing 21.3% and 8.5% to the difference, respectively. Differences in the percent cover of the film and two phyla contributed approximately 95% of the difference between fouling communities growing on the wire mesh at inshore and offshore locations. The biofilm and cnidarians were more abundant on the offshore traps, contributing 65.7% and 4.6%, respectively, while CCA was more abundant on inshore traps contributing 20.9% to the difference. Finally, percent cover differences of the film and three

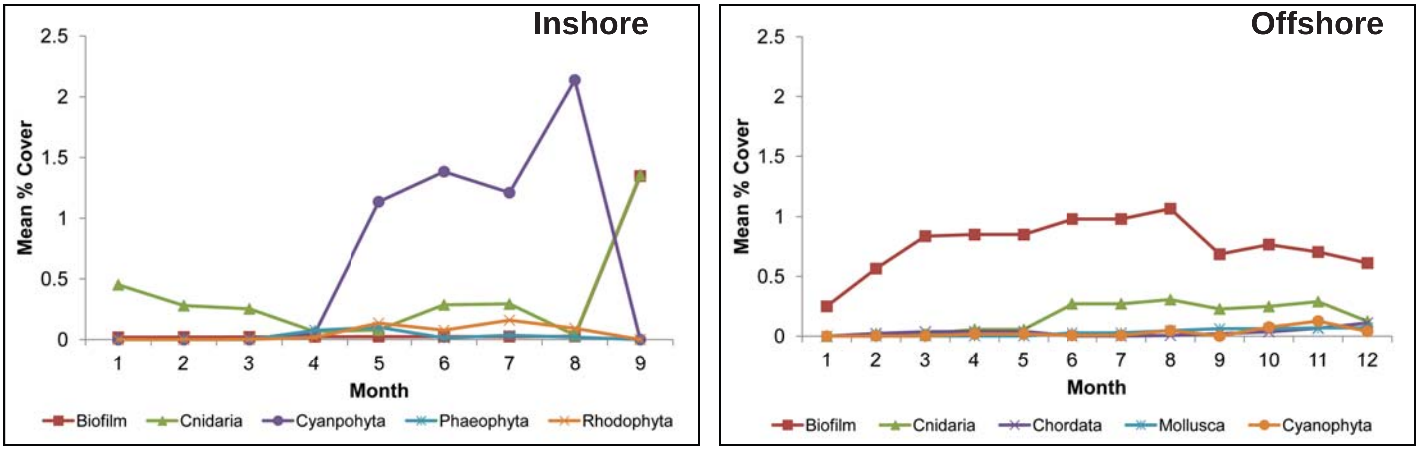

Figure 4.8. Inshore and offshore fouling communities on experimental traps with the highest percent cover growing over trap inner mesh.

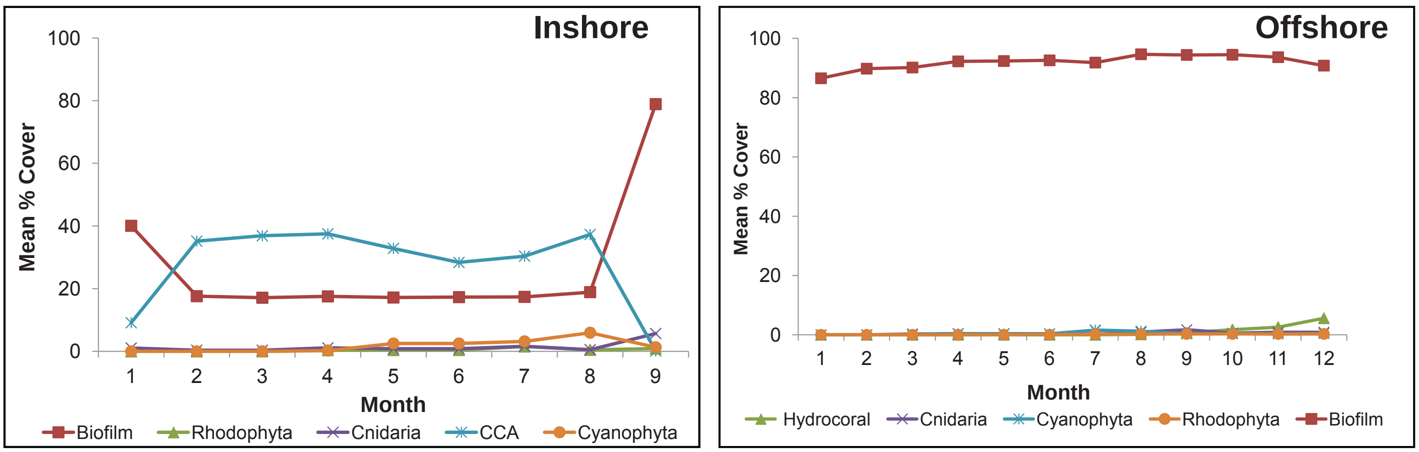

Figure 4.9. Inshore and offshore fouling communities on experimental traps with the highest percent cover growing over trap mesh.

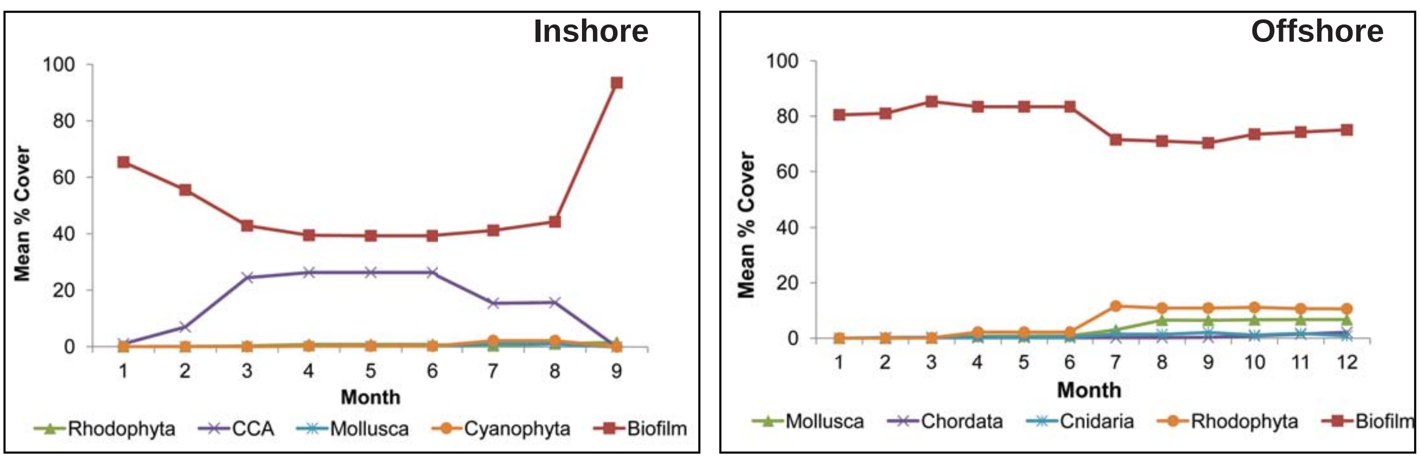

Figure 4.10. Inshore and offshore fouling communities on experimental traps with the highest percent cover growing on rebar supports.

Table 4.3. Analysis of similarities (ANOSIM) results comparing the percent cover of fouling community assemblages between locations and habitats. Asterisks indicate significant differences.

	Inshore vs. Offshore		Near-reef vs. Far-reef	
	Global R	p-value	Global R	p-value
Inner mesh	0.684	0.001*	0.035	0.001
Mesh	0.41	0.001*	0.033	0.001
Rebar	0.42	0.001*	0.095	0.001

phyla contributed approximately 77% to community differences growing in the rebar. The biofilm, Rhodophyta and Mollusca were more abundant offshore and contributed 27.3%, 13.7% and 9.2%, respectively. The CCA was again more abundant inshore contributing 26.8% to the differences.

The composition of fouling communities growing on the inner mesh area, mesh and rebar of the traps were not significantly different between near-reef and far-reef habitats as indicated by the low R-values and nMDS ordination plots (Table 4.3). The nMDS ordination plots (not shown) did provide a good representation of the data (stress values=0.13, 0.10 and 0.05 for the inner mesh, mesh and rebar, respectively), however a clear separation between habitats was not evident. Because of the high similarity, and lack of separation in fouling communities between the near-reef and far-reef habitats, a SIMPER analysis to determine what groups were driving the differences was not conducted.

The index of multivariate seriation values indicates how closely a community follows a sequential pattern of change. The IMS values for the inner mesh and mesh substrates were relatively low (closer to zero), but were much higher for the rebar. With the exception of the three inshore, near-reef traps (hurricane data excluded), all traps had a significant p-value associated with its IMS value; therefore, there was not a complete absence of seriation (Table 4.4). A distinct change in the fouling community composition can be seen in the nMDS ordination for the inshore near reef traps, the only three recovered from this site. The offshore trap that moved did not have a distinct change in community composition.

4.5. TRAP MOVEMENT

Trap movement measurements were compared monthly using a Wilcoxon test to test for differences in movement between inshore and offshore locations.

Movement was measured for four traps after the passage of Hurricane Earl through the territory on August 30-31, 2010 (Figure 4.11). The eye of the storm passed within 69 miles of the study site and wind speeds in excess of 65 mph and a storm surge of 1-3 ft were recorded (NHC, 2010; Gutro, 2010). Three traps located in shallow inshore waters (20 ft depth) moved distances of approximately 20 m, 133 m and 155 m from their original locations. Three other inshore traps moved and were not

Table 4.4. Index of Multivariate Seriation (IMS) values for each trap over the 12 month study period. Values in parentheses are the IMS and p-values that exclude data after traps were lost and recovered from Hurricane Earl.

Trap	Location	Habitat	Inner Mesh		Mesh		Rebar	
			IMS	p-value	IMS	p-value	IMS	p-value
1	Offshore	Near-reef	0.41	0.001	0.451	0.001	0.901	0.001
2	Offshore	Near-reef	0.52	0.001	0.474	0.001	0.88	0.001
3	Offshore	Near-reef	0.566	0.001	0.29	0.001	0.889	0.001
4	Offshore	Far-reef	0.279	0.001	0.284	0.001	0.787	0.001
5	Offshore	Far-reef	0.232	0.001	0.412	0.001	0.822	0.001
6	Offshore	Far-reef	0.33	0.001	0.42	0.001	0.71	0.001
7	Inshore	Near-reef	0.535 (0.676)	0.001 (0.001)	0.537 (0.543)	0.001 (0.001)	0.340 (0.568)	0.0044 (0.007)
8	Inshore	Near-reef	0.435 (0.296)	0.001 (0.001)	0.438 (0.352)	0.001 (0.001)	0.449 (0.447)	0.007 (0.025)
9	Inshore	Near-reef	0.246 (0.053)	0.001 (0.086)	0.445 (0.378)	0.001 (0.001)	0.581 (0.763)	0.001 (0.001)
10	Inshore	Far-reef	0.223	0.001	0.568	0.001	0.553	0.005
11	Inshore	Far-reef	0.226	0.002	0.537	0.001	0.585	0.002
12	Inshore	Far-reef	0.185	0.002	0.352	0.001	0.795	0.001

found. One offshore trap, located in 12 m of water, moved approximately 3 m. Visual surveys of the seafloor conducted between the original locations and the post-hurricane locations for both the inshore and offshore traps revealed no obvious trap related damage to the substratum in September 2010 (1-2 weeks after the hurricane had passed). Trap movement had not been detected prior to the hurricane.

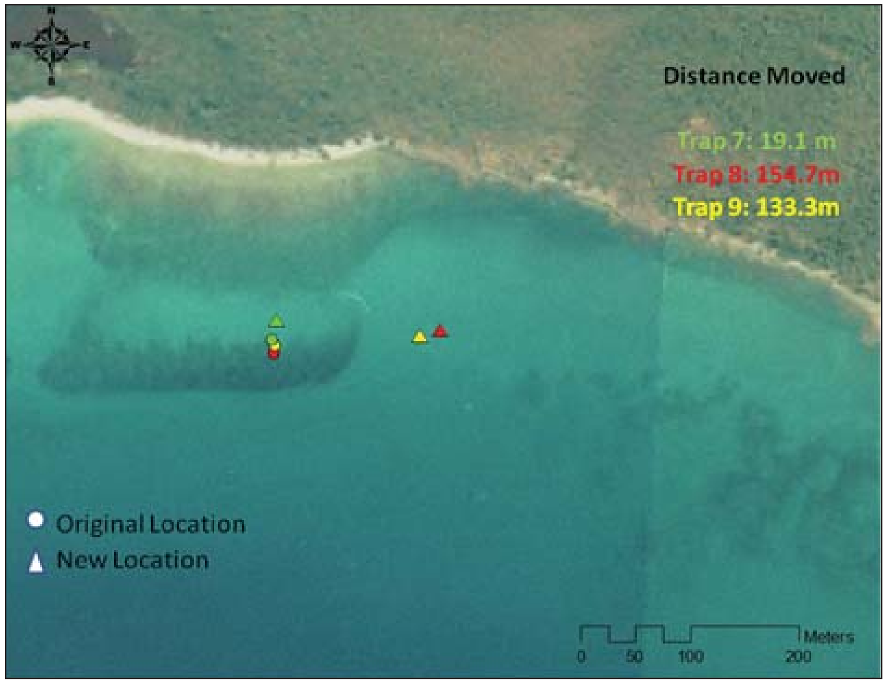

Figure 4.11. Trap movement as a result of Hurricane Earl, August 30-31, 2011.

4.6. DERELICT TRAPS AS FISH ATTRACTING DEVICES (FADS)

A total of 11,316 fish were observed within one meter on the outside of all traps. Only 85 fish were observed within one meter of the traps located offshore, while 11,231 were observed at the inshore traps. Approximately 97% of fish observed within one meter of the traps were located at the inshore seagrass bed. The Tomtate (*Haemulon aurolineatum*) was most abundant, accounting for 85% of all observed species. Spotted Goatfish (*Pseudupeneus maculatus*), Yellowtail Snapper, Striped Parrotfish and Bar Jack (*Carangoides ruber*) comprised the remainder of the top five species. Species documented at the seagrass traps were juveniles that slowly began to recruit to the traps during February and March of 2010, the second and third months of the study (Figure 4.12). The juveniles appeared to remain close to the traps as the cohorts were observed growing in size over the six months from post-settled juveniles to larger juveniles. During the recorded storm event, the trap rolled out of the area and this artificial habitat was lost.

4.7. CONCLUSIONS

Overall, fish mortality in actively-fishing traps, simulating ghost fishing, was unexpectedly low, with 95% of fish able to leave the traps unaided. Nevertheless, our experimental derelict fish traps did result in fish mortality, demonstrating that ghost fishing does occur with intact DFTs with escape panels closed. In contrast, when escape panels were open, prolonged entrapment and subsequent mortality was very rare (one mortality, barracuda, 107 cm TL). The mortality rates and costs/ trap are low on a per trap basis but can be extrapolated to the larger scale of the commercial fishery in St. Thomas. Expansion of mortality

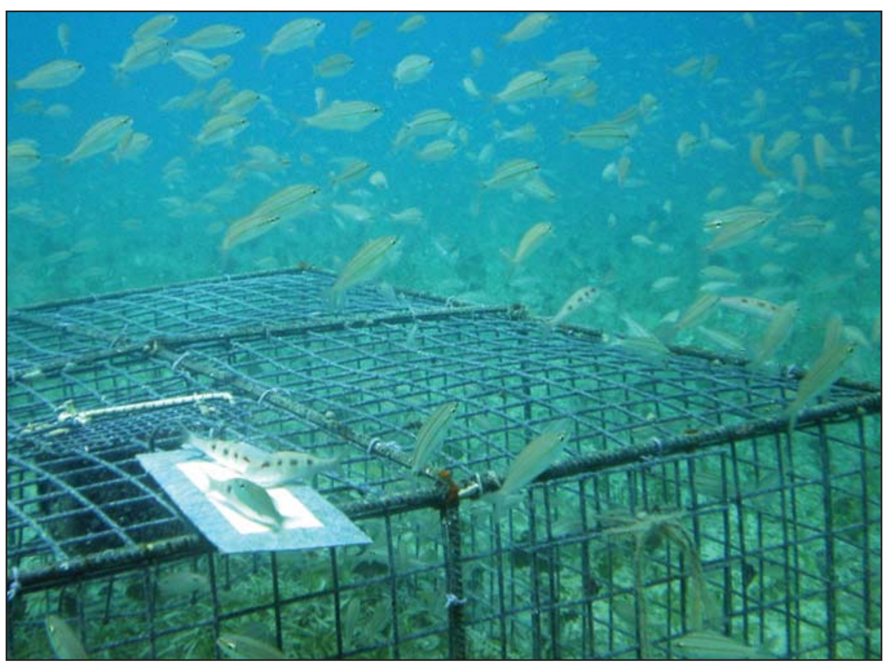

Figure 4.12. Juvenile grunts at an experimental trap site.

rates per trap per year suggests a loss of 184,931 lbs of marketable fish (50.5 lbs per trap x 3662 fish traps) or an economic loss of $191,340, annually. Earlier estimates of 8000-9000 traps operating in USVI waters (Sheridan et al., 2006) would suggest an even greater impact if these figures were representative. The average biomass/trap caught in our study, as well as the percent mortality, were approximately two to three times lower than estimates from the only other comparable study, a bycatch study conducted by the STFA which indicated an average biomass/trap of 1 kg and 9% mortality (Olsen, 2008). The fact that our research was restricted to shallow nearshore waters probably accounts for some of the differences between catch rates and levels of mortality and might require careful consideration of cost and impact extrapolations. Future research should target deeper waters where the commercial industry appears to be focusing their effort.

Over the one year study period, traps showed little to no signs of degradation. The rebar frames were observed rusting after one month but did not break down. The inshore traps that were moved as a result of Hurricane Earl did incur some damage, with the wire mesh bending, but they were all still intact and capable of fishing. These annualized extrapolations assume that traps remain relatively intact over a period of many months. Little is known, however, about the condition of genuine DFTs, although it is likely that they exist in a range of conditions from intact and newly lost to damaged, eroded and intentionally discarded. Further research should investigate the true life span of derelict traps in marine environments.

Anecdotal information from local fishermen indicated that traps no longer usable are typically discarded at sea with escape doors opened or removed to minimize risk of ghost fishing. We observed several accumulations of traps in remotely operated vehicle (ROV) surveys that appeared to be disposals (see Chapter 3). Our experimental results suggest that this practice will almost entirely eliminate the risk of ghost fishing in derelict fish traps, although the presence of intact mesh may still, albeit rarely, result in mortality and sub-lethal physical damage due to fish panic behavior. It further assumes that discarded traps do not land with the escape panel blocked in any way. Behavioral observations indicated that a majority of the fish caught in the traps attempted to escape by colliding into the mesh, often leaving bruises and abrasions on the area between the eyes and snout on the fish. Species that experienced stress and or injuries while trapped may have experienced post-escapement mortality due to infection from abrasions (Al-Masroori et al., 2004; Bullimore et al., 2001), although this was not evaluated during our study.

In a shorter duration study, Munro (1971) documented 50% escapement after traps had soaked for 14 days. Munro (1974) also suggested opposing dynamics: gradually more fish escape during each successive soak day, while at the same time, fish of the same species (conspecifics) may attract additional fish into traps. We observed fish of the family Acanthuridae, Sparidae, Ostraciidae and Haemulidae interacting with conspecifics in experimental DFTs when exhibiting schooling behavior inside the traps, entering and exiting trap,s and swimming alongside an untrapped conspecific on the outside of the trap. Similar behavior has also been extensively examined by Luckhurst and Ward (1987).

Our offshore traps had higher species richness than traps set inshore, but mean total length of fish caught was slightly smaller than those caught inshore. Environmental differences such as distance from shore, depth and vertical relief between locations, may contribute to differences in what species were caught and how they behave (Brokovich et al., 2006). Smith et al. (2008) documented that reef complexes in the USVI are influenced by their distance from shore; the mid-shelf island of Flat Cay was found to have lower sedimentation rates, higher coral cover and overall better coral health than the nearshore site of Perseverance Bay. Residence time differences of fish caught in near-reef and far-reef habitats may be due to the difference in habitat structure and reef complexity as well as the catch composition. Certain species may be more accustomed to navigating reefs and swimming through crevices and therefore may enter traps more than other species but spend less time in the traps (e.g., squirrelfish, surgeonfish; Robichaud et al., 2000). Traps may provide a temporary structural refuge that is otherwise lacking in areas near reef habitats (Wolff et al., 1999). Interestingly, the average number of consecutive days fish spent in traps during this study corresponds closely with the reported average soak time of seven days utilized by local fishermen in the USVI (D. Olsen, STFA unpub. data).

Some fish species appeared to take up residency in the traps and were recorded on subsequent sampling dates over periods of almost three months before escaping or expiring. On return surveys, fish were identified from photographs and size estimates. This may have led to some confounding residence times estimates, particularly where conspecifics had few distinguishing features. Future studies could use a mark or tag to provide greater certainty in identification of individuals (Bullimore et al., 2001). Twenty four hour video surveys documented 14 distinct behaviors of trapped fish (Renchen et al., in review). Overall, fish allocated the majority (78%) of their time attempting to escape by butting in the trap mesh or corners, in some cases resulting in injury. Of the remaining 22%, half were relatively sedentary, 8.9% were observed grazing and 1.2% were entering and exiting the trap.

As expected, fouling communities were significantly different between habitats and locations. The phylum Cnidaria was more abundant growing on the shallower inshore traps than any other habitat or location. Hydroids were the major component of this phylum, which have been documented in several studies as having decreased abundance with depth, thus accounting for their lower presence at the deeper offshore site (Hobbs and Azadan, 2010). Hobbs and Azadan (2010) also documented that substrate or bottom type did not present a clear pattern to discern differences in fouling community patterns. Colonization by fouling organisms is structured by physical factors, such as currents, water depth, distance to shore and water quality (Svane and Petersen, 2001), and also by biological factors, such as larval availability, recruitment and survival. Time of immersion, for instance during different seasons, can cause variations in fouling community colonization due to the seasonality associated with larval availability and recruitment (Saldhana et al., 2003; Fitzhardinge and Bailey-Brock, 1989).

Patterns in fouling community structure were present, albeit weakly, and it is likely that they were the result of seasonal changes of taxonomic groups rather than community succession (e.g., cnidarians); however, the duration of this study was not long enough to determine if the changes in fouling organisms were in fact due to changes in season or random events. Patterns of emergence of the calcifying groups tended to occur during approximately the same month for each group (e.g., tube worms, hydrocoral, and stony coral). Substrate attached organisms that calcify may serve as better indicators of trap age as they are not grazed and their size can be monitored through time. Baseline data collected by Saldhana et al. (2003) documented that the maximum shell size of bivalves could be used to estimate the age of derelict fishing nets. Although the process of fouling community development is complex, in most studies, the initial steps of establishment and development follow the same basic pattern regardless of location and substrate type, known as the sequence fouling model developed by Wahl (1989). Once initial settlement has occurred, further sequencing of the fouling community is difficult due to the trouble in distinguishing between true succession and seasonal progression of the communities (Scheer, 1945). The three inshore traps that were lost and recovered after the passage of Hurricane Earl experienced dramatic changes in fouling community composition. The traps likely rolled several times; the two were found upside down. This disturbance opened up space for new fouling organisms as the communities growing prior to the hurricane were much different (Cifuentes et al., 2007).

The fouling communities growing on the experimental derelict traps may also have been altered by grazing herbivores. Herbivores, such as Striped Parrotfish, Princess Parrotfish, Doctorfish and Blue Tang, were frequently observed grazing on traps, especially at the inshore site. Herbivores have the ability to suppress the growth of fleshy algal forms and alter fouling succession (Belliveau, 2002; Burkepile and Hay, 2006), and grazing may have facilitated greater percent cover of crustose coralline algae on the inshore traps. Gut content analysis were not done to verify herbivore diet but one can speculate that grazers influenced the pattern of limited macroalgal growth and greater percent cover of CCA on the inshore traps. More information will need to be gathered to support this observation.

In order to accurately estimate the age of a derelict fish trap, more than just the percent cover of fouling organisms needs to be assessed. Calcifying organisms should be sampled and sectioned to determine their age, which can be used as a proxy for the trap age. The fouling organisms recorded were those that could only be observed with the naked eye, and it is likely that many more were present.

Damage to sensitive habitats such as coral reefs and seagrass beds, is often assumed with derelict fishing gear due to gear movement and entanglement, although many existing studies have focused primarily on nets and fishing line (Matsuoka et al., 2005; Pawson, 2003). Lewis et al. (2009) examined the effects of lobster traps moving under the influence of storm conditions and found significant damage to benthic organisms along the path of movement. In the U.S. Virgin Islands, we demonstrated that hurricane storm conditions are capable of moving DFTs over large distances, presumably through a rolling motion due to high wave action, although the exact path could not be discerned as no scarring or other habitat damage was observed. Shallow traps moved considerably farther than deeper traps suggesting that traps in deeper water are less affected by wave energy and are more stable. While this finding was highly variable, it is supported by sightings of heavily fouled DFTs in deeper water colonized by a high diversity of coral reef organisms, including scleractinian corals and sponges.

Commercial trap fishermen set traps in a range of habitat types, including algal plains, low-relief pavement and sand areas close to reef slopes. These targeted habitats are usually in waters deeper than 16 m, where trap movement is less likely to occur, although Lewis et al. (2009) found that the much heavier Florida lobster traps would move when winds exceeded 15 knots for at least two days. Traps set at 12 m deep moved under these or more turbulent conditions so moderate depth is certainly no guarantee against trap movement. In the St. Thomas trap fishery, many traps were reported as lost over the past two years, indicating that traps in deeper water are susceptible to movements either by storms, large ground swells, or snagging and dragging. Many unregistered traps are set in nearshore shallow water where they are much more likely to be mobilized by lesser storms or major swell events, and therefore more likely to impact sensitive habitat structure.

Responsible management of fisheries impacts is essential to sustainable fisheries and the maintenance of fisheries livelihoods. In the Caribbean, high uncertainty exists regarding the impacts of derelict fish traps due to the lack of targeted scientific studies. Using controlled field experiments we have provided quantitative data on mortality associated with derelict fish traps in the USVI, as well as information on a wide range of associated behavioral interactions. Reliable impact assessment is required to inform fishing communities and management agencies to help prioritize actions that may include design modifications to gears and fishing practices and mitigation actions such as trap disposal programs.

4.8. REFERENCES

Al-Masroori, H., H. Al-Oufi, J.L. McIlwain, and E. McLean. 2004. Catches of lost fish traps (ghost fishing) from fishing grounds near Muscat, Sultanate of Oman. Fisheries Research 69 (3), 407-414.

Appeldoorn, R.S., M. Nemeth, J. Vasslides, and M. Scharer. 2000. The effect of fish traps on benthic habitats off La Parguera, Puerto Rico. Report to the Caribbean Fishery Management Council. Unpubl. MS. Department of Marine Sciences, University of Puerto Rico, Mayagüez, Puerto Rico. 33 pp.

Belliveau, S.A. and V.J. Paul. 2002. Effects of herbivory and nutrients on the early colonization of crustose coralline and fleshy algae. Marine Ecological Progress Series 232:105–114.

Brokovich, E., A. Baranes, and M. Goren. 2006. Habitat structure determines coral reef fish assemblages at the northern tip of the Red Sea. Ecological Indicators 6(3), 494-507.

Bullimore, B.A, P.B. Newman, M.J. Kaiser, S.E. Gilbert, and K.M. Lock. 2001. A study of catches in a fleet of ghost fishing pots. Fishery Bulletin 99:247-253.

Burkepile, D.E. and M.E. Hay. 2006. Herbivore vs. nutrient control of marine primary producers: context-dependent effects. Ecology 87:3128-3139.

Cifuentes, M., C. Kamlah, M. Thiel, M. Lenz, and M. Wahl. 2007. Effects of temporal variability of disturbance on the succession in marine fouling communities in northern-central Chile. Journal of Experimental Marine Biology and Ecology 352(2): 280-294.

Clarke, K.R., R.M. Warwick, and E.E. Brown. 1993. An index shoring breakdown of seriation, related to disturbance, in a coral-reef assemblage. Marine Ecology Progress Series 102:153-160.

Clarke, K.R. and R.M. Warwick. 2001. Change in marine communities: an approach to statistical analysis and interpretation, 2nd edition. PRIMER-E, Plymouth.

Evans, L.V. 1981. Marine algae and fouling; a review with particular reference to ship fouling. Botanica Marina. 24:167-171.

Fitzhardinge, R.C. and J.H. Bailey-Brock. 1989. Colonization of artificial reef materials by corals and other sessile organisms. Bulletin of Marine Science 44:567–579.

Froese, R. and D. Pauly. Editors. 2011. FishBase. World Wide Web electronic publication. www.fishbase.org, version (12/2011).

Gutro, R. 2010. Hurricane season 2010: Hurricane Earl (North Atlantic Ocean). NASA Goddard Space Flight Center, Greenbelt, MD. Online: http://www.nasa.gov/mission_pages/hurricanes/archives/2010/h2010_Earl.html (Accessed 9April 2012).

Hobbs, N. and C. Azadan. 2010. Marine invasive species benthic fouling study. Salem Sound Coastwatch Report. 25pp.

Holmes, N.J., V.J. Harriott, and S.A. Banks. 1997. Latitudinal variation in patterns of colonization of cryptic calcareous marine organisms. Marine Ecological Progress Series 155:103-113.

Kendall, M.S., C.R. Kruer, K.R. Buja, J.D. Christensen, M. Finkbeiner, and M.E. Monacao. 2001. Methods used to map the benthic habitats of Puerto Rico and the U.S. Virgin Islands. NOAA/NOS Biogeography Tech Report. Silver Spring, MD. 45 pp.

Lewis, C.F., S.L. Slade, K.E. Maxwell, and T.R. Matthews. 2009. Lobster trap impact on coral reefs: effects of wind driven trap movement. New Zealand of Marine and Freshwater Research 43:271-282.

Luckhurst, B. and J.W. Ward. 1987. Behavioral dynamics of coral reef fishes in Antillian fish traps at Bermuda. Proceedings of the Gulf and Caribbean Fisheries Institutue. 38:528-546.

Matsuoka, T., T. Nakashima, and N. Nagasawa. 2005. A review of ghost fishing: scientific approaches to evaluation and solutions. Fisheries Science 71(4), 691-702.

Munro, J.L. 1971. Dynamic factors affecting the performance of the Antillean fish trap. Proceedings of the Gulf and Caribbean Fisheries Institutue 23:184-194.

Munro. J.L. 1974. The mode of operation of Antillean fish traps and the relationships between ingress, escapement, catch and soak. Journal Conseil International pour l'Exploration de la Mer. 35(3):337-350.

National Hurricane Center (NHC). 2010. Hurricane Earl. Online: http://www.nhc.noaa.gov/archive/2010/al07/al072010.fstadv.019.shtml (Accessed 9 April 2012).

Olsen, D. St. Thomas Fishermen's Association. St. Thomas, U.S. Virgin Islands. Personal communication.

Olsen, D. 2008. Bycatch studies of reef fish in support of Caribbean Fishery Management Council preferred options for fishery management in St. Thomas, U.S. Virgin Islands. MARFIN- NA06NMF4330052. 26 pp.

Pawson, M.G. 2003. The catching capacity of lost static fishing gears: introduction. Fisheries Research 64 (2-3):101-105.

Quandt, A. 1999. Assessment of fish trap damage on coral reefs around St. Thomas, USVI. University of the Virgin Islands, St. Thomas, USVI. Unpublished Report, 11 pp.

Renchen, G.F. 2011. Assessing the ecological and economic impact of derelict fish traps in the U.S. Virgin Islands. MS Thesis, University of the Virgin Islands. 96 pp.

Renchen, G.F., S.J. Pittman, and M.E. Brandt. In press. Investigating the behavioral responses of trapped fish using underwater video surveillance. Journal of Fish Biology. 33 pp.

Robichaud, D., W. Hunte, and M.W. Chapman. 2000. Factors affecting the catchability of reef fishes in Antillean fish traps. Bulletin of Marine Science 67(2):831-844.

Saldanha, H.J., G. Sancho, M.N. Santos, E. Puente, M.B. Gaspar, A. Bilbao, C.C. Monteiro, E. Gomez, and L. Arregi. 2003. The use of biofouling for ageing lost nets: a case study. Fisheries Research 64:141-150.

Scheer, B.T. 1945. The development of marine fouling communities. Biological Bulletin 89(1):103-121.

Sheridan, P., R. Hill, G. Matthews, R. Appeldoorn, B. Kojis, and T. Matthews. 2005. Does trap fishing impact coral reef ecosystems? An update. Proceedings of the Gulf and Caribbean Fisheries Institute 56:511-519.

Sheridan, P., R. Hill and B. Kojis. 2006. Trap fishing in the U.S. Virgin Islands: how and where effort is exerted. Proceedings of the Gulf and Caribbean Fisheries Institute 57:175-188.

Smith, T.B., R.S. Nemeth, J. Blondeau, J.M. Calnan, E. Kadison, and S. Herzlieb. 2008. Assessing coral reef health across onshore to offshore stress gradients in the U.S. Virgin Islands. Marine Pollution Bulletin 56(12):1983-1991.

Svane, I. and J.K. Petersen. 2001. On the problems of epibioses, fouling and artificial reefs, a review. Marine Ecology 22(3):169-188.

Wahl, M. 1989. Marine epibiosis. I. Fouling and antifouling: some basic aspects. Marine Ecology Progress Series 58:175-189.

Warwick, R.M., C.M. Ashman, A.R. Brown, K.R. Clarke, B. Dowell, B. Hart, R.E. Lewis, N. Shillabeer, P.J. Somerfield, and J.F. Tapp. 2002. Inter-annual changes in biodiversity and community structure of the macrobenthos in Tees Bay and the Tees estuary, UK, associated with local and regional environmental events. Marine Ecology Progress Series 234: 1-13.

Wolff, N., R. Grober-Dunsmore, C.S. Rogers, and J. Beets. 1999. Management implications of fish trap effectiveness in adjacent coral reef and gorgonian habitats. Environmental Biology of Fishes 55:81-90.

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Chapter 5: Summary and Recommendations

Randy Clark[1]

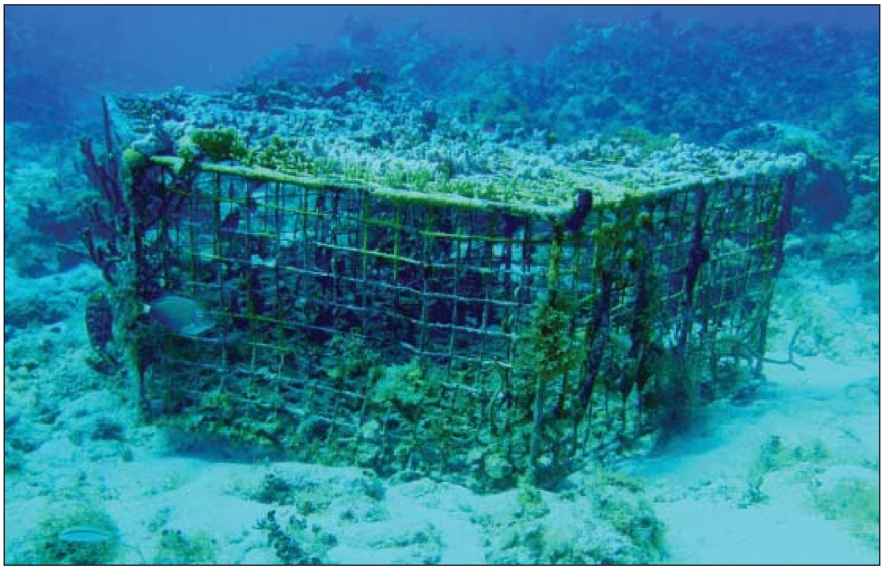

Figure 5.1. Derelict fish trap on seafloor. Photo: NOAA/NCCOS/CCMA Biogeography Branch.

Disposal of old traps at sea is a common practice in the Caribbean commercial trap fisheries industry (Figure 5.1). Based on the St. Thomas Fishermen's Association's (STFA) data, ten percent of all fish and lobster traps used each year become derelict and ten percent of those traps are from fishermen discarding them. Considering that there are 6,500 traps (estimate from 2009/2010) used in the St. Thomas and St. John fishery, there is a potential for the accumulation of approximately 13,000 traps on the seafloor over a 20 year period. This estimate does have a degree of uncertainty due to a lack of reliable data on trap use, dereliction rates, and disposal behavior. In addition, the number of derelict traps contributed by unlicensed subsistence fishermen is uncertain because there is limited data on the quantity of unregistered traps in the U.S. Virgin Islands (USVI) fishery. Possible actions that might reduce intentional dumping of traps include providing convenient land-based disposal locations and/or providing economic incentives for properly disposing of gear.

One of the primary concerns related to discarded traps is ghost fishing. The impact of ghost fishing is a function of trap age, condition, and particularly the functionality of escape panels. Federal and territorial regulations in the USVI require that fishermen have at least one escape panel that is tied shut using a biodegradable rot cord; however, compliance with these regulations and the life span of the rot cord and other commonly used materials need to be further investigated. Despite relatively low levels of mortality documented in this study, mitigation through trap disposal programs would reduce the occurrence of derelict fish traps (DFTs) and their capability to ghost fish.

There are multiple ways in which traps impact the marine fauna around the islands. Based on field experiments, DFTs can cause a five percent mortality rate. Extrapolation of this figure suggests a loss of 184,931 lbs of marketable fish (50.5 lbs per trap x 3,662 fish traps) which adds up to an economic loss of approximately $200,000 per year. This is likely an underestimate as the field experiments conducted here were in shallow waters which do not have the density of fish that would typically be found in deeper water locations (where the commercial fishermen focus their efforts). Additionally, DFTs cause mortality when fish die outside the trap after escape due to infection from abrasions sustained during entrapment. Also of importance, it has been observed that DTFs damage substrate and habitat when they move across the seafloor.

Theft and vandalism are major concerns in the region and may play a large role in the cause of trap dereliction. Attempts by fishermen to prevent theft and vandalism usually include setting traps without surface buoys. Most fishermen are adept at finding these traps, but combined with vandalism, this

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may be the leading cause of trap loss in the region. Voluntary trap reduction by fisherman is one possible solution. Recently, committees comprised of fishermen have agreed to reduce trap numbers by 20% over the next three years.

Shipping lanes are also a major concern and may have a large impact on lost traps, as fishermen may not mark traps on the surface to avoid ships/boats tangling with the line. The lack of proper lanes for cruise ships entering/exiting Charlotte Amalie in St. Thomas has been observed by fishermen as a factor that promotes the loss of traps. The region is active, especially with cruise ships visiting the ports of St. Thomas and the British Virgin Islands.

DFT density was low due to the small percentage of fishing grounds surveyed in the region. A more thorough survey is recommended to obtain a more comprehensive estimate of DFT density and to determine if these traps pose a biological threat. Using the U.S. Navy's Naval Surface Warfare Center autonomous underwater vehicle (NSWC AUV) program is recommended due to the cost effectiveness and the suite of technology that accompanies their AUV system.

Before implementing derelict trap removal programs, managers must determine the true overall impact of the traps and they need to consider the effectiveness of removing traps. As seen in Chapter 4, some derelict traps have enhanced the surrounding habitat resulting in an increase in biodiversity. Those situated in deeper water and covered with diverse sessile reef organisms are less likely to move and removal may result in a greater impact to seafloor structure. Compliance with both territorial and federal regulations, use of a proper biodegradable rot cord for escape panels, as well as disposal of worn-out traps on land instead of at sea could potentially reduce mortality and reduce marine debris.

FUTURE RESEARCH NEEDS

Many fishermen dispose of old or damaged traps at sea. Research needs to focus on disposal alternatives, including recycling or disposing of the traps in an environmentally friendly manner. Affordable, convenient and accountable land based disposal alternatives need to be explored.

Theft and vandalism were identified as major problems that are poorly quantified. These issues need to be examined more closely to determine if enforcement can play a role in reducing this problem.

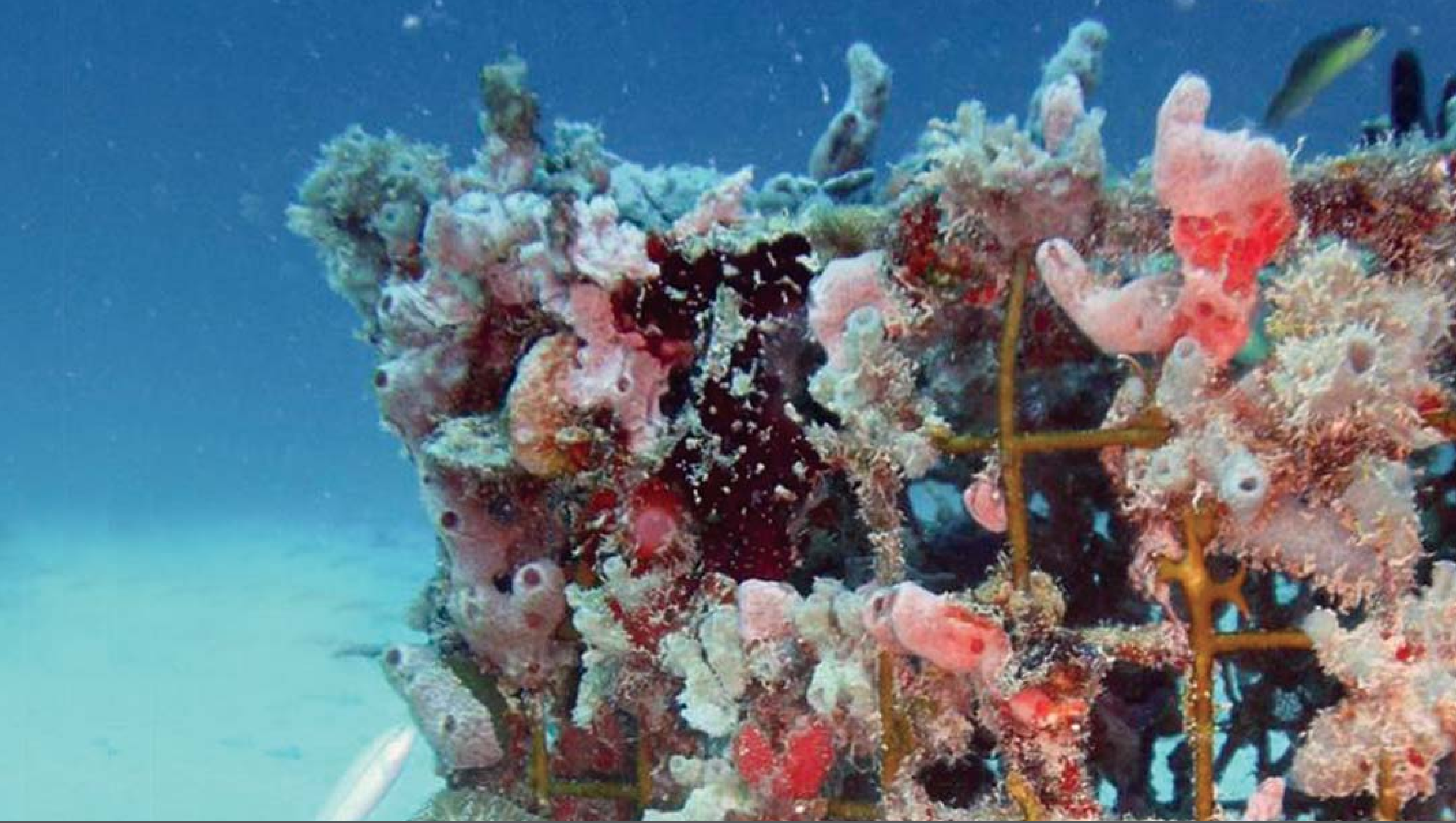
This project provides an initial estimate of DTFs, but no information currently exists on how many DFTs may be contributed by subsistence fishers that typically fish in waters that are less than 20 m in depth. In addition, the study only collected data for the 20-40 m depth habitats. To have a full understanding of the magnitude of the problem, additional fishing effort and derelict trap data needs to be collected in areas less than 20 m and greater than 40 m in depth. In conjunction with this, much needed shipping and boating information needs to be gathered and compared with the effort data to establish the optimal locations for shipping lanes or exclusive fishing areas.

A method was established to quantify derelict trap abundance in the region using AUVs. Additional surveys need to be conducted to provide more precise estimates of DFT abundance. The AUV techniques used here could serve multiple purposes including: searching for DFTs, examining areas where they tended to accumulate creating trap piles, mapping benthic features at a specified resolution, and searching for archaeological artifacts, etc.

The fate of derelict traps over the course of one year was examined. This time period was insufficient in length to answer the question of how long a trap maintains its structural integrity once it has gone derelict. Further research on the true life span of derelict traps in marine environments needs to take place. Derelict fish traps vary in condition from newly lost to damaged, eroded and intentionally discarded. Longer time periods should be examined to understand how the fouling communities grow and to understand how DFTs may or may not be incorporated into the surrounding natural habitat. If one wants to accurately estimate the age of a derelict trap it is recommended that calcifying organisms be sampled and sectioned to determine their age which can be used as a proxy for the trap age. This information is critical in the establishment of trap removal criteria. There are many concerns regarding trap removal that need to be addressed, including: what to do with traps once removed, which traps get removed, who will remove them, how much will it cost, and who will pay for the removal.

Longer term fish mortality studies need to be conducted to verify the initial mortality estimates. Experimental designs for these studies should also include a wider range of habitats.

This project has provided valuable insight on many issues surrounding the impacts of derelict fish traps in the study area. In many cases, this research is the first of its kind in the St. Thomas and St. John region. While questions remain, the initial studies conducted have laid the groundwork for future research activities that can help provide potential solutions to reduce the number and impacts of DFTs in the region.



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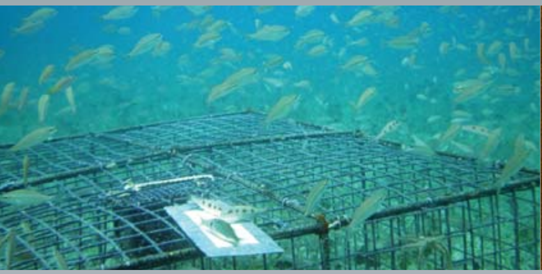

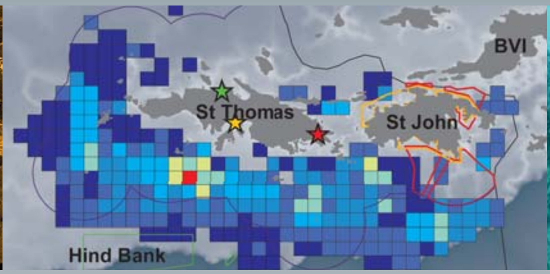

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