Evaluating the Effectiveness of Marine No-Take Reserves in St. Eustatius, Netherlands Antilles

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Abstract

Marine reserves, in which exploitation of marine organisms and other human activities are restricted or prohibited, can be valuable tools for marine conservation and fisheries management. No-take marine reserves support greater density, size, and diversity of fishes than exploited areas, with most benefits accruing to heavily exploited species. Protected areas can also export biomass to sustain or increase the overall yield of adjacent fisheries. These effects can be especially significant for organisms with small home ranges, such as coral reef fish. Although there are many benefits to creating marine reserves they are also costly to manage and can infringe on people's recreation and livelihoods. For marine reserves to be used as management tools, a comprehensive knowledge of the ecosystems is needed and reserve impacts should be regularly monitored.

St. Eustatius Marine Park and two no-take marine reserves were established by the Statian government in 1996 to promote dive tourism and sustainable fisheries. No fisheries study was carried out prior to reserve implementation to determine the status of fish stocks. In the summer of 2008, surveys were undertaken inside and outside the marine reserves to census reef fish and habitat to establish a current baseline and to evaluate the effects of the marine reserves on reef fish populations. The data that was collected was used to answer three questions about the current state of the Marine Park: 1) are there habitat related differences between reserve areas and non-reserve areas or between reef types? 2) do fish density, diversity, or biomass differ between protected and unprotected sites or between reef types? and 3) do habitat characteristics, reserve status, or reef type influence fish density, diversity, and biomass?

This study determined that habitat differences among the survey sites affected reef fish populations more than reserve status. However, the no-take reserves may still be important for protecting reef fish populations. The two reserves have been in place for more than 10 years and it is unknown what happened to fish populations during that time. Future research, employing the methods of this study, can be used to monitor reef fish populations and habitat over time to better determine if the reserves are having beneficial effects.

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Introduction

Traditional single species management has led to the collapse of many fisheries, prompting alternative management methods that protect both habitat and all the species that utilize that habitat. Marine reserves, in which exploitation of marine organisms and other human activities are restricted or prohibited, can be valuable tools for marine conservation and fisheries management. Partially protected areas can provide some advantages over open access areas, and no-take marine reserves can offer even greater benefits, including significantly higher densities of organisms within their boundaries relative to neighboring partially protected sites (Lester and Halpern 2008).

No-take marine reserves support greater size, and diversity of fishes than exploited areas (Roberts and Polunin 1991, Dugan and Davis 1993) with most benefits accruing to heavily exploited species (Gell and Roberts 2003). By limiting fishing activities within a reserve, protected areas have also been shown to export biomass to sustain or increase the overall yield of adjacent fisheries (DeMartini 1993). In addition increases in fish density and size inside a reserve area can enhance neighboring fisheries through the export of larvae ("recruitment effect") and emigration of post settlement individuals ("spillover effect") (Gell and Roberts 2003); moreover, increases in spawning stock biomass within a reserve may result in greater larval production and increased settlement both within and outside a reserve. If post-settlement fishes move across reserve boundaries, increases in fish density within a reserve will result in a net emigration to adjacent fished areas as a result of random movements and density dependent habitat selection (Roberts and Polunin 1991, DeMartini 1993, Rakitin and Kramer 1996). Spillover of reef fish from no-take areas into neighboring fisheries can reduce the effects of growth overfishing (Russ et al 1992). However, some studies have shown the spillover effect to be species specific and influenced by physical habitat barriers (Tupper 2007).

Increases in fish density and size inside a marine reserve are primarily seen in sedentary organisms that will spend much of their lives inside reserves, such as coral reef fish. Spillover into adjacent fisheries will only occur in species that have an intermediate tendency to relocate their home ranges because species with small home ranges and a weak tendency to relocate will never leave the reserve (Kramer and Chapman 1999). On the other hand, species with a high propensity to relocate will not remain in reserves long enough to reach higher densities or sizes. Many coral reef fishes have pelagic larvae that spend weeks to months in plankton, allowing them to disperse widely before they settle and form an association with a reef (Leis 1991, *in* Kramer and Chapman 1999). This process may carry larvae outside both the reserve area and adjacent fisheries, limiting the recruitment effect, and making it important to have a network of no-take areas in a region for benefits to accumulate.

Marine reserves should be designed to incorporate controls for spatial and temporal variations in fish distributions (Russ and Alcala 1996), but often reserves are implemented without information on fish distributions (Russ 2002). Instead, marine protected areas are often designed around areas with high habitat quality (Russ 1985, cited in Chapman and Kramer 1999). In the absence of the lack of baseline data, the most common measure of the effectiveness of a marine protected area is the difference in fish density between sites in a reserve and similar sites that are fished (Polunin and Roberts 1993). However, the results of such comparisons may be due to variation in habitat rather than the level of protection. The abundance of reef fishes has been correlated with substrate topographic complexity (Luckhurst and Luckhurst 1978) and coral cover (Bell and Galzin 1984), and individual reef fish species biomass has been correlated to a variety of habitat characteristics (Jennings et. al. 1996). At the community level, habitat complexity has been positively associated with fish diversity and total abundance (Gratwicke and Speight 2005) and biomass (Grigg 1994). Grigg (1994) found that increases in habitat complexity increased reef fish biomass in both fished and un-fished areas. Chapman and Kramer (1999) found correlations between density and size of fish with depth, complexity, and substrate composition of sites. After controlling for the effects of habitat correlates, the difference in total density between reserve sites and non-reserve sites was still significant (Chapman and Kramer 1999).

Artificial reefs have also been suggested as a management tool to provide additional habitat for marine resources and as a means of creating new fishing grounds when traditional fishing areas are incorporated into restricted reserves. An artificial reef is described as "any material or matter deliberately placed in an area of the marine environment where that structure does not exist under natural circumstances for the purpose of protecting, regenerating, concentrating or increasing populations of living marine resources, or for enhanced recreational use of the area" (Anonymous 2003 *in* Claudet and Pelletier 2004). Artificial reefs are commonly derelict ships (Arena et. al. 2007). The benefits of artificial reefs are numerous: 1) increase in fish biomass, by providing shelter from fishing, 2) increase in diversity by providing new habitats, 3) reduced negative impacts on existing habitats, 4) improves cost-effective fishing practices by displacing costs for fishermen, and 5) economic benefits through fisheries or diving activities (Claudet and Pelletier 2004). There are higher abundances, densities, and biomasses of fish close to artificial reefs compared to undisturbed control areas (Claudet and Pelletier 2004 and Arena et. al. 2007). These differences are probably due to higher structural complexity of artificial reefs. Artificial reefs may either increase fish abundance or simply act as fish aggregation devices, although one study suggests that artificial reefs actually increase abundance, because mean abundance and species richness were not significantly different between natural reefs neighboring sunken-vessels and natural reef with no close artificial reefs (Arena et. al. 2007).

There are many benefits to creating marine reserves, but they can be costly and are also perceived as infringing on people's recreation and livelihoods. For example, even if a marine reserve benefits fisheries in the long-term, local fishermen may be immediately affected by the loss of fishing grounds and increased travel time to open access areas. For marine reserve and artificial reefs to be used as management tools, a comprehensive knowledge of their ecosystems is needed and their impacts should be monitored regularly (Claudet and Pelletier 2004).

St. Eustatius Marine Park was established in 1996 and became actively managed in 1997 to conserve and protect the marine environment. The Marine Park is managed by the St. Eustatius National Parks (STENAPA), a non-governmental, not for profit organization ('stichting'), which was founded in the Netherlands Antilles in 1988. STENAPA is legally mandated by the Island Government to manage the island's protected areas, which includes the Marine Park (AB1996, No.03). STENAPA is charged with enforcing regulations relating to boundaries, diving, and fishing, carrying out research and monitoring, and informing and educating users (pers comm. Nicole Esteban, STENAPA, 2008).

Figure 1. St Eustatius, Netherlands Antilles. St. Eustatius Marine Park surrounds the island from the high tide line to the 30 m depth contour. The Northern Marine Reserve starts at 17º 30.5' N following the 30 m depth contour to the northern most land point. The Southern Marine Reserve begins at 17º 28.5' N to White Wall and then south out to sea for half a nautical mile, to the west following the 30 m depth contour up to 17º 27.7' N then to the northern coordinate of 17º 28.5' N and back to Gallows Bay (White et al, 2005).

Two no-take reserves, where fishing and anchoring are prohibited, were established inside the Marine Park in 1996, to "protect flora and fauna and to promote dive tourism" (AB1996 No.03) (Figure 1). A study of St. Eustatius' (hereafter Statia) marine environment in 1992 recommended the placement of the no-take areas based on protecting dive sites from fishing activities (Sybesma et. al. 1993). This study made observations about fish populations and habitat health, but no thorough baseline was established. In 2004 STENAPA completed a study that collected baseline data on fish stock populations within in the marine park and its no-take reserves (White et. al. 2006).

The original goal of the present study was to replicate the 2004 surveys and to examine changes in reef fish populations over time, thereby offering an assessment of the conservation benefits of the marine park. However, the present study revealed inadequacies and incompatibilities in the methodology used due in the 2004 study, rendering a fair comparison impossible. With this in mind, the current study aims to describe the "baseline" status of reef fish populations and habitat, and to communicate results and management recommendations concerning the use of marine reserves and artificial reef to policy-makers and stakeholders in-country.

In summer 2008, surveys were undertaken to census reef fish and habitat to determine whether the no-take reserves were having beneficial effects on fish populations. Reserve and non-reserve areas were further divided into four "reef types": Southern Reserve, Northern Reserve, General Use, and Artificial Reef. The objective of was to establish baseline data on reef fish populations and habitat and research methods so that these resources can be monitored over time. The data were used to answer three questions about the current state of the Marine Park:

1) are there habitat related differences between reserve areas and non-reserve areas or between reef types? 2) do fish density, diversity, or biomass differ between protected and unprotected sites or between reef types? and 3) do habitat characteristics, reserve status, or reef type influence fish density, diversity, and biomass? Based on these results, what management recommendations can be made about the use of marine reserves and artificial reefs?

History of the Marine Park in St. Eustatius, Regulations, and Enforcement

In 1996, a Marine Environment Ordinance was passed by the Island Council that arranged for the creation of the St. Eustatius Marine Park and legal management of that park by STENAPA. The Marine Park was defined as the sea floor and the overlying waters around and adjacent to the island of St. Eustatius from the high water tidemark to the 30 meter depth contour. The areas designated for the park and reserves were taken directly from recommendations in the Marine Area Survey conducted by Caribbean Research and Management of Biodiversity (CARMABI) in 1992. The surveys, which took place at 30 sites around the island, reported reefs to be healthy (minimal physical damage, no signs of stress, and minimal garbage) (Sybesma et. al. 1993). However, fish abundance was low and particularly noticeable was an absence of large predators such as groupers (Serranidae) and snappers (Lutjanidae). Large herbivorous parrotfish (Scaridae) were also few in number, which was unusual because parrotfish had filled the role of predominant algal grazer after the 1983 mass mortality of *Diadema antillarum* in other parts of the Caribbean. The lack of predators and large fish was believed to reflect overfishing (Sybesma et. al. 1993). Fishermen in Statia agreed that the reef fish fishery had declined and blamed habitat destruction from anchoring and oil dispersant pollution for the reduction in fish abundance (Sybesma et. al. 1993).

There were no fishing regulations prior to 1991 because the "fishing activities in the waters surrounding the islands were relatively limited in scope and there were no indicators that they would form a threat to the fish stock" (Knoppel 1991). Over the years, fishing activities increased and foreign fishing boats came into Netherlands Antilles waters. One reason for the increase in fishing activity was the implementation, by most countries in the Caribbean, of fisheries zones outside of the territorial sea to 200 nautical miles from the coast. In 1991, only Aruba and the Netherlands Antilles had not done so, which resulted in a movement of fishing activities to the waters of these countries (Knoppel 1991). Fishing regulations were also created to provide better information on fish stocks in Antillean waters for management purposes (Knoppel 1991). The fishing community on Statia is small, but the aggregated value of the fishing industry is important to the local island economy (Dilrosun 2004).

The fishing policies inside the Marine Park regulate the capture of fish, conch and lobsters, all of which are economically and socially important species to residents and tourists. Fishing is prohibited within the no-take reserves; an exception is made for Statian fishers who are going to or coming back from a fishing trip, who are allowed to troll a hand line through the reserves (AB1996 No.03). In the Marine Park, destructive fishing practices (poison, poisoned bait, chemicals, and explosives) are prohibited, as well as spear fishing on SCUBA or Hookah (AB1996 No.03).

Enforcement of the Marine Park regulations is mandated to STENAPA; patrols of the Marine Park take place once or twice weekly (STENAPA 2007). The patrols are conducted by Marine Park Rangers, employed by STENAPA and sworn in as Special Agents of Police (Buitengewoon Agenten van Politie) to uphold federal and island laws related to nature conservation (STENAPA 2007). In 2007 only six incidents required action from STENAPA staff, none of which involved fishing (STENAPA 2007). Most reported violations involved sand mining violations, oil emission from boats, yachts and cargo boats anchoring in the wrong areas, and misuse of dive sites by unpermitted organizations occurred (STENAPA 2004-2006). No action against fishermen was reported by STENAPA from 2004 to 2007. However, Park Rangers indicate that there has been some illegal fishing in the past by fishermen from St. Kitts as well as placement of fish traps inside the Reserves. Fish pots found inside Reserve boundaries are removed and confiscated from fishermen (pers. comm. Lee Munson, STENAPA, 2008).

User Conflicts in the Marine Park and Sources of Stress

The leeward shore of St. Eustatius, where most marine activities occur, is only about 10 km long. The main uses of this space are fishing, recreation (diving and snorkeling, beach recreation and swimming, and yachting), transportation and shipment (commercial harbor), and oil transshipment (terminal). There is only one dock in Oranjestad Bay to service most of these activities. The government of St. Eustatius wants to promote tourism, mainly through SCUBA diving; improve its harbor facilities; encourage the expansion of the oil terminal; and preserve its natural and cultural resources (Sybesma et. al. 1993). Often, both the user groups and the goals of the government come into conflict in this small marine environment.

Fisheries

There are fewer than 25 registered fishermen on St. Eustatius, all of whom use small boats of less than five meters (pers comm. Nicole Esteban, STENAPA, 2008). Most fishermen only fish part time and have other employment on the island. Their catch consists primarily of spiny lobster, reef fish, and queen conch. There are few fishing statistics available because most of the reef fish and conch is caught for local consumption and there is no processing plant or distribution center where lobster, fish or conch catch could be described and quantified (pers. comm. Nicole Esteban, STENAPA, and Joanna Spanner 2008). The lobster catch is the most important economically because it is exported to other islands (White et. al. 2006). When the lobster season ends fishermen target reef fish, principally employing fish traps, but hook and line, trolling, and SCUBA are also used (pers. observation 2008). The reef fish fishery does not bring much money into the economy, it is considered to be culturally significant (Sybesma et. al. 1993). Fishing grounds, where fish and lobster traps are placed, consists of shallow waters to about the 40 meter depth contour (Sybesum et. al. 1993). Prior to the creation of the Marine Park, fishermen reported placing traps in four areas around the island: 1) the Atlantic side from Bargine Bay to Compagnie Bay to depths of 25 m and up to one nautical mile from shore, 2) the area south of White Wall in 15 to 30 m depth, 3) the leeward side from Cocoluch Bay to Jenkin's Bay at depths of 30 m and up to one nautical mile from shore, 4) Oranje Bay in 25 to 35 m depth and Gallows Bay, south of the City Pier (Sybesma et. al. 1993) (See Figure 1). Since at least 1992, fishermen have been avoiding large predatory demersal fish, such as barracuda (*Sphyraena barracuda*), and large jacks (Carangidae), due to a bioaccumulating neurotoxin called ciguatera (Sybesma et. al. 1993).

Dilruson (2004) concluded that fishermen on the island have had a poor relationship with STENAPA due to the creation of the Marine Park. When the Park was established in 1996, there were no discussions with the fishermen about their use of the coastal area, important fishing grounds, or the economic or social importance of fishing. In addition, the number of fishermen has decreased over the years, in part due to the creation of the Marine Park no-take reserves, which fishermen have said were the best fishing grounds (White et. al. 2006). Now, fishermen complain that they must travel further, using more fuel, to set their lobster and fish traps (White et. al. 2006).

Recreation

There are three important recreational activities that use the marine park: the beach, yachting, and scuba /snorkeling. St. Eustatius has one beach along the coast of Oranjestad Bay. The beach was once very broad, but is now narrow and at times completely disappears. This beach is used mainly by locals for swimming and picnicking. The Atlantic side contains one large beach, Zeelandia, and many smaller beaches, but the high swells and currents discourage recreational use of these beaches. St. Eustatius is part of the yachting route along the Lesser Antilles island chain. There are 12 yacht moorings in Oranjestad harbor and 500 visiting yachts a year (STENAPA 2007). Because the harbor is unprotected, yachts spend comparatively little time in-country. St. Eustatius also has three foreign owned and operated dive shops, (Dive Statia, Golden Rock, and Scubaqua), that receive about 2,500 visitors annually (STENAPA 2007). The no-take reserves were created to benefit dive tourism (AB1996 No.03), as nearly all of the dive sites fall inside these protected areas leading to some conflict between fishermen and the diving operations. Fishermen perceive that foreign dive operators are benefiting from the Marine Park to the detriment of local fishermen. In the past, fishermen have accused dive shops of removing fish traps and freeing fish from traps, because tourist do not like to see caged fish. There is also a conflict of space in general use areas of the Marine Park where artificial reefs are located. Three artificial reefs have been created in the past ten years from derelict boats for the purpose of attracting fish to the benefit of the fishermen. Two of these reefs have become popular dive sites, which upsets some local fishermen (pers. obs. 2008).

Transportation and Shipment

The island of St. Eustatius is dependent upon imports for construction material, food, and other goods. These are mainly shipped from neighboring St. Maarten to the commercial harbor in Oranjestad Bay. Small boats bring fruit and vegetables from nearby islands, larger cargo boats (up to 175 ft long), fishing boats, and tugboats all share the harbor pier for unloading cargo and docking (St. Eustatius Government 2006). The current harbor dock, built in 1993, extends 80 meters from shore and then runs parallel to shore for 60 meters. The government plans to extend the dock 165 meters from shore, to dredge to a depth of 5.5 meters, and to accommodate vessels up to 100 meters in length. This would establish a long breakwater to the north of the harbor that could provide shelter for up to 60 yachts (St. Eustatius Government 2006). As it is, there is a great deal of boating traffic around the harbor area, which overlaps with important dive sites. The two most frequented dive sites are inside the shipping lanes (STENAPA 2004-2007), and often boats ignore, or come dangerously close to marked dive sites.

Oil Terminal

The main commercial activity is based around Statia Terminals N.V., which began operation in 1982 (Figure 2). It is a for-hire bulk liquid terminal that engages in third-party storage and handling, as well as a transshipment point for oil en route from the Middle East to the United States (White et. al. 2007). The terminal has 50 tanks for different petroleum products with a total capacity of 11.5 million barrels (White et. al.

2007). The petroleum products are taken as cargo by major tankers to the terminal dock. The average number of ship movements to the Terminal is 2.5 per day (pers. comm. anon. Statia Terminal employee 2008), with up to 1000 tankers docking each year (White et. al. 2007). The Terminal has one large jetty for two tankers and three floating docks for cargo shipments. Floating hoses are connected to the floating dock and connect vessels to transshipment gasoline, butane, and other highly flammable liquids (Sybesma et. al. 1993). The Terminal also operates a number of tugboats, utility boats, and work barges that move from the terminal dock to the dock at Oranjestad Bay. There is a restricted zone for single point mooring for very large tankers that overlaps with the Northern Reserve as well as three designated anchorage zones for smaller vessels (White et. al. 2007). The terminal also uses an area west of the designated zone for anchoring.

Figure 2. Aerial photograph of Statia Terminals on the northeast shore of St. Eustatius, showing holding tanks, terminal jetty, and cargo tankers.

The terminal is vital for the economic stability of the island, but it causes conflict with other users and may have an adverse effect on natural resources. The use of the undesignated anchorage area has caused problems between terminal operations and fishermen, as this is an important fishing ground. When tankers move through this area, they accidentally cut lines of fishing traps, creating “ghost traps” that continue to collect fish (Sybesma et. al. 1993). Anchoring in this area is also detrimental to the fishing and diving community because anchoring breaks up coral reefs. Although anchor damage is the main problem, studies also cite pollution caused by ballast water and a high risk of oil spills (White et. al. 2007). Statia is a transshipment point into the United States because Statia will accept petroleum products from single hulled vessels, which cannot come into the U.S., and transfer them to doubled hulled ships (White et. al. 2007).

Oil spill contingency plans exist as part of the Cartagena Convention, but at least three oil spills have occurred, in 1992 (Dahlin et al 1994), 2002 (Environmental Department 2004) and 2004 (STENAPA 2004). The 2002 spill occurred overnight as a tanker was pumping out ballast water contaminated with oil, which was not noticed until the next day when it had spread along the leeward sided of the island (Buth et al 2002). In 2004, volunteers cleaned up oil from a spill originating at the Terminal on a beach located on the opposite side of the island. No documentation exists on the extent of damage to the marine life after any of these events.

Methods

Study Area

This study was conducted around the island of Saint Eustatius (Statia), Netherlands Antilles (17º 20’ N; 63º 00’ W) from May 20 to June 23, 2008 (Figure 3). Statia, located in the northern Leeward Islands portion of the West Indies, is one of five islands that make up the Netherlands Antilles. The island sits on a submerged platform shared with St. Kitts and Nevis to the east. The 21 km^2 island is composed of two extinct volcanoes, an older (late Pliocene), heavily eroded volcano in the northwest and a younger volcano (Pleistocene) to the south (Adey and Burke, 1976).

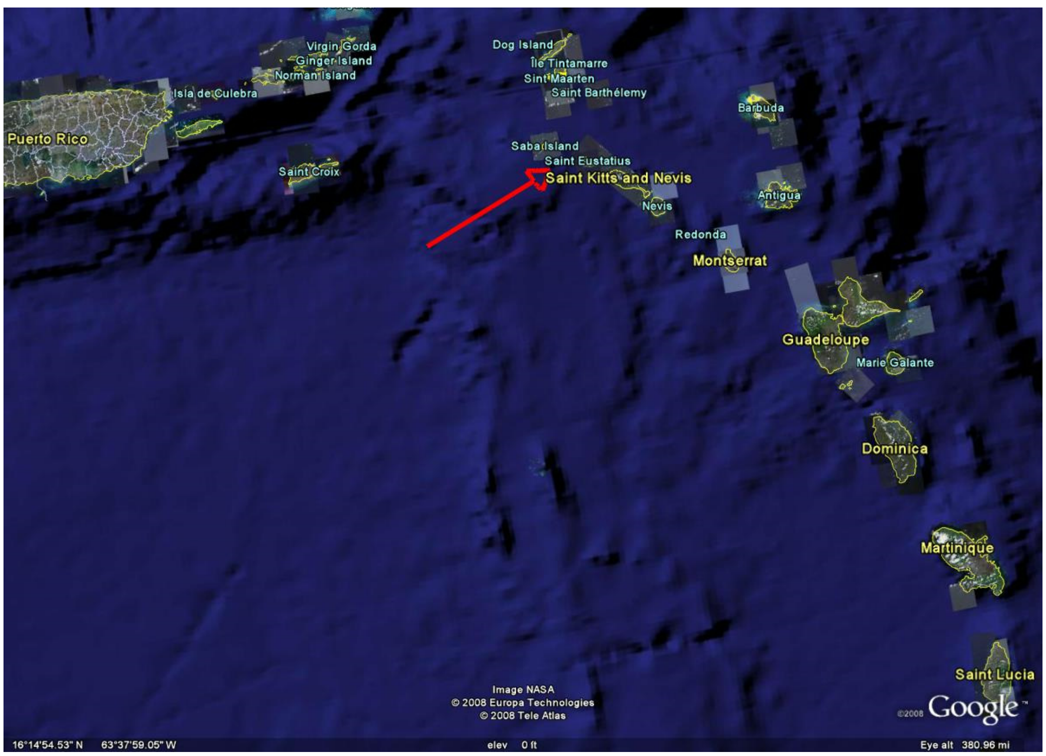


Figure 3. Location of St. Eustatius, Netherlands Antilles in the Southeastern Caribbean.

The island is surrounded by a narrow platform, between 22 and 30 m deep (Adey and Burke 1976). Fringing reefs are located mid-shelf off the southern and northwestern coasts. Lava flow formations from the southern volcano extend seaward, combined with large volcanic boulders, provide a base for coral growth that is 3 to 4 m in relief. A series of reefs in the south are about 30-50 m wide and are separated by sand channels of about 50 m. The highest coral cover occur med-shelf at the southern end of the island, where the underlying substrata is carbonate. These reef formations have vertical relief of 3 to 4 m and are between 30 to 100 m in width, but they do not exhibit the same orientations as the lava flow formations. In the north, coral fringes the rocky coast near shore, where dead *Acropora palmata* and live *Millepora* grow atop boulders in high surge zones. Farther from shore, coral heads are found along the reef flat. The remainder of the shelf is a flat, sandy plateau, with scattered small coral heads, soft corals, and sponges (site description adapted from Klomp and Kooistra 2003).

All surveys took place within the Marine Park, which surrounds the island from the high water line to up to and including the 30 m depth contour. The Marine Park consists of a General Use zone (22.61 km^2) and two no-take reserves (White et al, 2006). The Northern Reserve (1.61 km^2) encloses Jenkins Bay up to the northern most point of the island while the Southern Reserve (3.29 km^2) runs from Gallows Bay to White Wall (White et al, 2006).

A total of 17 reefs were surveyed in the study; five inside the Southern Reserve, three inside the Northern Reserve, five in the General Use areas, and four at Artificial Reefs (Table 1). The survey sites were selected based on two previous fish surveys

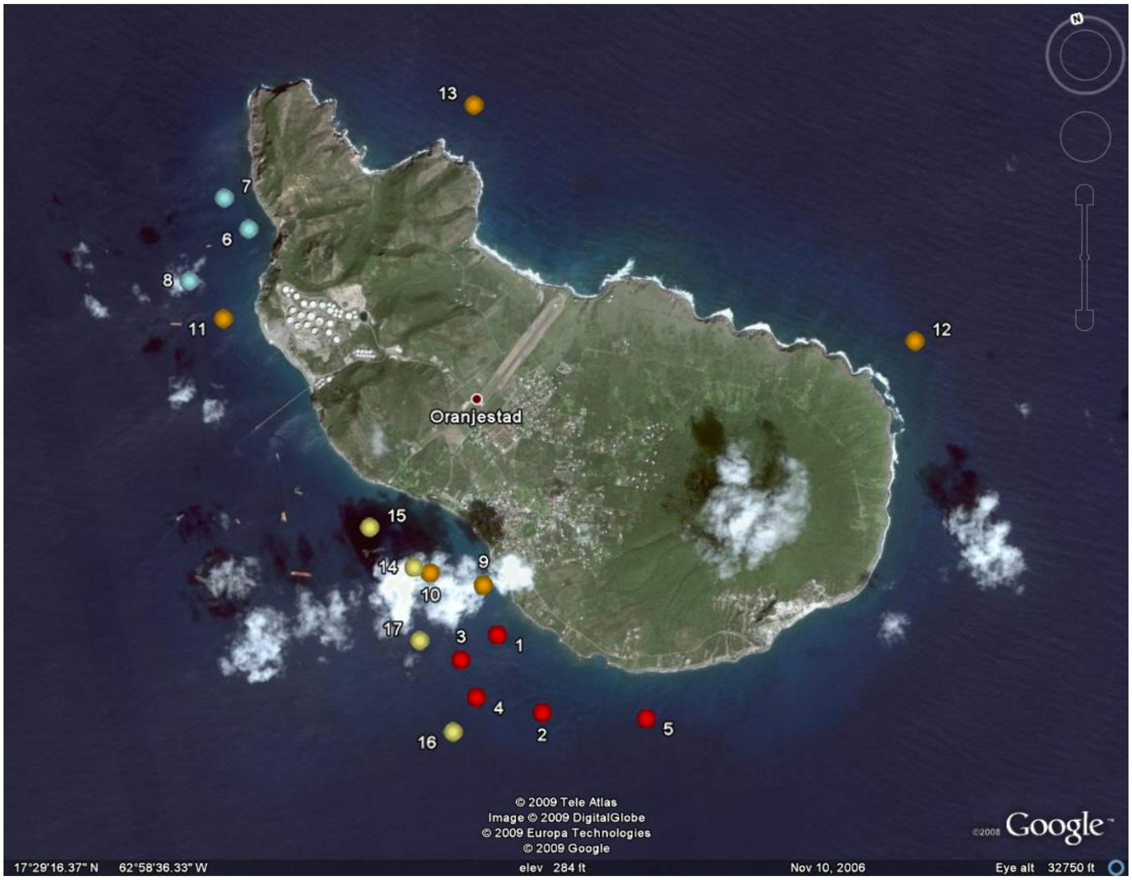


Figure 4. Locations of reef fish surveys: Red = Southern Reserve: 1 = Crook's Castle, 2 = The Blocks, 3 = Blair's Reefs, 4 = Barracuda Reef, 5 = Drop Off.; Blue = Northern Reserve: 6 = Jenkin's Bay, 7 = Twin Sisters, 8 = Doobie's Crack; Orange = General Use: 9 = South of Harbor, 10 = Sting Ray City, 11 = Aquarium, 12 = Corre Corre, 13 = Venus Bay; Yellow = Artificial Reef: 14 = Double Wreck, 15 = STENAPA Reef, 16 = Charles L. Brown, 17 = Miss Cathy.

(van't Hof et. al. 1993 and White et. al. 2005), but some changes to survey areas were necessary. An additional artificial reef site, Miss Cathy, was selected because it was sunk after the last fishery study was completed. Three of the sites first surveyed in 2004 (Venus Bay, Corre Corre, South of Harbor) did not have GPS locations, so new sites within these areas were surveyed. Based on the 2004 data and discussions with White, it seems the reef at Sting Ray City was not actually found during the 2004 surveys and a seagrass/sandy area was surveyed instead. This small reef was located and included in the present survey.

Table 1. Seventeen sites were surveyed in 2008 from four different "reef types" listed here. Sites that were surveyed in 1992 and 2004 are indicated, as well as current fishing activity, and site depths.

Reef Type	Site	1992	2004	Fishing	Depth (m)	> 15 m	15-19 m	20-30 m
Southern Reserve	Crook's Castle	Y	Y	Y	11	X		
	The Blocks	Y	Y		17		X	
	Blair's Reef	Y	Y		19		X	
	Barracuda Reef	Y	Y		23			X
	Drop Off	Y	Y		29			X
Northern Reserve	Jenkin's Bay	Y	Y		12	X		
	Twin Sisters	Y			18		X	
	Doobie's Crack		Y		29			X
General Use	South Harbor			Y	10	X		
	Sting Ray City	Y			16		X	
	Aquarium		Y		20			X
	Corre Corre	Y		Y	22			X
	Venus Bay	Y		Y	18		X	
Artificial Reef	Double Wreck	Y	Y	Y	17		X	
	STENAPA		Y	Y	17		X	
	Charles Brown		Y		30			X
	Miss Cathy			Y	21			X

Fish Density and Size

The fish survey conducted in July of 2004 consisted of three 20 meter by 5 meter transects at each site. The transects were laid out randomly at least 10 to 20 meters apart. Two scuba divers swam at a slow speed along the transect at the same time, recording fish encountered within 2.5 meters on either side and 5 meters above the transect. One diver counted all fish species present within the transect while the second diver estimated lengths of specific target species (goatfish (Mullidae), grouper (Serranidae), jacks (Carangidae), snappers (Lutjanidae), and wrasse (Labridae)). The size of the fish was estimated by total length in the following classes: > 5 cm, 5-9, 10-14, 15-19, 20-24, 25-39, 30-34, 35-40, 40-49, 50-59, 60-79, and 80+. The fish observers were trained in length estimation using fiddle-sticks, a technique described in English et al (1997).

Unfortunately, problems were discovered with this method when reviewing the results of the 2004 study. The problems result from having two scuba divers doing two types of fish counts simultaneously. The counts made by the two divers for groupers, snappers, wrasse, jacks, and goatfish were compared, and significant differences in reported numbers were found. Rarely did the counts from the two observers match and there are instances, in the case of wrasse, where counts were off by hundreds of individuals. This may indicate that the two divers were not identifying fish correctly, that the 5 meter belt was being misjudged, or that they were working at different speeds. Many reef fish move in and out of the belt transect, which could lead to different fish counts if divers were not looking at the same area at the same time. However, some fish, such as the blueheaded wrasse, *Thalassoma bifaciatum,* have very small home ranges, so counts for these types of reef fish should be very similar. This method resulted in high levels of error, so a new approach was adopted.

Before the 2008 surveys were begun, preliminary dives were carried out to establish a list of fish species present, to practice fish and substrate identifications, and to estimate fish sizes. The number and length of surveys was increased to four 30m belt transects based on pre-tests that determined a more accurate representation of fish populations was gained with longer transects over a greater area. Transects were laid out on top of reef patches with consistent depths at each site (Figure 5).[1] Divers conducted all surveys between 9:00 and 15:00 hr. During surveys, one or two transects lines were laid out at the site at least 10 meters apart and then the scuba divers returned to the beginning of the first transect allowing the disturbed fish to settle (5 to10 minutes). Two

[1] At the Miss Cathy, an Artificial Reef, only one transect was set around the boat because of its small size. It was replicated once the next week. The transects were restricted to the boat area, but future surveys may consider laying up to four transects on top of the boat and over the adjacent sandy areas.

observers were trained in fish identification and length estimation using a T-bar marked with 5 cm lines. One diver swam down the middle of the transect and identified to the lowest possible taxon all fishes visible within 2.5 meters to either side of the centerline and 5 meters above. Members of the Blennidae and Gobiidae families were excluded from these counts because of their small size and difficulty distinguishing species. Total length of the fish was estimated to 5 cm, based on size categories used in 2004. Swimming duration per transect varied from 10 to 15 minutes depending on fish abundance. The total area surveyed at each site was 3,000 m^2. A rover method was also used to record any fish observed in the study area before or after the survey, but not seen inside the belt transect. Additional species seen during the rover count were recorded separately and added to those seen inside the belt transect. Analysis involving species richness used the rover count instead of the transect count.

Figure 5. Surveyor recording reef fish along a 30m belt transect laid out along the reef at Double Wreck and estimating total length using a T-bar.

The biomass of five families of fishes (goatfish, groupers, jacks, snappers, and wrasse) was calculated using length estimates. Length estimates were converted to weight estimates using the following length-weight conversion: w = a*l^b, where a and b are constants for the allometric growth equation, l is standard length in mm, and w is weight in grams. Species parameters for length and weight were available from Fish Base (www.fishbase.org).

Fish were also classified according to their trophic level: carnivores, piscivores, mobile invertebrate feeders, sessile invertebrate feeders, omnivores, roving herbivores, territorial herbivores, and planktivores (based on categorizations by Dominici-Arosemena and Wolff 2005 and Halpern and Floeter 2008).

Habitat Characteristics

To control for potential habitat correlates of fish distribution, substrate composition and habitat complexity were measured at each site. Substrate composition was quantified using line intercept transects. A scuba diver swam behind the fish surveyor to minimize fish disturbance, and recorded the type of substrate directly underneath the transect every 5 cm along two 30 meter transects (n = 120 point observations per site). The categories recorded included hard corals, soft corals, bare substrate, rubble, and algae. Hard corals were recorded to the type: branching, brain, finger, fire, mound, and star (based on Reef Check protocol, www.reefcheck.org 2008). Algae groups included macroalgae, turf algae, and encrusting coralline algae. Macroalgae are larger (canopy height >10 mm) erect algae, that have more complex forms and include both fleshy and calcareous forms. Common macroalgae were *Dictyota* and *Halimeda* sp. Turf algae included filamentous algae and filamentous cyanobacteria less than 10 mm in height (AGRRA.org, 2008). Other types of substrate included fire coral, anemones, and other (bare metal). Percent cover of hard coral, soft coral, sponge, turf algae, macroalgae, encrusting algae, and sand was calculated for each site.

At each site, habitat complexity was visually assessed and sites were assigned rankings from 0 to 5 based on the amount, height, and type of relief (method adapted from Polunin and Roberts 1993) (Table 2). Visual estimates of habitat complexity have been correlated with measurements of rugostiy, reef height, and holes of 10-70 cm diameter (Wilson et. al. 2007).

Table 2. Habitat complexity rating for each survey site on a scale from 0-5, with 0 representing no bare substrate and 5 representing the highest complexity (adapted from Polunin and Roberts 1993).

Site	Complexity	Description
Crook's Castle	2	Low but widespread relief
The Blocks	3	Moderately complex
Blair's Reef	3	Moderately complex
Barracuda Reef	4	Very complex, with fissures and caves
Drop Off	3	Moderately complex
Jenkin's Bay	2	Low but widespread relief
Twin Sisters	3	Moderately complex
Doobie's Crack	3	Moderately complex
South of Harbor	2	Low but widespread relief
Sting Ray City	3	Moderately complex
Aquarium	4	Very complex, reefs 5 m high
Corre Corre	1	Low and sparse relief
Venus Bay	1	Low and sparse relief
Double Wreck	3	Moderately complex
STENAPA	5	Very complex, wreck with some coral cover
Charles Brown	3	Moderately complex, large wreck with little coral cover
Miss Cathy	2	High relief, but small area, little coral cover

Data Analysis

Fish density was calculated as the total number of fish at each site divided by the area surveyed. Fish density was tested for normality with the Shapiro-Wilks test and log-transformed if significant for non-normality. Measures of diversity were calculated for each site including species richness (R = number of species), Simpson Index of Diversity (1 – D, where $D = \Sigma (n/N)^2$, Simpson E ($E = (1/D)/R$), Shannon H ($H = - \Sigma p_i \ln p_i$) and Shannon E ($E_H = H/\ln R$). Simpson's Index of Diversity ranges from 0 (monotony) to 1 (infinite diversity). Shannon E determines equitability or evenness of species composition in a community, where 1 is complete evenness. Biomass was tested for normality using the Shapiro-Wilks test and corrected for normality with log-transformations. Total average biomass was calculated for each site and multiple linear regressions of total biomass against habitat variables were performed. Significant effects of habitat correlates on the total density of fish, the density of fish by family, species richness, species diversity, species evenness, and biomass were tested using backwards stepwise multiple regressions in JMP (SAS Institute) (Chapman and Kramer 1999). Analysis of variance (ANOVA) with poc-hoc Tukey-HSD and t-tests were used to determine significant relationships between protection and reef type on fish density, richness, diversity, evenness, and biomass.

A script was written in Python to calculate total fish per site, species richness, and biodiversity and to join this information to an attribute table of site locations in ArcGIS 9.3. Maps were created to visually represent this information around Statia (Appendix).

Results

Fish Recorded

Ninety-six species of fish from 35 families were seen at 17 sites around St. Eustatius (Appendix). Sixty-seven species were seen in the General Use area, 56 at Artificial Reefs, 75 species in the Southern Reserve, and 61 in the Northern Reserve (Figure 6). More fish were seen at protected sites (mean = 1206.8) than unprotected sites (mean = 1081.4). The Southern Reserve and Artificial Reefs had more fish (mean = 1291.6 and 1287.0) than the Northern Reserve and General Use areas (mean = 917.0 and 1100.3). There was no significant difference in the total number of fish observed between reserve and non-reserve areas or among reef types. The highest abundance of fish was observed at STENAPA, one of the Artificial Reef sites, while the lowest number of fish was seen at Venus Bay, a General Use site.

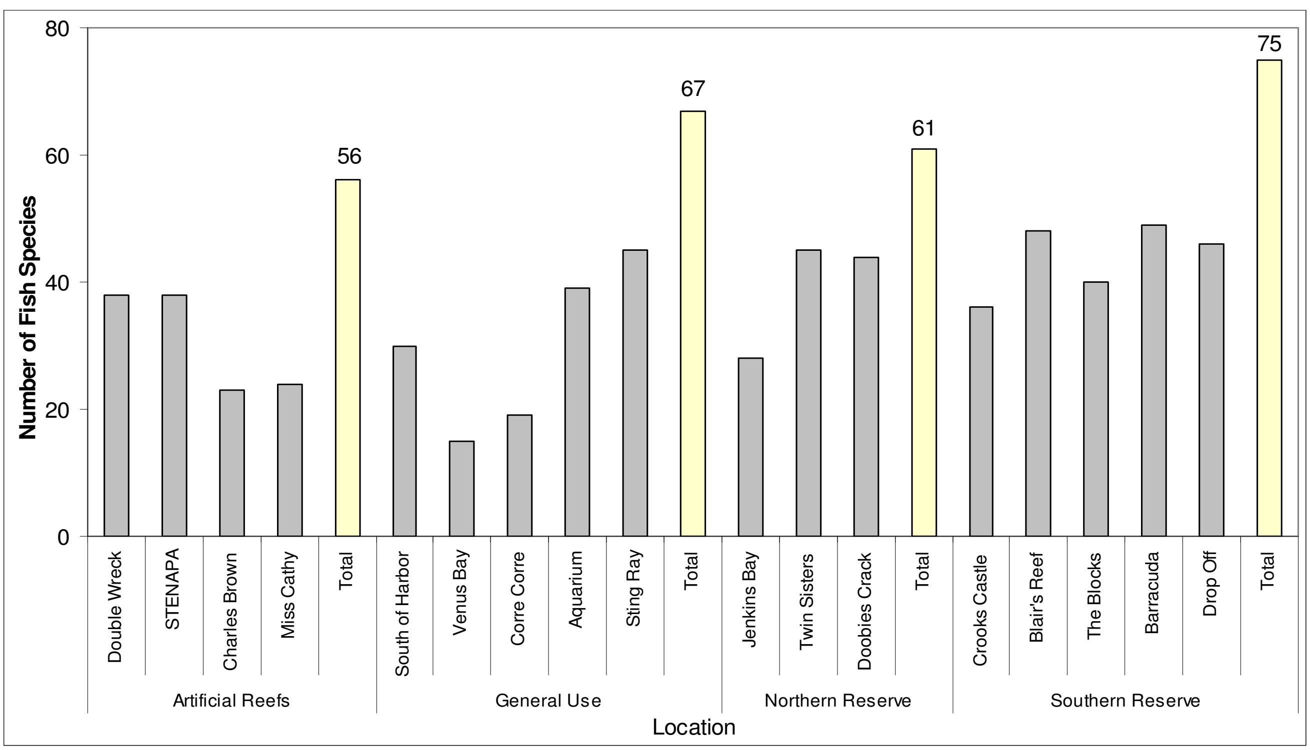


Figure 6. Species richness at each survey site and the total species richness in each reef type.

Carnivores were the dominant trophic group (31%), followed by mobile invertebrate feeders (25%) (Figure 7). The number of individual fish in each group revealed that planktivores were the most dominant fish, made up mostly of pomacentrids. Territorial herbivores, specifically *Thalassoma bifaciatum*, were the next most common. The most frequently observed fish were two types of acanthurids, *Acanthurus coerulus*, *A. bahianus*, one species of labrid, *Thalassoma bifaciatum*, and one serranid, *Epinephelus fulvus*, which were seen at all 17 sites.

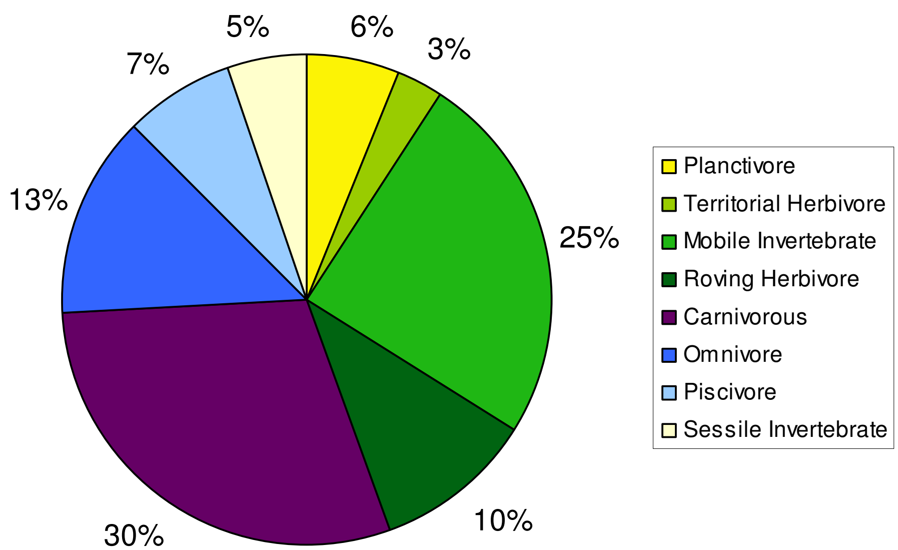


Figure 7. Categorization of the density of reef fish by trophic level.

Habitat

Turf algae was the most common substrate type at all four types of reefs (Figure 8). Macroalge was the next most common substrate type, with *Dictyota* sp. and *Halimeda* sp. being seen frequently. Transects were laid out on top of reef patches, but sand was still a common substrate type, especially in reserve sites. Sand cover was lowest at Artificial Reefs because transects were laid almost completely on top of wrecks. Hard coral cover was low, less than 15 percent at all sites.

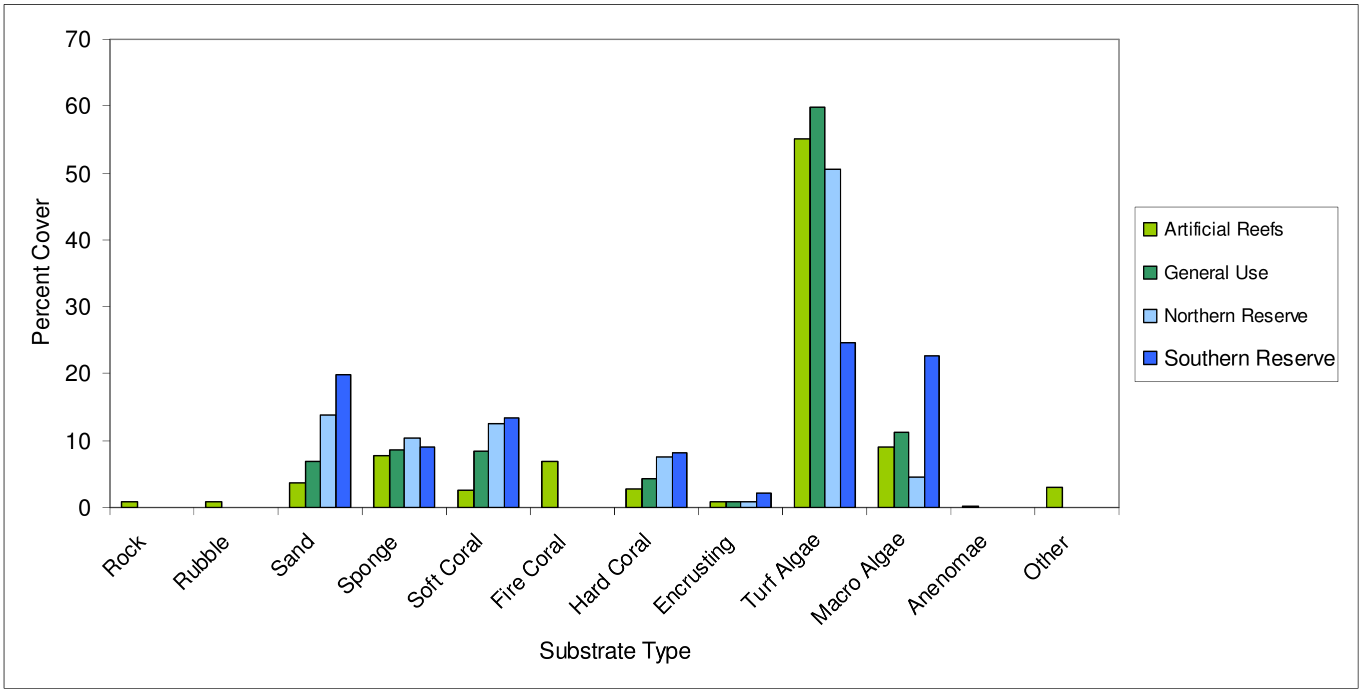


Figure 8. All substrate types found at survey sites in the Marine Park.

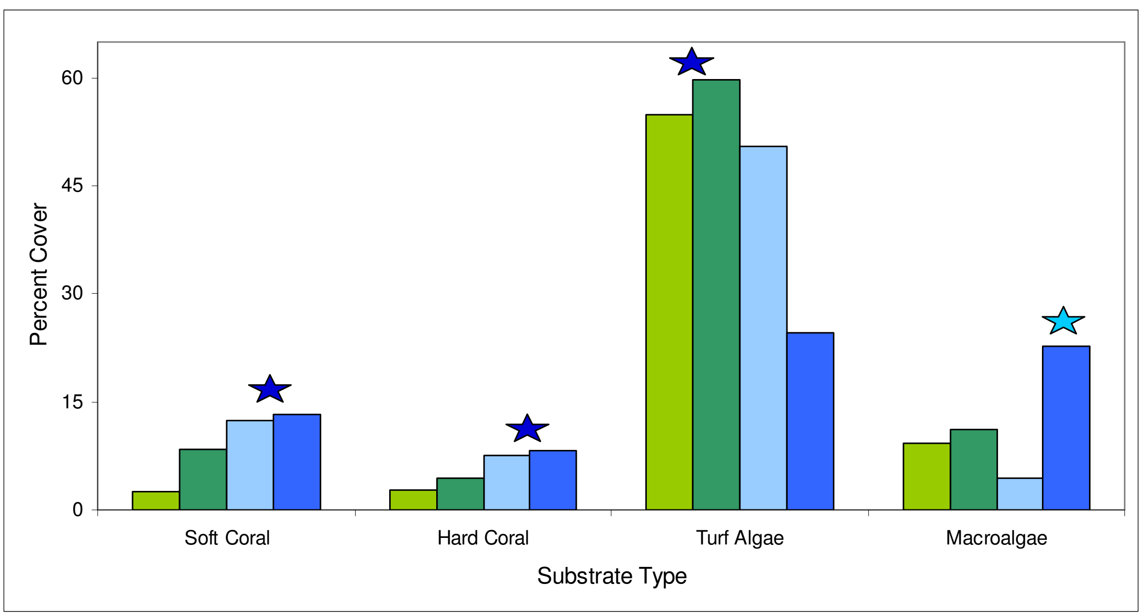


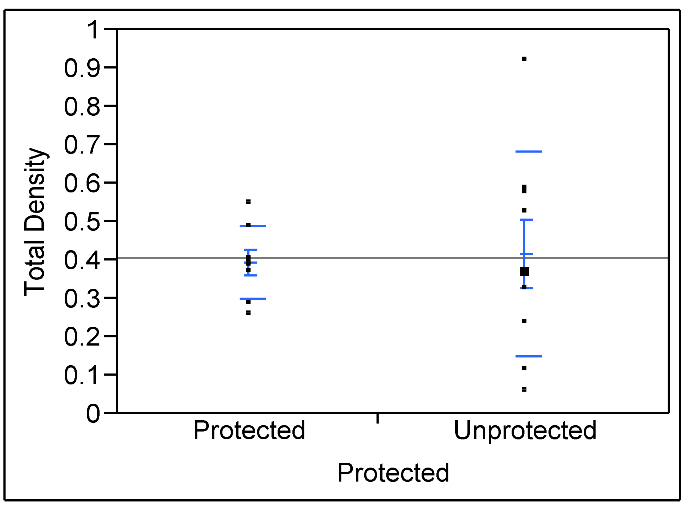
Figure 9. Most common substrate types at the survey sites. Dark blue stars indicate significance of reserve status on substrate type. Significantly more turf algae was seen in unprotected areas than in reserves and more soft and hard coral was observed inside reserves. Light blue star indicates a non-significant trend toward higher macroalgae cover in the Southern Reserve.

There was no difference in depth or habitat complexity between reserve and non-reserve sites (t-test, $p = 0.5515$; $p = 0.6818$). However, there were some significant differences in substrate cover between reserve and non-reserve sites (Figure 9). Hard coral cover was significantly greater inside the reserves (t-test, $p = 0.0009$). Soft coral cover was also significantly greater inside reserves (t-test, $p = 0.0106$). Turf algae was the only substrate type that was significantly greater outside of reserves (t-test $p = 0.0128$). The other substrate types measured: sponge, macroalgae, encrusting algae, and sand did not differ significantly between protected and unprotected sites.

There was a significant difference in hard coral cover between reef types (ANOVA, $F = 4.82$, $df = 3$, $p = 0.0181$). A Tukey HSD test showed that hard coral cover was greater in the Southern Reserve than at Artificial Reefs ($p = 0.0236$). There was also a significant difference between turf algae cover between the four reef types (ANOVA, $F = 3.78$, $df = 3$, $p > F = 0.0377$). The Southern Reserve had lower turf cover than other areas, and cover was significantly lower than cover in the General Use area (Tukey HSD, $p = 0.0361$). Sites in the Southern Reserve had the highest macroalgae cover, but the difference between cover in the Southern Reserve and other areas was not significant. The three sites with the highest macroalgal cover (The Blocks = 48 %, Barracuda Reef = 30%, and Blair's Reef = 22.5) were located in the Southern Reserve.

Fish Density

There was no significant difference found in the mean fish density between reserve and non-reserve sites or among reef types. However, the variation in densities in unprotected areas (range = 0.113 – 0.924 fish/meter^2) was significantly greater than the range at protected sites (range = 0.292 – 0.566 fish/meter^2) (ANOVA, $F = 0.108$, $df = 7,8$, p = 0.0041) (Figure 10). F-tests for two sample variances indicate that there were significant differences between the variance in total density of fish between the Artificial Reef sites and the Southern Reserve (F = 0.086, df = 4,3, p = 0.0192), between Artificial Reefs and the Northern Reserve (F = 0.006, df = 2,3, p = 0.0060), and between the General Use sites and the Northern Reserve sites (F = 0.019, df = 2,4, p = 0.0189). Average fish density was highest in the Artificial Reef, which contained the sites with both the highest (STENAPA) and lowest (Charles Brown) densities (Figure 11). The Southern Reserve had a slightly lower average density, and variation in densities among slights was significantly less.



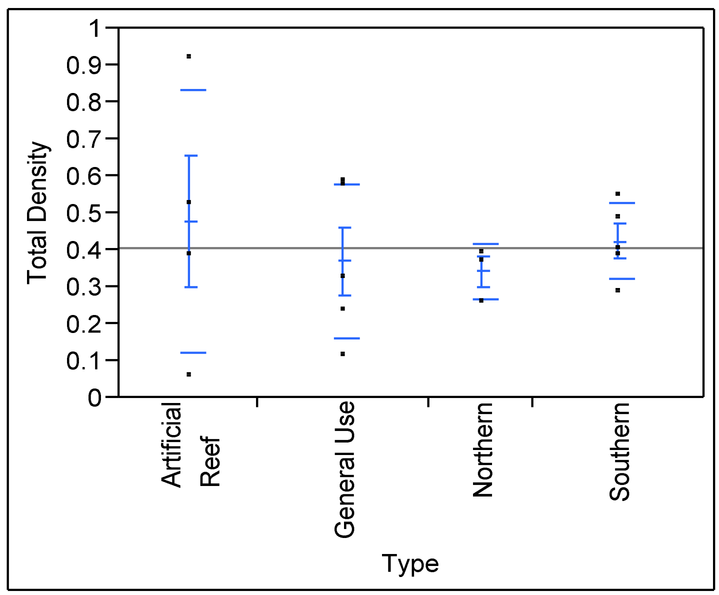


Figure 10. Total density varied significantly more at unprotected sites than inside the reserves, with the most variation occurring at Artificial Reef sites and the least variation in the Northern Reserve.

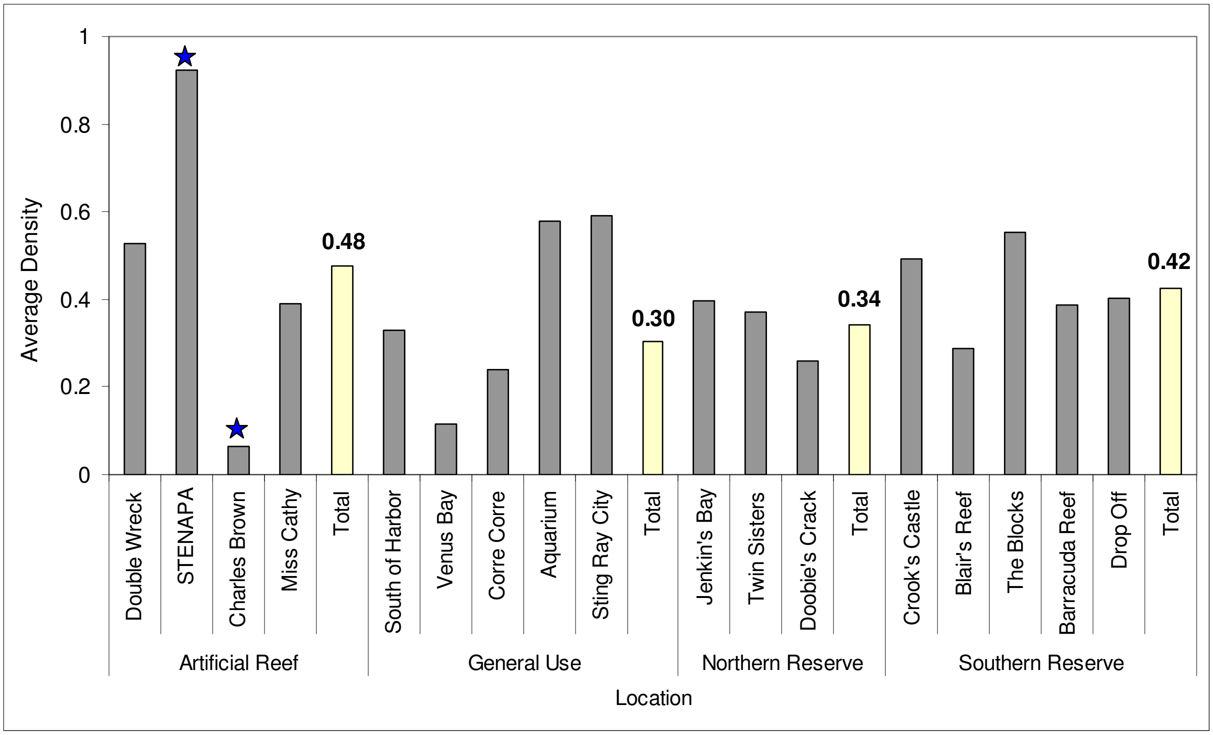


Figure 11. Average fish density at each survey site and the average total density at different reef types. Stars indicate the sites with the highest and lowest average density.

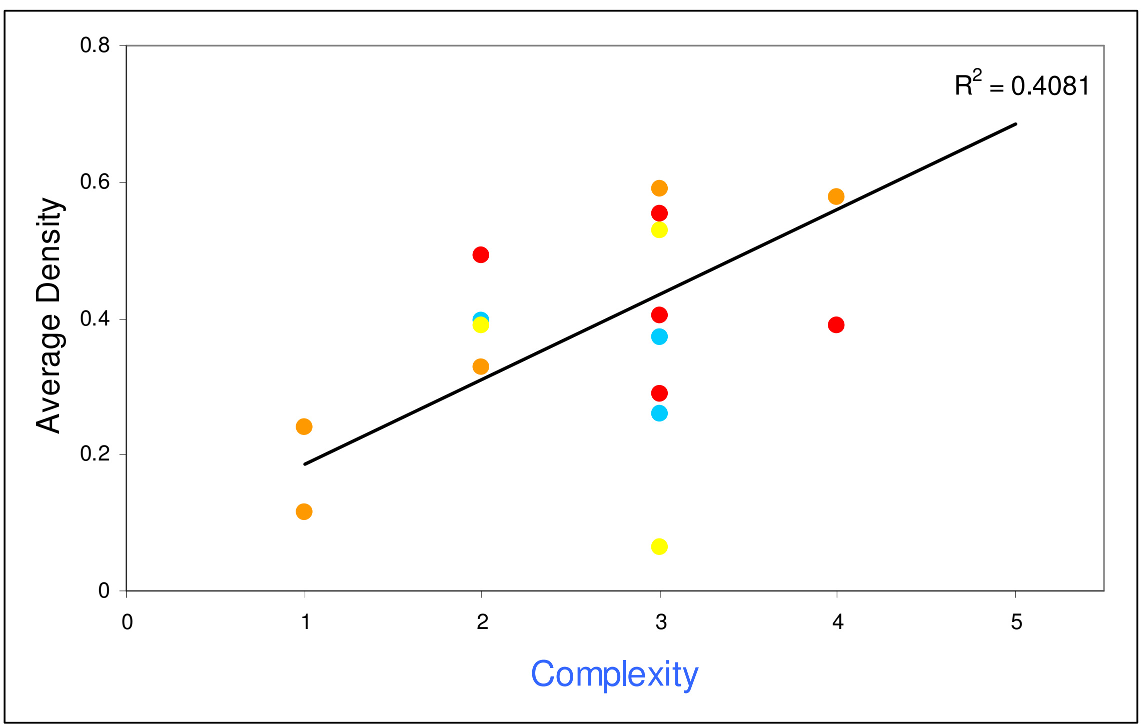


Figure 12. Average reef fish density had a significant positive relationship with habitat complexity (r^2 = 0.4081, p = ???). Red = Southern Reserve, Blue = Northern Reserve, Orange = General Use, Yellow = Artificial Reef.

A backwards stepwise model of fish density against all habitat variables showed a significant positive effect of complexity (intercept = 0.1559, p = 0.0004) and negative effect of depth (intercept = -0.0045, p = 0.0139) on density (whole model, $r^2 = 0.62$, p = 0.0011) (Figure 12). Backwards stepwise models also showed habitat characteristics to be correlated with the density of different fish families (Table 3).

Table 3. Backwards stepwise model results showing habitat variables that affect abundances of reef fish families. Significant model results are shown in bold.

	Model Fit		Model Coefficients									
Family	R^2	p	Intercept	Complexity	Depth	Hard Coral	Soft Coral	Sponge	Macro-algae	Turf	Encrusting	Sand
Ostraciidae	0.56	**0.0274**	-9.088	0.998	-0.016				0.060		-0.490	
Chaetodontidae	0.43	**0.0025**	-0.0003			0.0003						
Serranidae	0.76	**0.0027**	-0.035		0.0001	-0.001	0.0008	0.0006		0.0004	0.0030	0.0003
Haemulidae	0.46	**0.0315**	0.162		-0.0004	-0.0036			-0.0015	-0.0016		-0.0011
Carangidae	0.34	0.0574	0.0001				-0.0003			-0.0002	-0.0022	-0.0002
Scaridae	0.42	**0.0028**	0.0049						0.0003			
Lutjanidae	0.53	**0.0092**	-0.0107		0.0002	-0.0024	0.0012	0.0009				
Holocentridae	0.29	0.0604	-0.0143	0.0086					0.0011		-0.0110	
Acanthuridae	0.21	**0.0370**	0.0063							0.0002		
Balistidae	0.12	0.9170	0.0003								0.0002	
Labridae	0.65	**0.0003**	0.2408		-0.0022						0.0113	

Fish Diversity

At every site, the rover method for counting fish added species to the record from the four belt transects. Species richness varied from a low of 15 at Venus Bay in the General Use area to a high of 49 at Barracuda Reef in the Southern Reserve (See Figure 6). Among the reef types, species richness was highest in the Southern Reserve (75 species), where three of the sites with the highest species richness were located. Despite having the two sites with the lowest number of species, the General Use area had the next highest richness (66 species), followed by the Northern Reserve (60 species) and Artificial Reefs (58 species). Overall, there were more species observed in protected areas (82 species) than unprotected areas (74 species).

Linear regressions of fish species richness against all habitat variables showed significant positive effects of complexity ($p = 0.0217$), hard coral cover ($p = 0.0029$), soft coral cover ($p = 0.0130$), and sponge cover ($p = 0.0297$), and a significant negative effect of turf algal cover ($p = 0.0108$) on richness (Figure 13). A backward stepwise model of fish species richness against all habitat variables showed significant effects of complexity (intercept = 11.5798, $p < 0.0001$), soft coral (intercept = 1.3318, $p < 0.0001$), sand (intercept = 0.7595, $p = 0.0005$), turf algae (intercept = 0.6422, $p = 0.0026$), and macroalgae (intercept = 0.6832, $p = 0.0030$) on species richness (whole model, $r^2 = 0.898$, $F = 19.39$, $df = 5$, $p < 0.0001$).

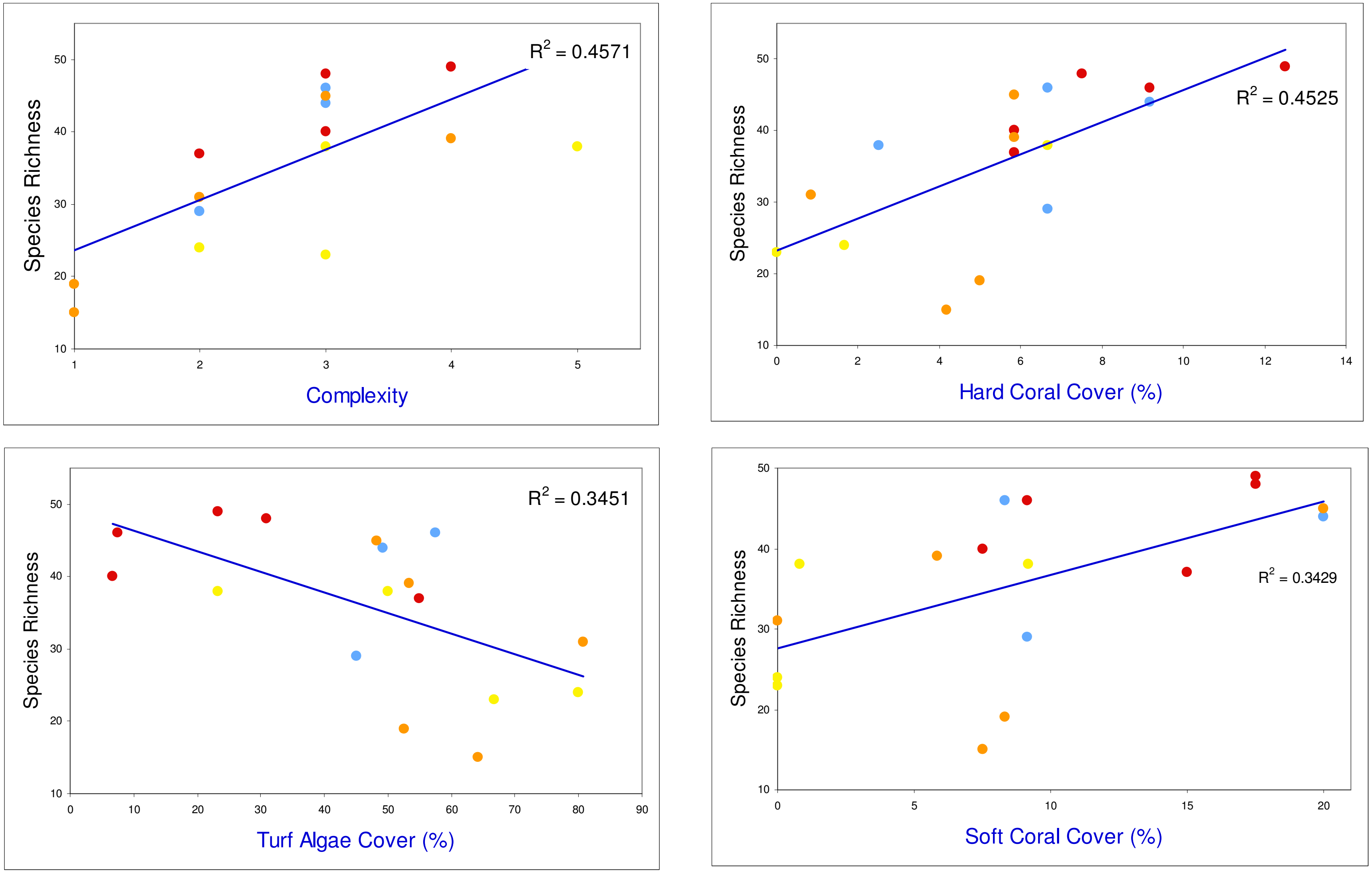


Figure 13. Significant relationships between species richness and increasing habitat complexity and hard and soft coral and decreasing turf algae cover. Red = Southern Reserve, Blue = Northern Reserve, Orange = General Use, Yellow = Artificial Reef.

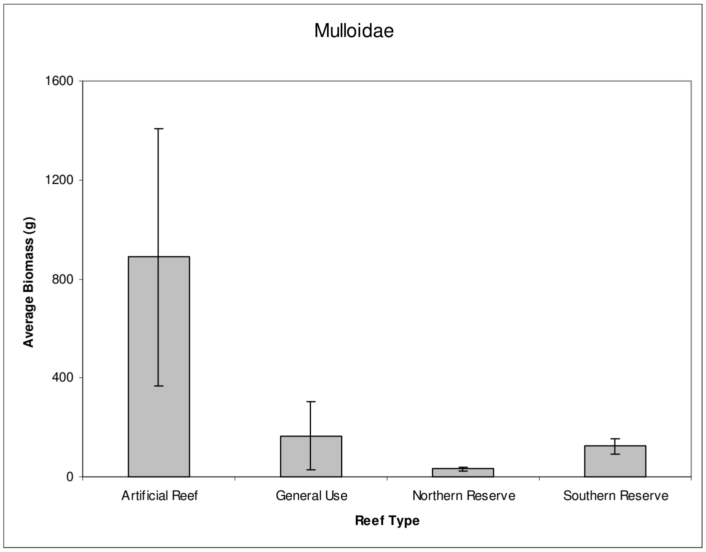
The Simpson Index of Diversity was slightly higher for the protected area than unprotected area and the Southern Reserve had the highest overall biodiversity. The Simpson Index results were very similar for all sites (scale 0-1, D = 0.600 to 0.866). However, the site with the greatest biodiversity was the Miss Cathy, which is an Artificial Reef. Multiple linear regressions showed insignificant trends toward increasing biodiversity with increasing hard and soft coral, sponge, and macroalgal cover. Species evenness was highest in the Northern Reserve and lowest in the General Use areas. Multiple linear regressions of Shannon E and habitat correlates showed an insignificant negative trend with habitat complexity and sand and an insignificant positive trend with turf algae.

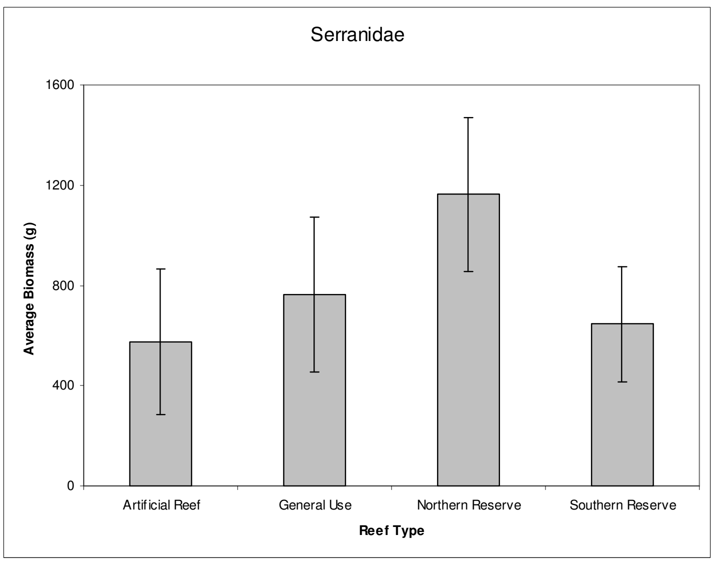
Table 4. Summary of survey site average density, species richness (comparison of counts from belt transects and rover surveys), Simpson D biodiversity, and Shannon E species evenness.

Site	Density	Richness Transect/Rover	Simpson D (1-D)	Shannon E
Crook's Castle	368.8	29/37	0.755297443	0.52740956
The Blocks	424.3	31/40	0.777414058	0.562403
Blair's Reef	215.3	35/48	0.805269822	0.59075782
Barracuda Reef	295.3	36/49	0.771545806	0.541032
Drop Off	307.5	32/46	0.675962721	0.44942064
Southern Reserve	**1291.6**	**75**	**0.804168**	**0.51823**
Jenkin's Bay	297.5	24/29	0.753613445	0.5731755
Twin Sisters	278.8	34/46	0.719103943	0.53787159
Doobie's Crack	194.3	31/44	0.815654043	0.66369413
Northern Reserve	**1100.3**	**61**	**0.798102**	**0.57250**
South of Harbor	248.8	29/31	0.703054973	0.49913944
Sting Ray City	404.5	29/45	0.826459439	0.64944753
Aquarium	437.25	35/39	0.749646698	0.52327362
Corre Corre	179.3	18/19	0.697680752	0.53042817
Venus Bay	86.3	14/15	0.716740181	0.62179381
Protected	**1206.8**	**82**	**0.816619**	**0.52031**
General Use	**917.0**	**67**	**0.779401**	**0.51375**
Double Wreck	396.8	38/38	0.815654043	0.62745313
STENAPA	693	32/38	0.606021824	0.44353806
Charles Brown	122.8	22/23	0.600636301	0.47941533
Miss Cathy	150.5	21/24	0.866899924	0.76090808
Artificial Reefs	**1287.0**	**56**	**0.791415**	**0.55792**
Unprotected	**1081.4**	**73**	**0.811248**	**0.54078**

Biomass

Biomass increased significantly with increasing depth ($r^2 = 0.3708$, $p = 0.0095$) and decreased with increasing macroalgae cover ($r^2 = 0.249$, $p = 0.0415$). The Charles L. Brown had the highest average biomass, but it was not significantly higher than the biomass at other survey sites. There was also no significant relationship between biomass and reserve status or biomass and reef type. There was no significant difference between the biomass of goatfish at protected and unprotected sites (Figure 14). The highest biomass of goatfish was seen at Artificial Reefs, but levels were not significantly higher than at other reef types. Biomass of jacks increased significantly with increasing depth ($r^2 = 0.43$, $F = 8.99$, $df = 1$, $p = 0.0111$). Unprotected areas had slightly higher biomass of jacks, but no significant difference was found (Figure 14). There were a large number of horse-eyed jacks (*Carax latus*) seen at the Charles Brown, but there was no significant difference in jack biomass between reef types. Biomass of goatfish, groupers, snappers and wrasse were not significantly correlated with any habitat variable. Biomasses of these fish were not different between protected and unprotected site or between any of the reef types (Figure 14). Groupers biomass was higher in the Northern Reserve than in other reef types, but no significance was found. Biomass increased significantly with depth ($R^2 = 0.3708$) and decreased with increasing macroalgal cover ($R^2 = 0.249$). (Figure 15).


Mulloidae
1600
1200
800
400
0
Average Biomass (g)
Artificial Reef
General Use
Northern Reserve
Southern Reserve
Reef Type


Serranidae
1600
1200
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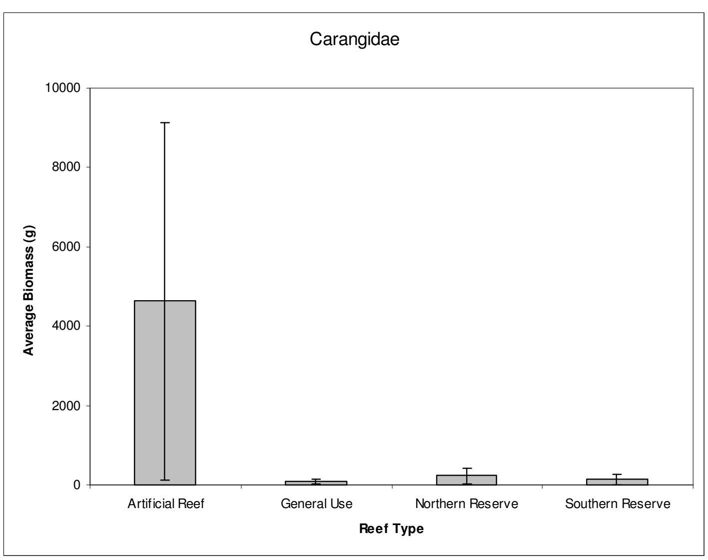
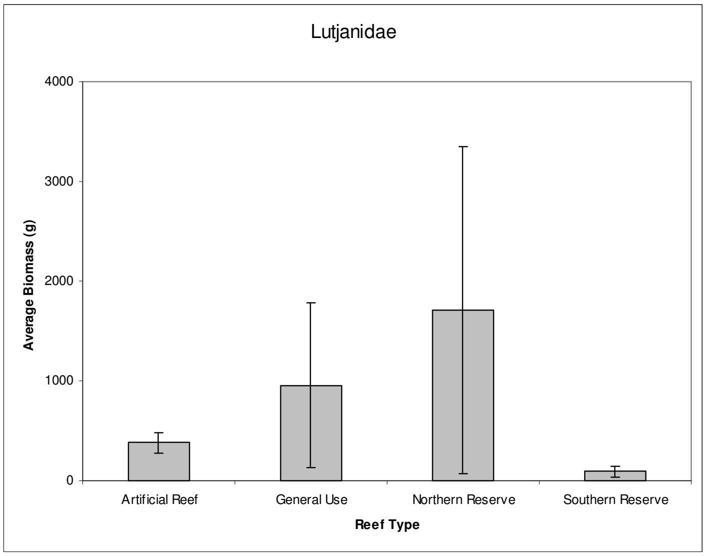
400
0
Average Biomass (g)
Artificial Reef
General Use
Northern Reserve
Southern Reserve
Reef Type


Carangidae
10000
8000
6000
4000
2000
0
Average Biomass (g)
Artificial Reef
General Use
Northern Reserve
Southern Reserve
Reef Type



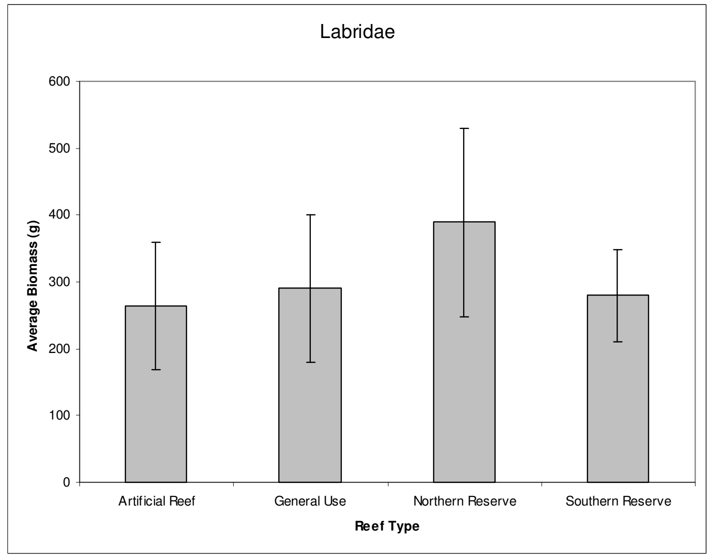
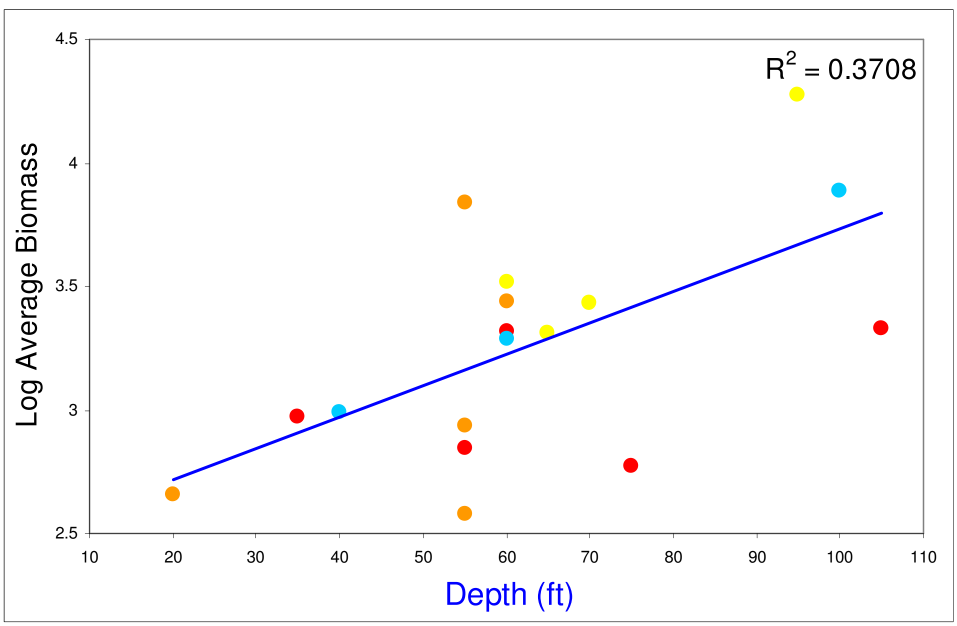


Figure 14. Average biomass of goatfish (Mullidae), grouper (Serranidae), jacks (Carangidae), snappers (Lutjanidae), and wrasse (Labrids) with standard deviation bars.



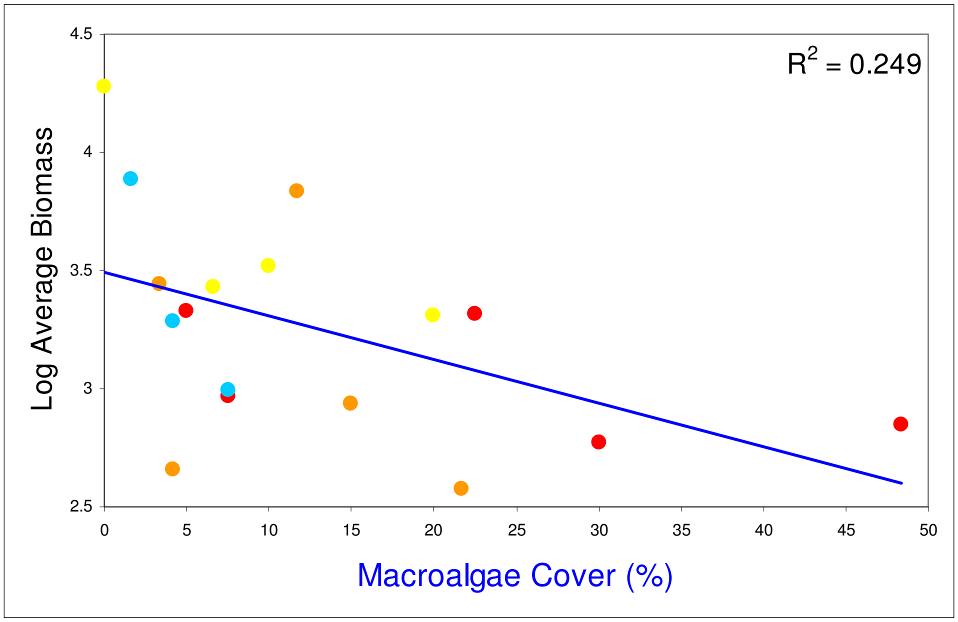


Figure 15. Log-transformed average biomass increased significantly with increasing depth ($R^2 = 0.3708$) and decreased with increasing macroalgal cover ($R^2 = 0.249$). Red = Southern Reserve, Blue = Northern Reserve, Orange = General Use, Yellow = Artificial Reef.

Artificial Reefs

Four Artificial Reefs of varying ages and depths were surveyed to determine if differences existed between the sites. Double Wreck sank about 300 years ago. It consists of mostly ballast stones that have fused together, and have been covered in turf and macroalgae and some hard coral and sponge. STENAPA Reef consists of a barge and tugboat that were sunk in 1997 to provide a new fishing area after the no-take reserves were created, but it is frequently used as a dive site instead. This reef has become more heavily encrusted with hard and soft coral and sponges than Double Wreck and contains less turf and macroalgae. The Charles Brown is a 100 meter long vessel that was sunk in 2002. The boat is mostly covered in turf algae, but also has large amounts of fire coral growing on its surface. Finally, the Miss Cathy, a small tugboat, was sunk in 2006 to be used exclusively for fishing purposes. This site contains the highest amount of turf algae, some macroalgae and sponge, and little hard coral.

Linear regressions of reef age against total density, biomass, and species richness were performed, but no strong relationships were found (R^2 = 0.013; - 0.245; 0.353) (Figure 16). Linear regressions of depth against density, biomass, and species richness revealed stronger correlations (R^2 = - 0.802; 0.860; 0.533) (Figure 16). Complexity was positively related to total density and species richness (R^2 = 0.533; 0.451) but not to biomass (R^2 = 0.0002) (Figure 16).

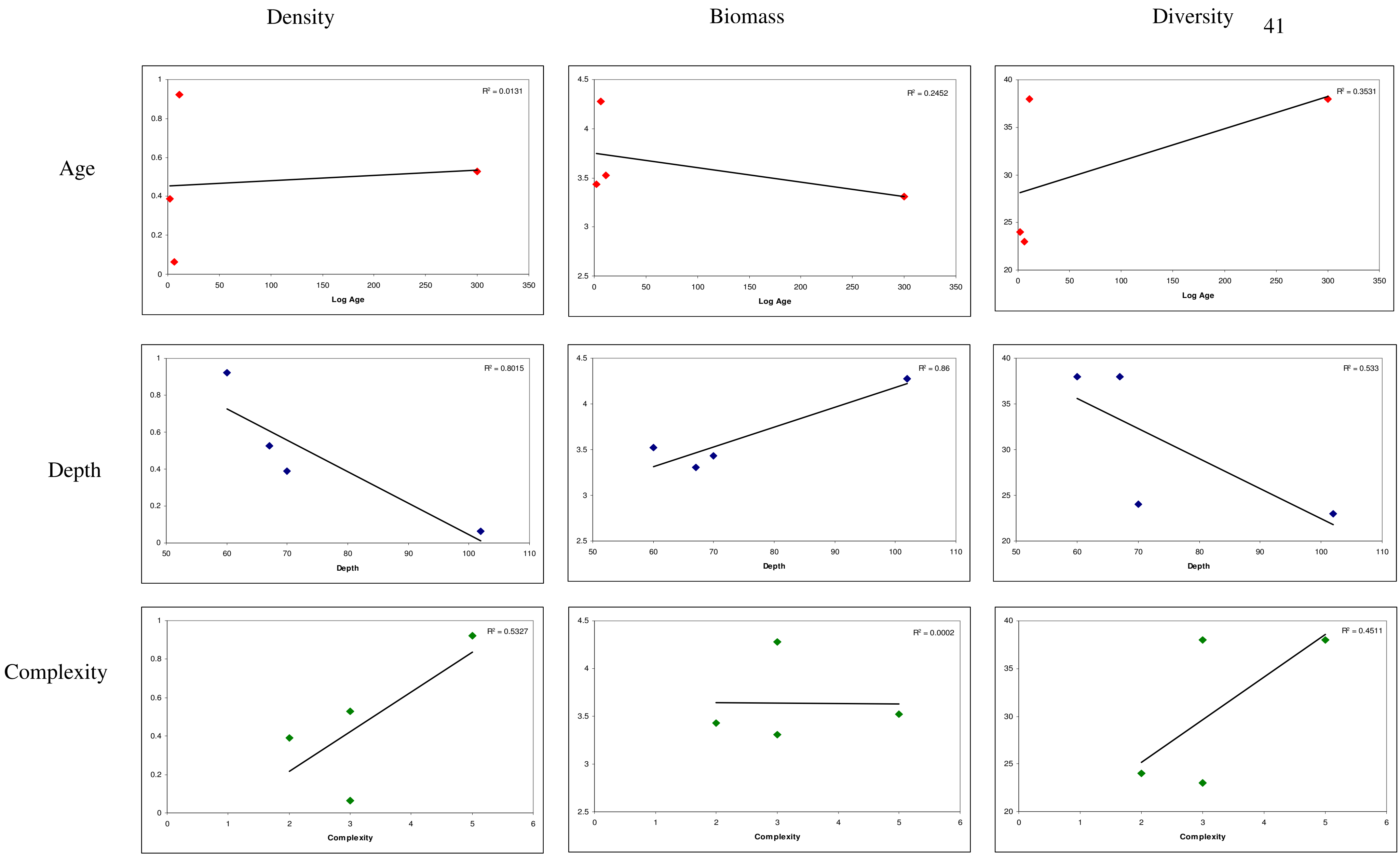


Figure 16. Relationships between reef age, depth, and habitat complexity and density, biomass, and diversity at Artificial Reefs.

Discussion

Habitat

Sometimes colinearity between reserve status and habitat variables can result in false significance of reserve status on fish density, diversity, and biomass (Chapman and Kramer 1999). Standardized transect surveys conducted in the St. Eustatius Marine Park reveal that habitat complexity and depth of sites did not differ between reserve and non-reserve sites or between reef types, but some substrate characteristics differed significantly between these two areas. In general, habitat was of higher quality inside the reserves than outside. Based on coral cover and habitat complexity, the Southern Reserve has the highest quality reef habitat overall. Turf algae was found in greater amounts outside of reserves than inside reserves, and was lowest in the Southern Reserve. Hard coral and soft coral cover were greater in reserve areas, especially in the Southern Reserve. However, sites in the Southern Reserve had high levels of macroalgae, which competes for space with corals. The General Use area had the lowest reef habitat quality. These areas had the highest levels of turf algae and the second highest level of macroalgae. It was expected that the protected areas would have higher quality reef habitat than unprotected areas because the reserves were designed to protect dive sites, which are usually situated in places where divers can see corals and associated reef fishes.

Fish density, diversity, and biomass

There were differences in reef fish populations found between protected and unprotected sites and between reef types. Fish density did not differ between protected and unprotected sites, but the variation in fish density was greater in unprotected sites. The difference in variance can be seen by comparing sites within the Artificial Reefs; STENAPA Reef had the highest density of fish, while the Charles Brown had the lowest. The sites in the Southern Reserve, on the other hand, exhibited little variation in fish density. Species richness was significantly higher in the Southern Reserve than in other reef types. The General Use area had the next highest number of species observed. This may be due to the fact that species richness is a count and not a mean, and does not account for the variation in sample sizes between reef types. The General Use areas also varied in habitat type and depth more than the other areas, with three sites on the leeward side of the island and two on the Atlantic side. Because there was variation in habitat type, fish were seen at some of these sites that were not seen anywhere else, including two labrids, *Halochoeres bivittatis* and *H. radiatus*. Biomass was highest at the Artificial Reef sites, but there was no significant difference found among sites. Biomass calculations were restricted to five families of fish, so total biomass may differ from that reported here.

Effects of habitat, reserve status, and reef type on fish density, diversity, and biomass

Regression models showed that habitat characteristics influenced fish populations more than reserve status. Habitat complexity was found to be the most important variable in predicting both fish density and diversity. Diversity was also influenced by hard coral cover and turf algae cover. Biomass was most strongly influenced by depth and macroalgae cover. When evaluating the effects of marine reserves, the role that habitat plays in influencing reef fish populations must be considered. Additional factors such as fishing pressure over time, pollution levels, and recruitment could also be influencing fish distributions and should be measured.

Despite the large influence of habitat, the no-take reserves may still be important for protecting reef fish populations. Marine reserve effectiveness can rarely be measured by comparing sites inside and outside of reserves because reserves are designed around areas of higher reef quality and larger fish populations. The sites surveyed in this study were significantly different, making it difficult to make direct comparisons. However, it is important to survey sites inside and outside reserves over time to detect changes. The two reserves have been in place for more than 10 years and there can be no conclusive determination or comparison of fish population trends during that time.

Changes Over Time

Habitat

An Atlantic and Gulf Rapid Reef Assessment (AGRRA) was carried out in the windward Netherlands Antilles in 1999. Ten sites in the Southern Reserve were surveyed and revealed coral cover to be higher (range = 12 +/- 4 to 47 +/- 12, mean = 22 +/- 13) in St. Eustatius than neighboring Saba and St. Maarten (Klomp and Kooistra 2003). This assessment found fleshy macroalgae (*Dictyota sp.*) to be abundant at only two sites that they surveyed, neither of which were assessed during this study.

A coral bleaching event occurred from late August through October of 2005 (Esteban and Kooistra 2005). During this time, corals at depths of 3 meters to greater than 30 m in depth of all morphologies were affected. On October 7, at the deepest dive site (Drop Off West, 37 m), between 70 and 80% of all hard corals were bleached, including fire corals (Esteban and Kooistra 2005). The corals in the Southern Reserve appeared to be more affected than those in the Northern Reserve (pers. comm. Nicole Esteban, 2008). Reef Check surveys from early 2005 indicate that sites in the Southern Reserve contained up to 30% hard coral and about 40% macroalgae. A Reef Check survey performed during the summer of 2008 at the same sites showed less than 10% hard coral cover and 65% macroalgal cover. Sharp declines in coral cover and increases in turf algae have been shown to correspond with parallel declines in fish biodiversity in both marine reserves and areas open to fishing; these declines appear to arise from habitat limited recruitment, and possibly adult mortality from loss of food and shelter (Jones et. al. 2004).

Fish

More species of fish (96) were recorded during this survey than any other carried out in St. Eustatius. In 1994, a survey was undertaken from the Terminal Jetty to Jenkin's Bay, in the Northern Reserve, to examine fish populations. During nine survey dives, 60 species of fish were observed (Grabowsky and Poort International nV 1994). In 2005, a total of 64 species were recorded at 15 sites around the island (White et al 2006). Although it is difficult to draw direct comparisons, because more sites were surveyed and additional time was spent in the water in 2008 than in previous surveys, the relative species richness between reef types remains similar. The Southern Reserve continues to have the highest species richness.

A comparison of fish observed from 2004 with those seen in 2008 shows many differences in species, which may indicate shifts in fish types or reveal additional problems with fish identifications in 2004. The 2008 surveys recorded species of fish that were not described during previous surveys. This included species that were commonly seen in 2008, such as harlequin bass, lane snapper, sand tilefish, trumpet fish, and fairy basslets. There were low numbers of boxfish seen during the 2004 surveys, but they were very common in 2008. The difference between the numbers of goatfish between years was even greater, with significantly more observed in 2008. There appeared to be a shift from sightings of a few large snapper species (cubera and dog) to larger numbers of smaller species (lane) from 2004 to 2008. This also seemed to occur with grouper species; rock hind and yellowfin grouper were recorded in 2004, but neither species was seen in 2008. Furthermore, coneys were seen at every site and graysbys were common at most sites in 2008, while coneys were observed in lower numbers in 2004 and no graysbys were recorded at any site. Although sharks were infrequently seen on dives in 2008, both reef sharks and nurse sharks were seen within the Southern Reserve, and a reef shark was observed at a deep site in the Northern Reserve. In 2004 only two nurse sharks were recorded. Finally, a wahoo and big eye tuna were recorded in 2004, but these species were never seen on any dive during 2008.

Conclusions: Future Action

Reef Fish and Habitat Surveys

This study established methods for future assessments of reef fish and habitat, so that changes over time can be quantified. Unfortunately, there was no reliable baseline established for reef fish populations before the Marine Park was created, so there is no way of knowing what changes have occurred in the last 11 years. The present study determined that habitat differences among the survey sites affect reef fish populations. Future research should monitor reef fish populations and their habitat to determine if the reserves are having beneficial effects.

The decline of coral communities in St. Eustatius may continue despite the protection provided by marine reserves. Coelho and Manfrino (2007) found that even a 20 year no-take marine reserve in Little Cayman, a relatively undeveloped Caribbean island, has not provided prevented coral community degradation. Marine reserves have the potential to increase herbivore populations to limit the competition between corals and macroalgae, but they cannot prevent disease outbreaks or bleaching events (Coelho and Manfrino 2007). Solutions to slow or reverse coral decline could take the form of coral restoration, disease control programs, bleaching prevention by shading corals when water temperatures become too high, and methods to increase herbivory, through *Diadema* aquaculture. Coelho and Manfrino (2007) believe that some solutions can be implemented at a local or regional level by involving local stakeholders, such as dive operators or management organizations like STENAPA. The keys to any of these proposed solutions is providing funding for materials and training, and organizing a support network that could manage data and offer expertise and guidance (Coelho and Manfrino 2007).

Artificial Reef Management

It was expected that the areas with the highest complexity would be correlated with high species richness (Luckhurst and Luckhurst 1978, Claudet and Pelletier 2004). A multivariate model of habitat variables and species richness showed that there was a significant relationship between complexity and richness. Overall, the Artificial Reef sites had the highest mean complexity (3.25), but they had the lowest species richness. High species richness was seen at two of the older Artificial Reef sites, Double Wreck and STENAPA (R = 38, 38) but the species richness at the two newer sites, Charles L. Brown, sunk in 2003, and the Miss Cathy, 2005, were much lower (R = 23, 24). Differences in habitat cover were not statistically significant, but the older Artificial Reef sites had more hard and soft coral, sponge, and macroalgae and less turf algae than the newer sites.

Comparisons with previous surveys at these sites indicate that species richness is increasing over time. The 2004 survey at the Charles Brown recorded 20 species of fish (White et al 2006) and a 2006 survey at Miss Cathy recorded 14 different fish species (Northrop and Polino, unpublished 2006). It seems likely that as the artificial reefs age, fish density, diversity, and biomass will also increase. Arena et. al. 2007 found a significantly positive relationship between mean species richness and artificial vessel-reef age. However, the results of the present study are not enough to support this argument; ages of the Artificial Reef did not have strong relationships with fish density, biomass, or diversity levels. The sites did not just differ in age, but also in depth and habitat complexity, both of which influenced fish density, biomass, and diversity to a greater extent. Double Wreck, as a 300 year old structure that looks more like a patch reef than a wreck, may not be comparable to newer metal boats. If this site is removed from analysis, both fish density and diversity increase with age, but biomass continues to be most influenced by depth.

Future use of artificial reefs as a management tool for fisheries in Statia should consider depth at the placement site and structural complexity. Experiments with artificial reefs in St. Thomas identified the type of design preferable for sustainable fisheries: artificial reefs should include both small holes, which provide refuges from predation for small fishes, and large holes, which act as home sites for predatory "target" species (Hixon and Beets 1989).

Outreach to Fishermen

STENAPA desires to have a better understanding of what areas of the Marine Park the fishermen use, what time of year they focus their efforts, which fish species they target, and how successful their efforts are in order to improve their management strategies (pers comm. Nicole Esteban, STENAPA, 2008). This is difficult because the fishing population on Statia is small and fish are caught for local consumption so no central processing facility or distribution center exists. The lack of organization or central meeting place for fishermen makes it difficult to determine the number and type of fish that are taken out of the Marine Park.

White et. al. (2006) carried out semi-structured interviews with fishermen and dive operators in the summer of 2004 and efforts were made to replicate these interviews during the present study. White was able to interview nine fishermen and two dive shop owners, who had all been fishing between 10 and 20 years in Statia. The fishermen indicated that they targeted pelagic species including wahoo and dolphin and reef associated species like snapper and grouper. However, fish traps catch a wide variety of reef associated fish that are often kept, even if not targeted (pers. obs. 2008).

As part of the present study, a survey was designed to supplement the information gathered in 2004, but attempts made to approach the fishing community with it were rejected (Appendex). The failure was due, in large part, to the researcher not establishing a relationship with the fishing community upon arrival in Statia. Future efforts to implement this type of interview survey would be more successful if it could be carried out by an individual with previously established trust among members of the fishing community or if better relations with STENAPA staff members could be formed to make this type of information more accessible.

Oil Terminal

Coral reef systems are among the coastal environments most likely to be adversely affected by oil contamination (Cram et. al. 2006). Oil pollution, especially chronic oil spills, can have adverse affects on coral reproduction, growth rates, colonization capacities, feeding, and behavioral responses (Loya and Rinkevich 1980). Statia Terminals nV's tanker anchorage has been studied (White et. al. 2007), but no attempts have been made to evaluate the impact of oil contaminants on the local marine environment. Studies commonly use two contaminants, polycyclic aromatic hydrocarbons (PAHs) and heavy metals, to assess oil pollution (Cram et. al. 2006). PAHs have a large environmental risk because they have carcinogenic and mutagenic properties, but heavy metals bioaccumulate in the food chain and can reach toxic concentrations in body fluids, tissues, and organs (Cram et. al. 2006). Results from one study in Mexico found that some pollution was directly related to oil company activities, but could also be attributed in part to high levels of marine traffic. In addition, ballast water from tankers was a primary source of pollutants. Future research in the Marine Park should monitor oil pollutants to determine if the Terminal's activities are affecting water quality, coral communities, and fish populations.

Conclusions: Greater Caribbean

The results from this study should be viewed in the context of larger Caribbean trends. Paddack et. al. (2009) analyzed data from 318 Caribbean reefs and showed that overall reef fish density declined significantly with changes in benthic cover. Declines in fish density were not related to fishing; declines were similar among fished and non-fished fishes, and fish at higher trophic levels, which are impacted by fishing the most, showed no greater declines than other groups (Paddack et al 2009). Declines in reef associated fish have been correlated with loss of coral cover. Since the mid-1970s, coral cover has been reduced by 80% in the Caribbean (Gardner et al 2003). Declines in fish populations in the Indo-Pacific have been shown to lag 5 to 10 years after discrete catastrophic coral mortality, but in the Caribbean, coral decline has been gradual and fish population decline has only been significant in the last decade (Paddack et al 2009).

With this in mind, marine parks and reserves, such as those in St. Eustatius may play a crucial role in the long term recovery of fisheries exploited at subsistence and commercial levels. While data available at present may simply reflect yet another example of declining coral reefs and fish populations in the Caribbean, a progressive regulatory framework and a willingness among stakeholders to work together offer the hope that multiple use management will complement sustainable use goals over the long term. Standardized field surveys, like the one developed in the present study, to monitor the ensuing effects are critically important.

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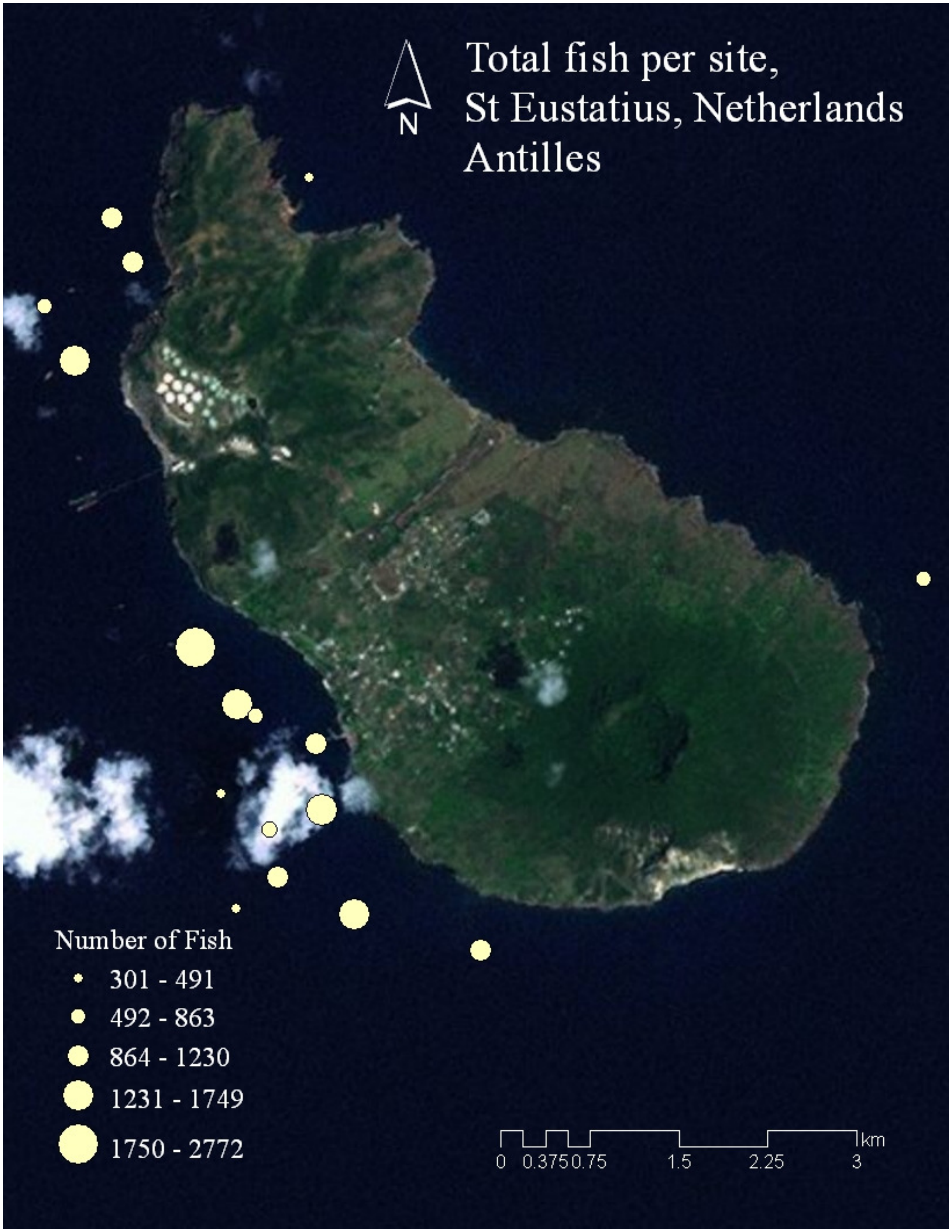
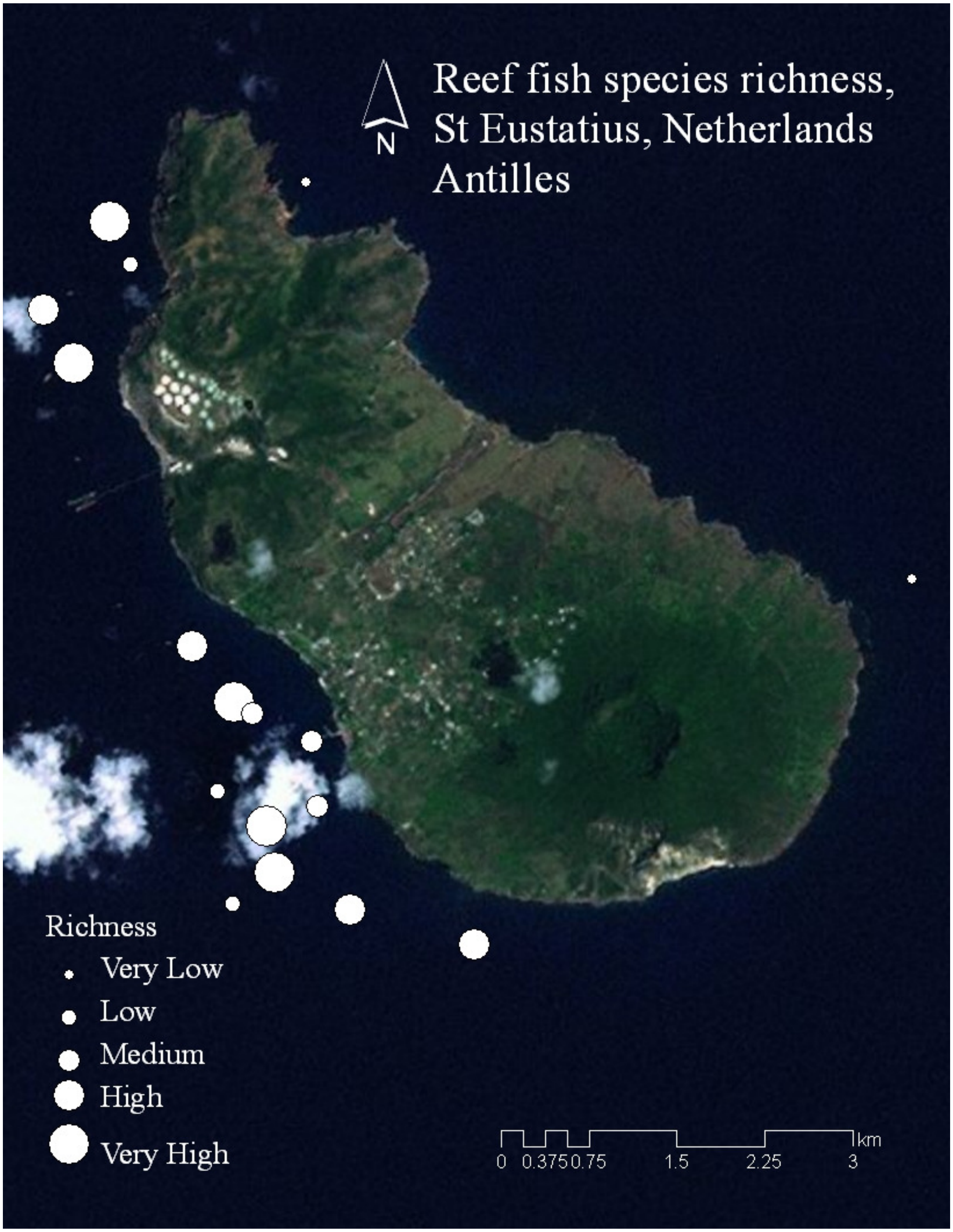
Appendix


Total fish per site,
St Eustatius, Netherlands
Antilles
N
Number of Fish
301 - 491
492 - 863
864 - 1230
1231 - 1749
1750 - 2772
0 0.375 0.75 1.5 2.25 3
km


Reef fish species richness, St Eustatius, Netherlands Antilles
N
Richness
Very Low
Low
Medium
High
Very High
0
0.375
0.75
1.5
2.25
3
km

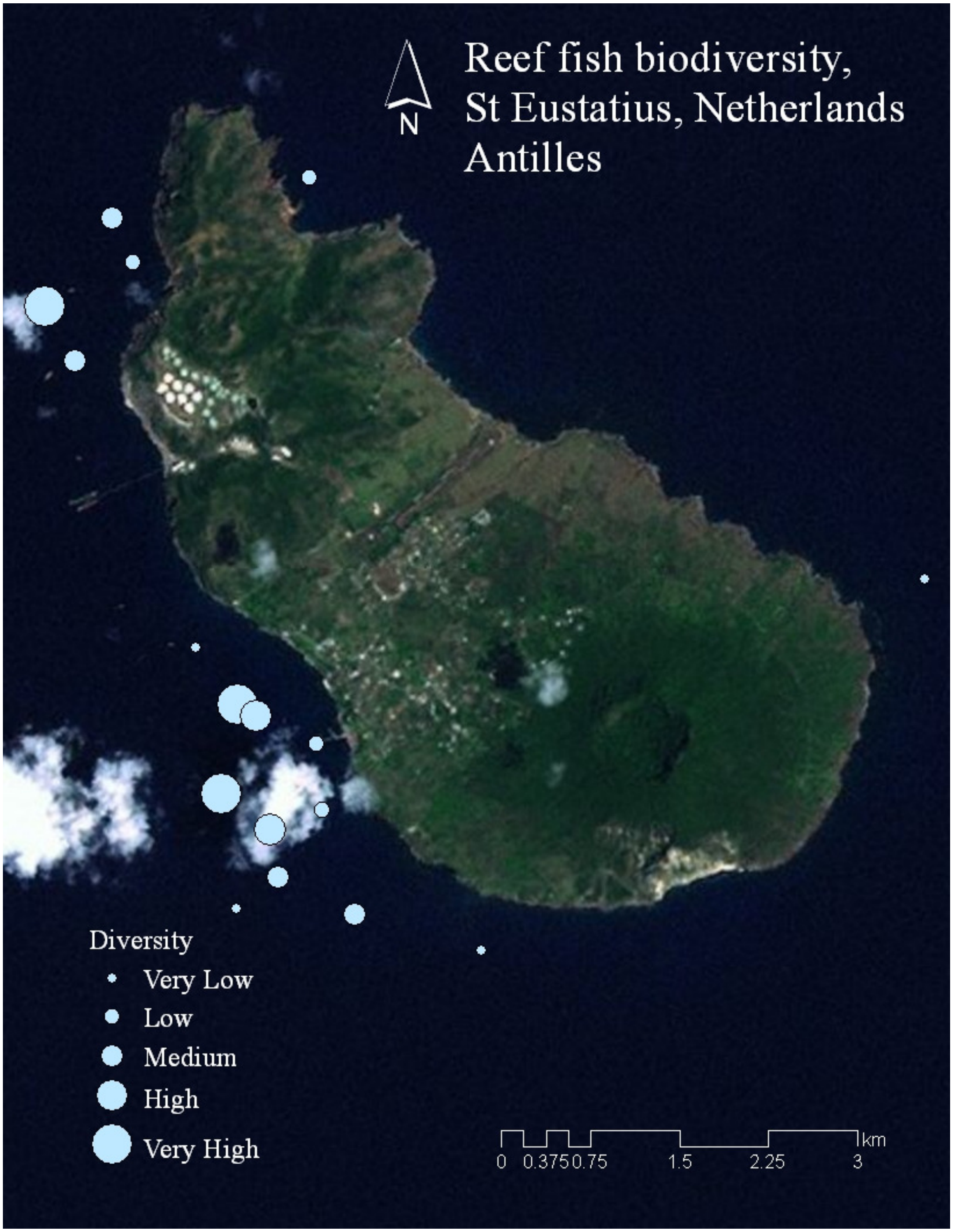

Reef fish biodiversity,
St Eustatius, Netherlands
Antilles
N
Diversity
Very Low
Low
Medium
High
Very High
0
0.375
0.75
1.5
2.25
3
km

Table 1. Presence of fish species at survey sites and overall frequency of occurrence. Survey Sites: 1 Double Wreck, 2 STENAPA, 3 Charles Brown, 4 Miss Cathy, 5 Sting Ray City, 6 Venus Bay, 7 Corre Corre, 8 Aquarium, 9 South of Harbor, 10 Jenkin's Bay, 11 Twin Sisters, 12 Doobie's Crack, 13 Barracuda Reef, 14 Crook's Castle , 15 Blair's Reef, 16 Drop Off, 17 The Blocks
Trophic Levels: C carnivore, P piscivore, MI mobile invertebrate feeder, SI sessile invertebrate feeder, O omnivore, PL planktivore, RH roving herbivore, TH, territorial herbivore.

Species	TL	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	Total	Frequency %
Acanthuridae																				
Acanthurus coeruleus	RH	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	17	100.00
A. chirugus	RH	1	1	1	1	1	1		1			1							8	47.06
A. bahianus	RH	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	17	100.00
Atherinidae																				
Atherinidae	PL			1		1												1	3	17.65
Aulostomidae																				
Aulostomus maculates	C	1	1			1						1	1	1		1	1	1	9	52.94
Balistidae																				
Balistes vetula	SI							1					1			1			3	17.65
Melichthys niger	O							1	1		1	1	1	1	1	1	1	1	10	58.82
Carangidae																				
Caranx ruber	P	1	1	1	1	1	1		1	1	1		1	1	1	1	1	1	15	88.24
C. crysos	P						1					1	1						3	17.65
C. latus	P			1										1					2	11.76
C. lugubris	P			1													1	1	3	17.65
Trachinotus falcatus	MI			1															1	5.88
Chaetodotidae																				
Chaetodon striatus	SI					1						1		1		1		1	5	29.41
C. capistratus	SI							1				1	1	1	1	1	1	1	8	47.06
C. ocellatus	O												1	1					2	11.76
C. aculeatus	SI												1	1		1		1	4	23.53
Cirrhitidae																				
Amblycirrhitus pinos	C					1		1			1	1							4	23.53
Diodontidae																				
Diodon holocanthus	MI							1											1	5.88
D. hystrix	MI	1							1				1						3	17.65
Echeneidae																				
Remora remora			1																1	5.88
Grammatidae																				
Gamma loreto	MI	1	1						1	1				1	1		1		7	41.18
Haemulidae																				
Haemulon flavolineatum	MI	1							1	1		1		1	1	1	1	1	9	52.94

Species	TL	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	Total	Frequency %
H. striatum	MI															1			1	5.88
Anisotremus surinamensis	C					1			1			1			1	1			5	29.41
H. carbonarium	MI	1	1	1					1					1	1	1			7	41.18
H. album	MI	1	1			1						1					1		5	29.41
H. parra	MI	1				1													2	11.76
H. melanurum	MI					1													1	5.88
H. aurolineatum	MI	1	1			1	1				1	1	1			1	1		9	52.94
Holocentridae																				
Holocentrus adscensionis	C	1	1	1	1	1	1	1	1	1		1	1	1		1	1		14	82.35
H. rufus	C		1				1	1		1	1	1	1	1	1	1	1	1	12	70.59
H. marianus	MI										1			1					2	11.76
Myripristis jacobus	C	1	1		1	1			1	1		1	1	1	1	1	1	1	13	76.47
Kyphosidae																				
Kyphosidae	RH															1	1		2	11.76
Labridae																				
Clepticus parrae	MI													1			1		2	11.76
Halichoeres garnotti	MI	1	1			1	1	1	1	1	1	1	1	1	1	1	1	1	15	88.24
Thalassoma bifaciatum	PL	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	17	100.00
H. garnoti	MI					1													1	5.88
H. radiatus	MI									1									1	5.88
H. bivittatus	MI									1									1	5.88
Bodianus rufus	MI	1	1		1	1	1		1	1	1	1	1	1	1		1	1	14	82.35
Lutjanidae																				
Ocyurus chrysurus	C			1	1				1	1			1		1		1		7	41.18
Lutjanus apodus	C				1				1						1	1	1		5	29.41
L. synagris	C	1				1			1		1	1	1						6	35.29
L. griseus	C												1						1	5.88
L. mahogoni	C		1	1		1			1		1	1	1				1		8	47.06
L. analis	C		1																1	5.88
Malacanthidae																				
Malacanthus plumieri	C		1			1		1	1	1		1	1		1				8	47.06
Monacanthidae																				
Cantherhines pullus	O								1		1		1					1	4	23.53
Aluterus scriptus	O											1	1						2	11.76
C. macrocerus	O					1								1		1	1		4	23.53
Mullidae																				
Pseudopeneus maculates	C	1	1		1	1						1	1	1	1	1	1	1	11	64.71

Species	TL	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	Total	Frequency %
Mulloidichthys martinicus	MI	1	1	1	1	1	1		1		1		1	1	1	1	1	1	14	82.35
Opistognathidae																				
Opistognathus aurifrons	PL											1		1					2	11.76
Ostraciidae																				
Lactoprys polygonia	O		1			1			1	1	1		1	1	1	1			9	52.94
L. triqueter	O	1	1	1	1	1			1	1	1	1	1	1	1	1	1	1	15	88.24
Pomacanthidae																				
Pomacanthus paru	O	1		1					1			1	1	1	1		1	1	9	52.94
Holacanthus ciliaris	O		1						1	1				1		1	1		6	35.29
H. tricolor	O					1						1	1		1	1	1	1	7	41.18
Pomacentridae																				
Stegastes partitus	TH	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	16	94.12
Stegastes sp.	TH	1	1			1				1	1	1			1	1		1	9	52.94
Abudefduf saxitilus	O	1	1	1		1			1	1		1	1		1			1	10	58.82
Microspathodon chrysurus	TH	1							1		1			1	1	1		1	7	41.18
Chromis multilineata	PL	1	1		1	1			1	1	1	1	1	1	1		1	1	13	76.47
C. cyanea	PL	1	1		1	1			1	1	1	1	1	1	1	1	1	1	14	82.35
Priacanthidae																				
Heteropriacanthus cruentatus	MI	1	1						1										3	17.65
Scaridae																				
Scarus vetula	RH		1															1	2	11.76
Sparisoma viride	RH			1	1							1		1		1		1	6	35.29
Scarus iserti	RH					1				1		1			1	1	1		6	35.29
Sparisoma aurofrenatum	RH	1	1		1	1	1	1	1	1	1	1	1	1	1	1	1	1	16	94.12
Sparisoma chysopterum	RH									1									1	5.88
Scarus taeniopterus	RH	1	1	1		1		1			1	1	1	1	1	1	1	1	13	76.47
Sciaenidae																				
Equetus punctatus	MI													1					1	5.88
Pareques acuminatus	MI															1			1	5.88
Scombridae																				
Scomberomorus regalis	P											1					1	1	3	17.65
Serranidae																				
Epinephelus striatus	C					1													1	5.88
E. cruentatus	C	1	1			1			1		1	1		1	1	1	1	1	11	64.71

Species	TL	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	Total	Frequency %
Mycteroperca bonaci	C																1		1	5.88
E. guttatus	C	1	1		1	1	1	1				1	1	1	1	1	1	1	13	76.47
E. fulvus	C	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	17	100.00
Serranus tigrinus	MI	1				1		1	1	1	1	1	1	1	1	1		1	12	70.59
S. tabacarius	MI																1		1	5.88
Paranthias furcifer	PL			1										1			1		3	17.65
Rypticus maculates	C				1														1	5.88
Hypoplectrus puella	C													1		1	1	1	4	23.53
H. guttavarius	C															1			1	5.88
Sparidae																				
Calamus calamus	SI											1			1	1	1		4	23.53
Sphyraenidae																				
Sphyraena barracuda	P		1	1		1				1		1		1		1			7	41.18
Synodontidae																				
Synodus sp.	P	1			1					1				1					4	23.53
Tetraodontidae																				
Canthegaster rostrata	O	1	1		1	1		1	1			1	1	1		1	1	1	12	70.59
Muraenidae																				
Gymnothorax moringa	C				1	1			1				1	1		1			6	35.29
G. miliaris	C								1										1	5.88
Myrichthys breviceps	C													1					1	5.88
Dasyatidae																				
Dasyatis americana	C	1				1				1	1		1				1		6	35.29
Carcharhinidae																				
Carcharhinus perezii	C			1									1	1				1	4	23.53
Rhincodontidae																				
Ginglymostoma cirratum	C											1				1			2	11.76
Total Species		38	38	23	24	45	15	19	39	31	29	46	44	49	37	48	46	40		

Fisheries Questionnaire (adapted from White et al 2004)

Nationality:

Occupation 1:

Hours per week:

Years:

Occupation 2:

Hours per week:

Years:

Occupation 3:

Hours per week:

Years:

How long have you lived on Statia?

How many times do you go fishing each week?

How long is each fishing trip?

What type of fishing equipment do you use?

- Boat
- Fish traps
- Lobster traps
- Nets
- Trawling
- Spear guns
- Snorkel
- Scuba
- Hand line
- Other

What types of fish do you aim to catch?

- Reef fish
- Lobster
- Conch
- Pelagic fish

Specifics:

What types of fish would you throw back if caught? Why?

Do you sell your catch or keep them for yourself/family?

Which species are the most valuable to sell?

What are the seasonal changes in fishing activities?

	Jan	Feb	Mar	Apr	May	June	July	Aug	Sept	Oct	Nov	Dec
Fish Type: (Reef fish, Conch or Lobster)												
Gear Used												
Fishing Locations												

Do you participate in any other marine activities around Statia? If yes, what?

Inside Marine Park

Outside Marine Park

What effects, if any, do you think these activities have on the marine environment?

- Less fish
- Physical damage to reef from anchoring
- Pollution (oil, chemicals)
- Sedimentation
- Trash
- Other
- None

What do you think the biggest threat to Statia's marine environment is?

- Fishing
- Anchoring
- Sedimentation
- Pollution
- Trash
- Other
- None

Compared to 10 years ago:

Have you noticed changes in the coral reefs? What?

Has the amount of fish you catch increased, decreased, or stayed the same?

Have you noticed changes in the types of fish you catch?

The size of fish you catch?

Have the quantities of conch and lobster caught increased, decreased, or stayed the same?

Has the size of conch or lobster changed?

Have you had to change the areas that you fish in? If yes, why?

Do you have any thoughts to why these things have changed?

Have you changed any of your activities in response to the above changes?