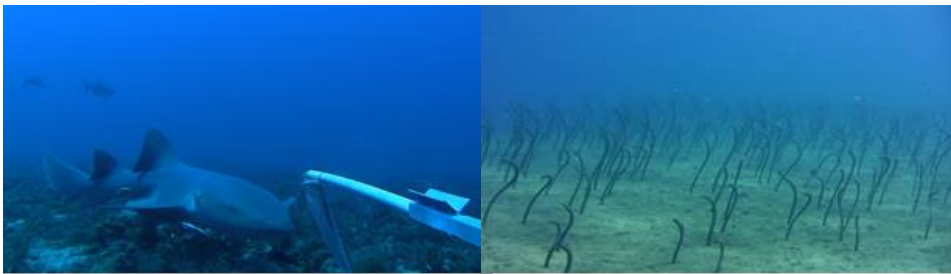


Baseline assessment of the coral reef fish assemblages of St. Eustatius

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Abstract

The shallow marine habitats surrounding St. Eustatius fulfil critical ecosystem services in terms of fishery production, recreation, dive tourism and coastal protection. In this a key role is played by the fish communities. In this report we document the relative finfish community composition, density and distribution in the shallow coastal waters of the St. Eustatius Marine Park based on 104 baited video stations distributed among two management sectors and five habitat classes ranging in depths from 8-30 m. In doing so we introduced to the Dutch Caribbean the use of a technology-based method that provides more precise length data than common visual transects, is applicable to wide range of depths and sampling conditions, and is more sensitive for detecting and monitoring apex (top) predatory fish species such as shark.

Compared to earlier survey results our findings highlight the virtual disappearance of large grouper species from the reefs of St. Eustatius. The natural absence of mangrove nursery habitat is one key driver of fish community structure in St. Eustatius and the loss of former seagrass beds is a second key factor probably accounting for the lack of typical mangrove and seagrass-associated scarids (e.g. *Scarus coeruleus* and *guacamaia*), snappers (e.g. *Lutjanus apodus* and *griseus*) and grunts (e.g. *Haemulon sciurus*).

The most important local determinant of fish community structure was found to be habitat three-dimensional structure while the measured effect of designated fishing reserve zones was much less pronounced. Nevertheless, mean overall fish size was slightly higher in the fishing reserves. However, our community sampling was insufficient to meaningfully compare densities and size structure of main commercial target species. Community abundance of planktivores and herbivores were notably inversely related, with low-structured sandy habitat being dominated by planktivores and higher-structured hard-bottoms being dominated by herbivores. Low-structured habitat which offered little critical shelter to small fish had the highest mean fish size of all five habitats.

Our results indicate a relatively low quantifiable effect of the present fishing reserves. This may have to do with either or a combination of a) low finfish fishing effort and/or low fishing selectivity, b) problems in the enforcement of the fishing reserves or, c) geographic scale issues due to the movement of fish between defined zones which act to blur potential effects of stated management regimes, and finally our sampling design as a fish community baseline lacking focus on targeted commercial species. **Further directed research is needed to properly evaluate and enhance the functioning of the marine park reserves which are accorded an important role in the future socio-economic development of St. Eustatius.**

The reefs of St. Eustatius are characterised by very low levels of three-dimensional structure (Risk's Index: 1-1.3, see Debrot et al. 2014), which was found to be the most important local determinant of fish abundance and distribution. The potential for habitat enhancement to jointly help achieve fishery and conservation goals seems evident. **We recommend that measures to enhance such three-dimensional structure may be useful to help increase fish abundance to the benefit of both fishing and biodiversity stakeholders such as the conservation, dive and tourism sectors.**

The relatively high presence of sharks (Caribbean reef sharks and nurse sharks) around St. Eustatius is encouraging in the context of conservation, valuable for dive tourism, and interesting for research. As top predators, these sharks play an important ecological role in healthy reefs and their higher abundance around St Eustatius compared to most other areas of the Caribbean may contribute to and be a useful indicator of overall coastal ecosystem health. **Further studies of these important species are called for.**

Our assessment of the use of the BRUV method for fish community assessment showed that, due to the deployment strategy used, our test power to detect changes in both numerical and species changes in the communities studies was relatively low. **A combination of extensive BRUV surveys (once every 3 years) in combination with yearly fish surveys (i.e. protocol Global Coral Reef Monitoring Network) at a small number of permanent sites with high structural complexity is recommended to ensure the timely detection of trends in reef fish populations.**

1. Introduction

Quantitative insight into coral reef fish stocks, and factors governing their abundance, species and size composition are key to understanding the provision of ecosystem services, and consequently also provide an essential underpinning for marine spatial planning, management and conservation. Fish communities exert important herbivory that is critical to protect coral reefs from overgrowth by macro-algae. Predation within the fish community helps maintain ecosystem balance by exerting top down control and help direct productivity up into the food chain and into harvestable fish for consumption. Healthy fish communities with abundant and large fish finally play a decisive role in supporting dive tourism and the quality of recreation. Essential background information on fish communities is either lacking or outdated for most of the Dutch Caribbean (islands of Aruba, Bonaire, Curaçao, Saba, St. Eustatius, St. Maarten and the Saba Bank)(Debrot and Sybesma 2000). Therefore, the need for baseline descriptive marine resource assessment has been highlighted as a priority within the EEZ management plan for the Caribbean Netherlands (Meesters et al. 2010) and forms an essential part of the implementation of the latest Nature Management Plan for the Caribbean Netherlands (MinEZ 2013). Consequently, several studies that quantify habitat diversity and associated fish stocks throughout the Dutch Caribbean, including the Saba Bank (Toller et al. 2010, Stoffers 2014), Lac Bay Bonaire (Davaasuren and Meesters 2012, Debrot et al. 2012) and Saba (van Looijengoed 2013, van Beek et al., in prep.) have recently been undertaken.

The abundance and quality of coral reef fish populations critically contribute to the total economic value of nature for the island of St. Eustatius (Tieskens et al. 2014). Bervoets (2010) estimated the economic value of the island's coral reef resources to be roughly USD 11 million per year and concluded that active coral reef conservation and research were a priority for sustainable economic development. More recently, van der Lely et al. (2014) estimated the total economic value of nature (TEV) (both marine and terrestrial) for the island at USD 25.2 million per year but declining in the future. Fish populations are of importance as a basis for the growing dive industry as well as for the small scale fishing that takes place in island waters. The principal target species of St. Eustatius at present are mixed reef fish (annual landing 5 t), the Caribbean Spiny lobster (*Panulirus argus*; annual landing 4 t), and the Queen conch (*Lobatus gigas*; annual landing 2 t) (Poiesz 2013, Meijer zu Schlochtern 2014, de Graaf et al. 2014). Reef fish are mainly caught in fish traps and the most common reef fish species (in number) in the catch of the artisanal, coastal fishery are Acanthuridae (3 species 35%), two small grouper species, coney (*Cephalopholis fulva* 11%) and red hind (*Epinephelus guttatus* 11%), honey combed cowfish (*Lactophrys triqueter* 9%) and squirrel fish (*Holocentrus adscensionis* 6%).

Only five baseline studies on finfish stocks of St. Eustatius have been conducted before the start of the Caribbean Netherlands fish and fisheries research programme in 2012: a non-quantitative inventory by Sybesma et al. (1993) which formed the basis for the legal establishment of the St. Eustatius Marine Park four years later in 1997, a quantitative assessment of coral reef fish populations at 10 shallow water sites in the Southern Reserve in 1999 (Klomp and Kooistra 2003), a quick assessment of catches by the local fishery (Dilrosun 2004) and two underwater visual census (UVC) surveys using SCUBA of fish populations at the leeward side of the island (White et al. 2006, McClellan 2009). McClellan (2009) compared fish populations inside and outside the fishery reserve but was unable to make any firm conclusions with respect to the effectiveness of the reserves because of the confounding effects of habitat differences. A more extensive quantitative assessment of fish abundance and distribution in the waters around St. Eustatia was thus required.

In this study we present a fishery independent survey of the finfish populations at depths ranging from 5-30 m in the shallow reef habitats around the island of St. Eustatius.

Such baseline information is a critically needed prerequisite to help address a number of important management questions which include: a) what is the current status and recent developments of reef fish stocks?, b) which, if any, target species are possibly experiencing overfishing? C) can the effects of current marine park zoning be discerned?, d) what is the trophic structure of fish populations of St. Eustatius?, e) what are the key drivers of fish community structure operating around St. Eustatius and at various spatial scales (e.g. fishing mortality, habitat quality, availability of nursery habitat, productivity, connectivity)? , and finally, f) what measures might be taken to stimulate finfish productivity?

Underwater visual census (UVC) surveys using SCUBA are limited to safe diving depths, by weather conditions, and are affected by several sources of bias and imprecision (Willis 2001). Considering the large aerial coverage and diversity of reef-associated habitats, depths and sea conditions, that apply to the biological marine resources of the Dutch Caribbean, there was a strong need to use methods that allow better standardization and which are applicable to a wider range of depths and field conditions. Therefore, a key goal of this work was to also introduce a new technology-based survey methodology for greater precision, efficiency and standardization, and broader applicability to a range of depths and environments in the Dutch Caribbean EEZ. For this we chose stereo Baited Remote Underwater Video (stereo BRUV) (Watson et al. 2005). This relatively new method is gradually being applied more widely and the expanding literature documents the relative advantages, disadvantages, biases and implications on BRUV-derived data compared to other sampling techniques (Cappo et al. 2004, Watson et al. 2005, Shortis et al. 2007, Colton and Swearer 2010, Langlois et al. 2010, Brooks et al. 2011, Harvey et al. 2012). One of the key advantages of stereo-BRUV over diver-based methodologies is that it is more likely to detect the larger predatory fish species such as sharks (Watson et al. 2005, Harvey et al. 2007b) which may be repelled in the presence of divers and not detected during traditional visual surveys (Chapman and Atkinson 1986, Willis and Babcock 2000). Large predators are relatively less abundant than many other guilds in coral reef ecosystems but very important as indicators of ecosystem health and also very sensitive to ubiquitous fishing pressure. This makes methods that are sensitive to this crucial component of the finfish community extra valuable for long-term and precise monitoring of community change.

In this report we present the results of the BRUV survey focussing on the effect of habitat complexity and management zoning on abundance (number and biomass), fish size and species richness of coral reef fishes and sharks.

The three main objectives were to:

- Quantify and document fish communities in terms of parameters and indices typically used for fish population and community assessment as an ecological baseline for future monitoring.
- Use the parameters to describe, compare and contrast fish communities in order to evaluate the effects of habitat and fishery-reserve status on fish distributions around St. Eustatius.
- Evaluate our use of the BRUV technique for fish community assessment and make recommendations for its further applicability and use.

2. Methods

2.1 Study area

The island of St. Eustatius lies in the north-eastern Caribbean between 17°28' and 17°32' N latitude and 62°56' and 63°0' W longitude. Its nearest neighbouring islands are Saba (27 km to the northwest) and St. Kitts (12 km to the east). The surface area of the island is 21 km² and the island is part of the inner arc of the Lesser Antilles (Freitas et al. 2014).

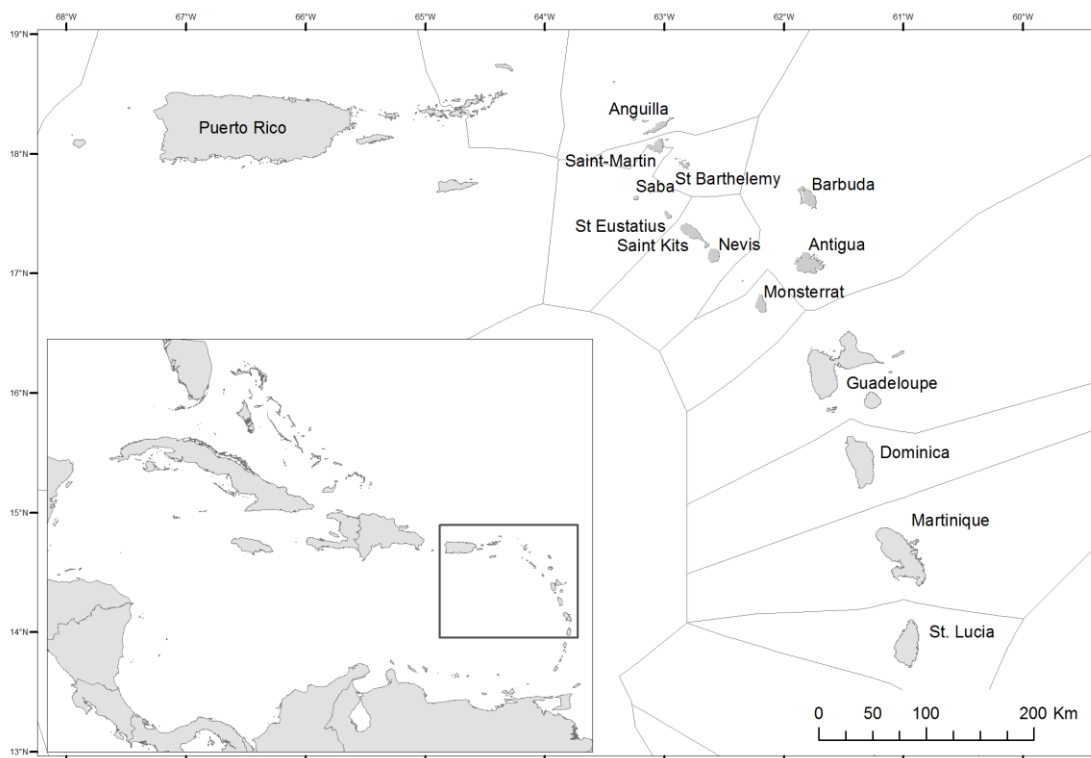


Figure 1. Geographic location of St Eustatius in the Wider Caribbean region.

All marine habitat from the low tide level out to the depth of 30 m is legally designated as the St. Eustatius Marine Park by means of the St. Eustatius Marine Environmental Ordinance (AB 1996, No. 3) (McRae and Esteban 2007). The total surface area of the marine park is 27.5 km². The park is managed by STENAPA (St. Eustatius National Parks Foundation), which also manages two terrestrial parks on the island. Inside the St. Eustatius Marine Park, two reserves have been designated in which no fishing or anchoring is allowed (Figure. 1). Habitats include coral reefs (drop off walls, volcanic 'fingers' and 'bombs'), 18th century shipwrecks and artificial reefs. The island's marine environment is home to, and migratory stop-over or breeding site for four IUCN Red List Species, 10 CITES Appendix I species and 98 Appendix II species (McRae and Esteban 2007).

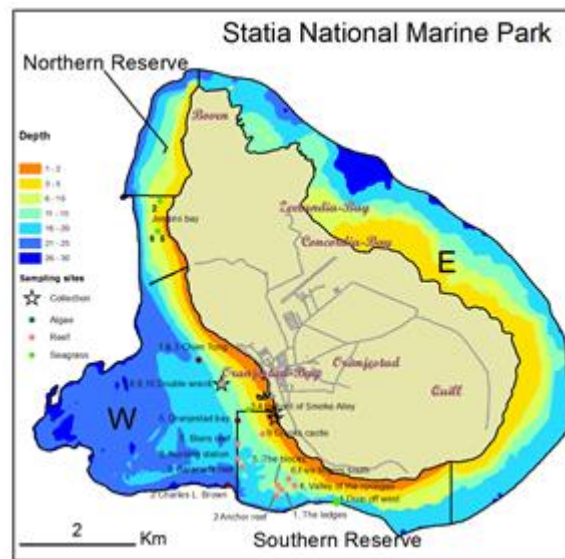


Figure 2. Bathymetric map of the St. Eustatius Marine Park, showing key landmarks (source: STENAPA).

The benthic habitats of St. Eustatius have been recently mapped by Debrot et al. (2014) which allows certain useful generalizations. The densest coral and seagrass habitats of St. Eustatius are concentrated at depths of about 24 m and only amounted to 4 and 5 percent resp. of the island shelf habitats studied. Whereas coral areas are essentially limited to the southern and south-western island shelf areas, seagrass beds are confined to the northern island shelf area. Including patch reef habitats, total hard coral-scape habitat for the St. Eustatius Marine Park amounts to about 19% of the area surveyed and about 475 ha of habitat. Significant coral mortalities and declines have been documented over the last 15 years, while macro-algal cover appears to have increased).

2.2 Field technique: BRUV

The field data for this study was collected between March and August, 2013. For this we used the technique known as "Baited Remote Underwater Video"(BRUV) from here on referred to as BRUV (Watson et al. 2005). With this video technique two Canon HFG10 cameras are mounted in waterproof housings on a steel frame (Figure 3). The two cameras enable the recording of fish from different angles, which allows for stereo-viewing and thus accurate estimation of individual fish lengths. In this study a total of two steel frames with two cameras each were used. For each video session four sardines (*Sardinops sp*) were perforated to enhance the spread of their smell and used as bait to attract fish. The camera frames were gently lowered to the bottom and left to record for a minimum of 50 minutes, after which they were pulled up and the memory (SD) cards retrieved. For a field protocol on BRUV procedures see van Kuijk (2013).

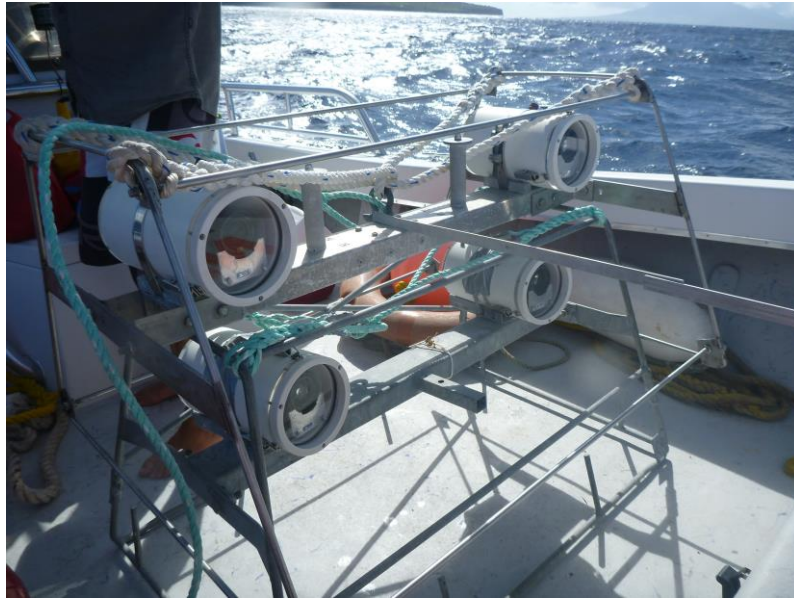


Figure 3. Two Baited Remote (stereo) Underwater Video setups ready for deployment (ie. dropping).

2.3 Sampling distribution and design

To spread the number of drops equally between fishing and reserve areas, the drops were divided over the four sectors representing fished and unfished zones as shown in figure 2. In total a number of 104 drops were performed (tables 1 and 2). In order to avoid overlap of the sampling area of two simultaneous drops due to bait plume interaction, a minimum distance of 500 meters was used between all drops on the same date. When a frame was dropped the GPS coordinates and depth at that point were measured. The location of all drops can be seen in figure 4.

Table 1: Number of BRUV deployments per management zone.

Management Zone	No. BRUV drops
Northern Reserve (NR)	16
Southern Reserve (SR)	31
Western Fishing Zone (WFZ)	19
Eastern Fishing Zone (EFS)	38
Total	104

Table 2. Number of BRUV drops per habitat category in the Marine Reserves and Fishing zones within the boundaries of the four sectors of the SNMP.

Habitat Category	Marine Reserves (NR & SR)	Fishing Zone (EFZ & WFZ)	Total
0	12	12	24
1	6	25	31
2	12	19	31
3	14	1	15
4	3	0	3
Total	47	57	104

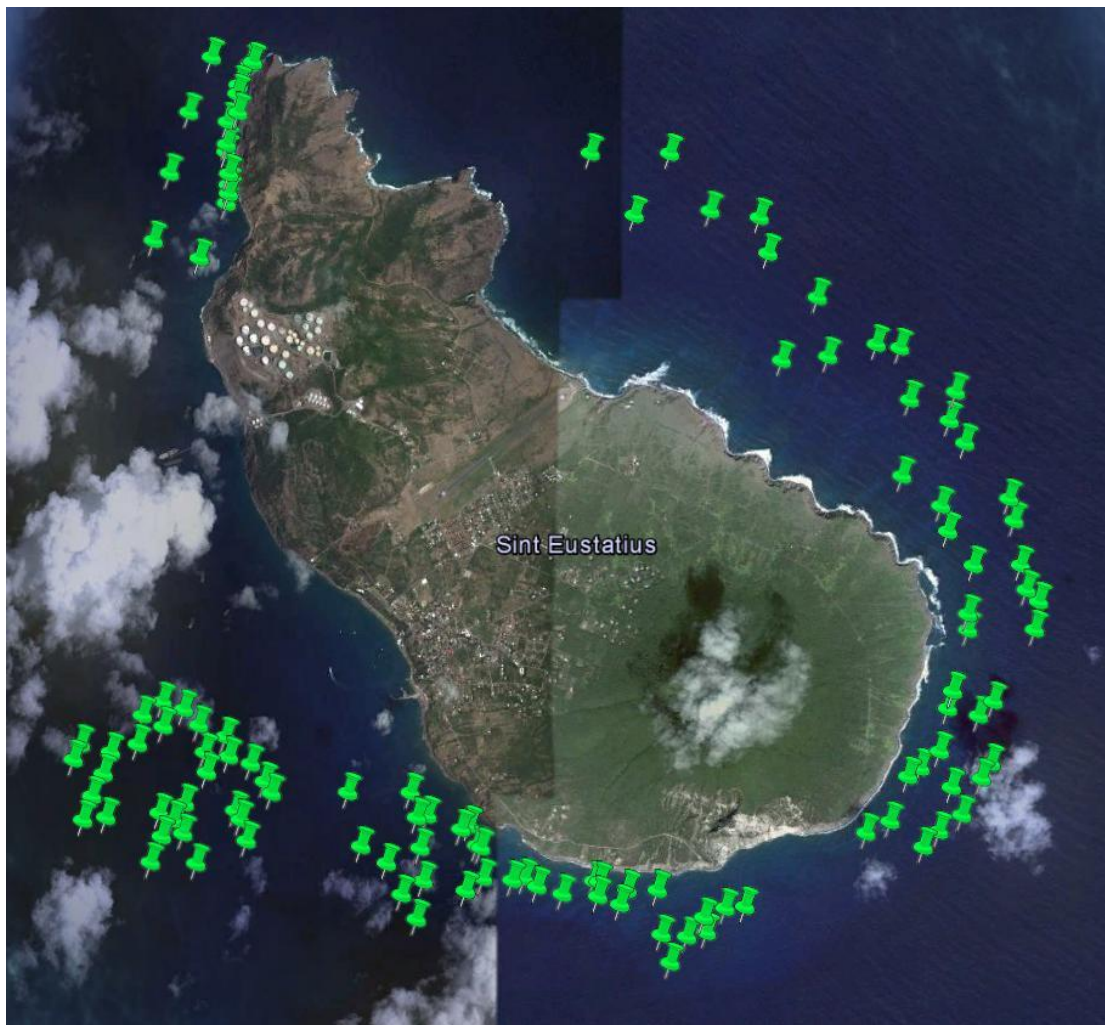


Figure 4. Map of Sint Eustatius showing the locations of all BRUV drops around the island.

2.4 Video analysis

2.4.1 Data extraction

From each drop, 45 minutes of video footage was analysed using the computer program Event Measure 3.5 (www.seagis.com.au/event.html), counting from the moment the cameras landed on the bottom. In this program the footage of a single drop from both the left and right camera from one steel frame can be synchronized and displayed simultaneously. Since the cameras are mounted on the steel frames under a slightly different angle, it is possible to use the computer program "Event Measure" to calculate both the distance of a fish from the camera and measure its length, provided that the fish is fully visible to both cameras (Cappo et al. 2006). The video footage was analysed by identifying all fish species within a distance of 8 meters from the cameras. We used Humann and Deloach (2002) for species identification and consulted with experts in cases of doubt. The maximum number of individuals visible in a single movie frame was counted for all life stages of all species, and added up to obtain the abundance value (maxN) of each separate species. Although the total number of individuals per species might be underestimated by counting only individuals visible in a single frame, any repeated measurements of the same individuals were avoided. Since individuals from the same species in different life stages are certainly different individuals, their abundance was combined to calculate the total relative abundance for those species per drop. The maxN values thus obtained therefore represent a measure of relative abundance and will from this point on be referred to as "abundance data". In addition to abundance, per species, the fork or total length (depending on the anatomy of the species) was measured in mm for the highest number of individuals possible that were visible in a single movie frame. Because of their importance as top predators, species of sharks were measured even if they were beyond 8 meters away from the cameras. Also for sharks maxN was based on the total number of sharks observed during a single drop rather than the maximum number of individuals on a single movie frame, while repeated measures were prevented by measuring each sharks size and removing repeated counts of the same-sized individual.

2.4.2 Habitat categories

A screenshot was also taken from the left camera footage of each drop in order to assign habitat type using a 6 point scale as described by Polunin and Roberts (1993). This scale was used to describe habitat spatial structure, ranging from habitat category 0, bare substratum, to category 5, exceptionally complex with high coral cover, numerous caves and overhangs (Figure 5). For habitat images and GPS coordinates for all drop sites, we refer to van Kuijk (2013).






Habitat images					
6 point scale (Polunin 1993)	0=bare substratum	1=low and sparse relief	2=low but widespread relief	3=moderate complexity	4=high complexity with cave systems

Figure 5. The first five habitat categories of the 6 point scale described by Polunin 1993, with BRUV screenshots for visual reference.

In the BRUV-footage, habitats from categories 0 to 4 of this scale were observed, as shown in Figure 5. For habitat categories in which less than 5 drops were performed in either the marine reserves or the fishing zones, the data was statistically analysed but not shown in most graphs since these mean values and confidence intervals were not meaningful due to the low number of samples.

2.5 Data analysis

Because pelagic schooling species do not inhabit reefs, they are typically excluded from analyses of coral reef fish community structure (e.g. Hylkema et al. 2014). Species from the following families were excluded. From the family Myliobatidae: *Aetobatus narinari*; Carangidae: *Caranx latus*, *Caranx lugubris*, *Decapterus macarellus*, *Seriola rivoliana* and *Trachinotus falcatus*; Echeneidae: *Echeneis naucrates*; Kyphosidae: *Kyphosus sectatrix* and *K. incisor*, and from Scombridae: *Scomberomorus regalis*. Because sharks are important top predators associated with healthy reefs and are known to be highly mobile, species of sharks were analysed separately (Robbins et al. 2006, Bond et al. 2012). The maxN data and length data obtained by means of the computer program "Event Measure" were combined to calculate average and total biomass, using biomass calculation factors taken from Fishbase.org (Froese and Pauly 2006, www.fishbase.org, van Kuijk 2013). When length measurements were not available for some individuals of a species encountered in a drop, the mean weight of the individuals of that species that did have length data or a mean length was calculated or approximated and assigned to those individuals without length measurements in order to calculate total biomass. One common species, *Heteroconger longissimus*, is a burrowing species and was never recorded fully outside its burrow in this study. Therefore, no length data could be obtained for this species. Trophic levels were assigned to each fish species as based on Fishbase.org (van Kuijk 2013). The number of individuals was multiplied by the trophic level assigned to their species, after which this data was averaged to calculate the mean trophic level of the fish assemblages per drop (Pauly et al. 1998). Shannon's H diversity was calculated in the statistical computing program R (v. 2.15.2, <http://www.r-project.org/>) with the package vegan, using the maxN abundance dataset from Event Measure.

2.6 Community and taxon indices and parameters

The fish counts measurements and identifications based on video and data analysis allowed the calculation of a number of community and taxon indices and parameters for each habitat class as typically used for fish population assessments. These were: fish numerical density (nMax/drop), fish biomass density (gMax/drop) and fish size (mm), (all as per species, family, trophic guild and total community), species richness (S), and finally Shannon-Weaver diversity index (H').

2.7 Sampling adequacy: species accumulation curves and power analysis

Species accumulation curves can be used to assess the adequacy of the sample size for community description and as a means to estimate the total expected number of species for a habitat. Species accumulation curves were plotted for all samples and per habitat category with the computing program R with the package vegan. The function used to plot species accumulation curves in R finds the mean number of species and automatically gives the corresponding standard deviations as output, using a subsampling without replacement method (Ugland et al. 2003, Colwell et al. 2004, Kindt et al. 2006). With this method standard deviations are based on estimations of the total (extrapolated) number of species and are not conditioned on the dataset, giving standard deviations > 0 for all samples.

Power analysis was performed to calculate the number of samples (drops) needed in future studies in order to detect a minimum change of 25% in species richness or fish abundance in numbers. Such changes would be ecologically very important. The number of samples required to obtain a power of at least 0.8 (Cohen 1992) for detecting changes was calculated with the program G*Power (version 3.1.7) (Erdfelder et al. 1996) using mean values and standard deviations for species richness and number of fish. Because the variability in species richness and number of individuals differs between both habitat structure and fishing and reserve zones, the data was pooled for habitat structures and fishing zones.

Since the data is not normally distributed, power was calculated according to a non-parametric test: the two-sided Wilcoxon-Mann-Whitney test (Feltovich 2003).

2.8 PERMANOVA and ANOVA

To test for the effects of habitat structure, fishing zone and their interaction on the various dependent variables we used permutational analysis of variance (PERMANOVA) (Anderson 2001) was applied in the computing program R with the package *vegan*. The model used to perform the PERMANOVA was:

$$Y = \text{habitat structure} + \text{marine protection} + \text{habitat structure} * \text{marine protection} + \text{error}$$

for each variable separately. A Bray-Curtis dissimilarity matrix was first created from the dataset, after which PERMANOVA computes 999 permutations of the data by default to obtain P values. Since the PERMANOVA test can only reveal the presence or absence of a significant effect from habitat structure and marine protection on the entire fish community, the use of community structure visualization techniques is essential for further understanding of these effects. For this we chose NMDS. Additional analysis was done on abundance and length data for the families Acanthuridae, Scaridae and Serranidae, based on both their role in the reef ecosystem and their importance as commercial species (Polunin and Roberts 1993, Sluka et al. 1998, DeMartini et al. 2008, Ault et al. 2013). This was performed using an analysis of variance (ANOVA) in the program SPSS (IBM SPSS Statistics version 20). All species were pooled for the families Acanthuridae and Scaridae, while for the Serranidae only the species *Epinephelus guttatus*, *Cephalopholis fulva*, *Cephalopholis cruentata* and (rare juvenile) *Mycteroperca venenosa* were combined as the other smaller species were not caught by the fishermen (Poiesz 2013). Besides analysis on the family level, the specific species *Acanthurus coeruleus*, *Acanthurus tractus*, *Balistes vetula*, *Cephalopholis fulva*, *Epinephelus guttatus* and *Scarus taeniopterus* were also investigated since they are the species most often landed by fishermen (Poiesz 2013) or have a high economic value due to their larger size. These datasets were taken as the dependent variables, with the factors habitat, fishing zone and the interaction between these factors as fixed effects. A Tukey-Kramer post hoc test was applied to detect which groups differed statistically, and 95% confidence intervals were calculated.

2.9 Non-metric MultiDimensional Scaling (NMDS)

We used non-metric multidimensional scaling (NMDS), to visualize variation between fish assemblages (Minchin, 1987). Ordination prior to NMDS plotting was done using a Bray-Curtis dissimilarity matrix based on (usually square root) transformed abundance data (Faith et al. 1987). The Bray-Curtis dissimilarity measure is widely used in ecology for ordination methods (Minchin 1987; Legendre and Gallagher 2001) and emphasizes the difference in abundance between samples (Anderson and Willis 2003). The advantage of this is that more numerically abundant species weigh heavier than very rare species (Field et al. 1982). The dissimilarity matrix is used to map points from the multidimensional species abundance data onto a lower dimensional space, where the distance between points are in proportion to their dissimilarity in species composition (De'ath 1999). The resulting plots visualize the difference in species composition between samples, making it easier to detect patterns in community structure. In this study the statistical computing program R was used, with the package *vegan*, in which the function automatically square root transforms the relative abundance data and then calculates a Bray-Curtis dissimilarity matrix to create two-dimensional NMDS plots.

3. Results

3.1 Sampling adequacy: species accumulation curves and power analysis

For the more-detailed assessments on sampling adequacy we refer to Appendix 1. Only the main points are summarized here:

- The rate at which new species were encountered during successive drops differed between habitat categories.
- The lowest level of structure shows a slow accumulation of species with an increasing number of drop samples but the species accumulation curves of the other habitat categories rise quickly with the first few drops but levels off as the number of drops increase.
- When aiming to detect a 25% change in species richness with a power of 0.8, a minimum of 45, 30, 15 and 20 drops are needed for habitat categories 1, 2, 3 and 4, respectively, whereas even 95 drops are insufficient to characterize the fish community of habitat category 0. To detect a change of 25% in total fish numerical abundance for each habitat category, power was even lower where the number of drops needed even exceeds 65. Our chosen sampling design is not adequate for detection of significant changes in abundance. However, our sampling design was meant to give a broad indication of fish communities and distribution.

Clearly, for long-term monitoring of ecological change, a sampling strategy using repeated measures from a smaller number of selected high-diversity permanent monitoring sites is to be recommended.

3.2 Fish species and numerical abundance

In total, 4732 fish were recorded, belonging to 107 species divided over 37 families (Appendix 2, including sharks). A synopsis of community description indicating the five most abundant species and key biodiversity descriptors for the different habitat and management zone categories are given in table 3.

Table 3. List of key parameters for each habitat category in both reserve and fishing zones.

Reserve zone		Habitat category 1 (N=6)		Habitat category 2 (N=12)		Habitat category 3 (N=14)		Habitat category 4 (N=3)	
	Habitat category 0 (N=12)								
Total # individuals	236	151		575		891		207	
Total # species	12	36		69		66		35	
Total # families	12	23		27		26		18	
Total biomass (g)	38325	14674		85239		124953		12845	
Top five species									
(# individuals)	<i>Heteroconger longissimus</i>	<i>Thalassoma bifasciatum</i>		<i>Thalassoma bifasciatum</i>		<i>Thalassoma bifasciatum</i>		<i>Thalassoma bifasciatum</i>	
	<i>Caranx ruber</i>	<i>Cephalopholis fulva</i>		<i>Stegastes partitus</i>		<i>Acanthurus coeruleus</i>		<i>Scarus taeniopterus</i>	
	<i>Xyrichtys martinicensis</i>	<i>Acanthurus tractus</i>		<i>Chromis cyanea</i>		<i>Melichthys niger</i>		<i>Acanthurus tractus</i>	
	<i>Sphyraena barracuda</i>	<i>Opistognathus aurifrons</i>		<i>Cephalopholis fulva</i>		<i>Stegastes partitus</i>		<i>Cephalopholis fulva</i>	
	<i>Malacanthus plumieri</i>	<i>Caranx ruber</i>		<i>Scarus taeniopterus</i>		<i>Chromis cyanea</i>		<i>Acanthurus coeruleus</i>	
(biomass)	<i>Sphyraena barracuda</i>	<i>Dasyatis americana</i>		<i>Mycteroperca venenosa</i>		<i>Sphyraena barracuda</i>		<i>Pomacanthus paru</i>	
	<i>Caranx ruber</i>	<i>Epinephelus guttatus</i>		<i>Epinephelus guttatus</i>		<i>Acanthurus coeruleus</i>		<i>Halichoeres garnoti</i>	
	<i>Dasyatis americana</i>	<i>Sparisoma aurofrenatum</i>		<i>Cephalopholis fulva</i>		<i>Melichthys niger</i>		<i>Cantherhines macrocerus</i>	
	<i>Xyrichtys martinicensis</i>	<i>Cephalopholis fulva</i>		<i>Sparisoma aurofrenatum</i>		<i>Sparisoma aurofrenatum</i>		<i>Cephalopholis fulva</i>	
	<i>Dactylopterus volitans</i>	<i>Caranx ruber</i>		<i>Haemulon album</i>		<i>Mycteroperca venenosa</i>		<i>Acanthurus tractus</i>	
Fishing zone		Habitat category 1 (N=25)		Habitat category 2 (N=19)		Habitat category 3 (N=1)		Grand total (N=104)	
	Habitat category 0 (N=12)								
Total # individuals	225	1239		1165		43		4732	
Total # species	35	65		57		18		107	
Total # families	17	29		24		11		37	
Total biomass (g)	38448	147171		132702		4447		598803	
Top five species									
(# individuals)	<i>Caranx ruber</i>	<i>Stegastes partitus</i>		<i>Thalassoma bifasciatum</i>		<i>Thalassoma bifasciatum</i>		<i>Thalassoma bifasciatum</i>	
	<i>Stegastes partitus</i>	<i>Thalassoma bifasciatum</i>		<i>Scarus taeniopterus</i>		<i>Stegastes partitus</i>		<i>Stegastes partitus</i>	
	<i>Xyrichtys martinicensis</i>	<i>Caranx ruber</i>		<i>Stegastes partitus</i>		<i>Cephalopholis fulva</i>		<i>Scarus taeniopterus</i>	
	<i>Acanthurus tractus</i>	<i>Opistognathus aurifrons</i>		<i>Chromis cyanea</i>		<i>Melichthys niger</i>		<i>Caranx ruber</i>	
	<i>Thalassoma bifasciatum</i>	<i>Halichoeres garnoti</i>		<i>Acanthurus tractus</i>		<i>Halichoeres garnoti</i>		<i>Acanthurus tractus</i>	
(biomass)	<i>Caranx ruber</i>	<i>Sphyraena barracuda</i>		<i>Sparisoma aurofrenatum</i>		<i>Halichoeres cyanocephalus</i>		<i>Caranx ruber</i>	
	<i>Sphyraena barracuda</i>	<i>Caranx ruber</i>		<i>Epinephelus guttatus</i>		<i>Cephalopholis fulva</i>		<i>Epinephelus guttatus</i>	
	<i>Balistes vetula</i>	<i>Balistes vetula</i>		<i>Balistes vetula</i>		<i>Sparisoma aurofrenatum</i>		<i>Sparisoma aurofrenatum</i>	
	<i>Acanthurus tractus</i>	<i>Epinephelus guttatus</i>		<i>Sphyraena barracuda</i>		<i>Melichthys niger</i>		<i>Balistes vetula</i>	
	<i>Xyrichtys martinicensis</i>	<i>Cephalopholis fulva</i>		<i>Cephalopholis fulva</i>		<i>Epinephelus guttatus</i>		<i>Cephalopholis fulva</i>	

When comparing number of species between the fishing and reserve zones (Figure 6), the mean values for habitat categories 1 and 2 are higher in the fishing zone than in the reserve zone, but slightly lower in category 0 (Table 4; pseudo-F=6.45 and P=0.001). With the exception of habitat category 1 in the reserves, the number of fish increased with the level of habitat structure in both the fishing and reserve zones. While statistical analysis reveals a significant effect of habitat on abundance (pseudo-F=21.8, P=0.001), no significant interaction was found between the factors habitat structure and management zone category.

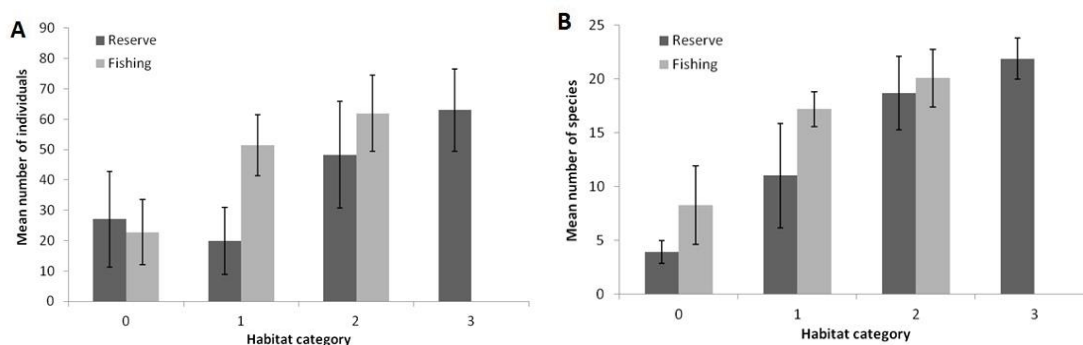


Figure 6. The mean number (\pm 95% CI) of a) individual fish and b) species per habitat category for the reserve and fishing zones.

3.3 Overall mean fish length

As shown in figure 7, overall mean fish length also differs significantly between the reserve and fishing zones, with higher values for mean length in the reserves for all categories (PerMANOVA, $F=7.00$, $P=0.001$). The effect of habitat on overall mean fish length was also found to be significant (pseudo- $F=21.31$, $P=0.001$; Table 4). The difference in overall mean fish size between fishing and reserve zones could not be ascribed to any size differences in the principal fish families (Acanthuridae, Scaridae, Serranidae; Table 5). The Scaridae did show significant size differences between habitats (Table 5). Appendix 2 shows that this was not importantly due to differences in species composition and that means that areas outside the reserves simply had more juveniles than inside the reserves. This is probably due to juveniles preferably using the seagrass and algae beds (as nursery habitat). When comparing mean length between the habitat categories, the lowest habitat category has the highest mean length value in the reserve zones, while all other habitats show little difference.

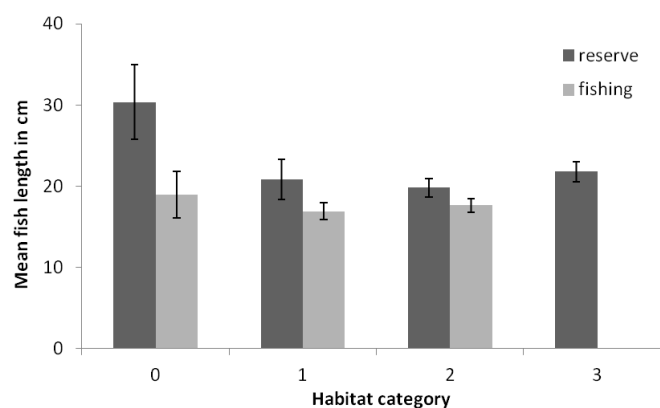


Figure. 7. Mean (\pm 95% CI) fish length for all habitat categories in both the reserve and fishing zones.

3.4 Mean lengths of fishery target species

While comparisons of overall mean fish size may provide insight into community processes, more interesting from the management perspective is the question how the size of fishery target species may differ between areas inside and outside the fishing reserves. This issue was also thus also explored (Fig. 8).

The results suffer from small sample sizes as our baseline study was not specifically designed to address this question. For the largest species with highest market value (red hind, coney and queen triggerfish, no size differences could be detected. For the ocean surgeonfish and the princess parrotfish a small but significant difference was found in mean fish size between the fishing reserves and non-reserve areas. There is no evidence to believe that these small differences are caused by any purported difference in fishing pressure.

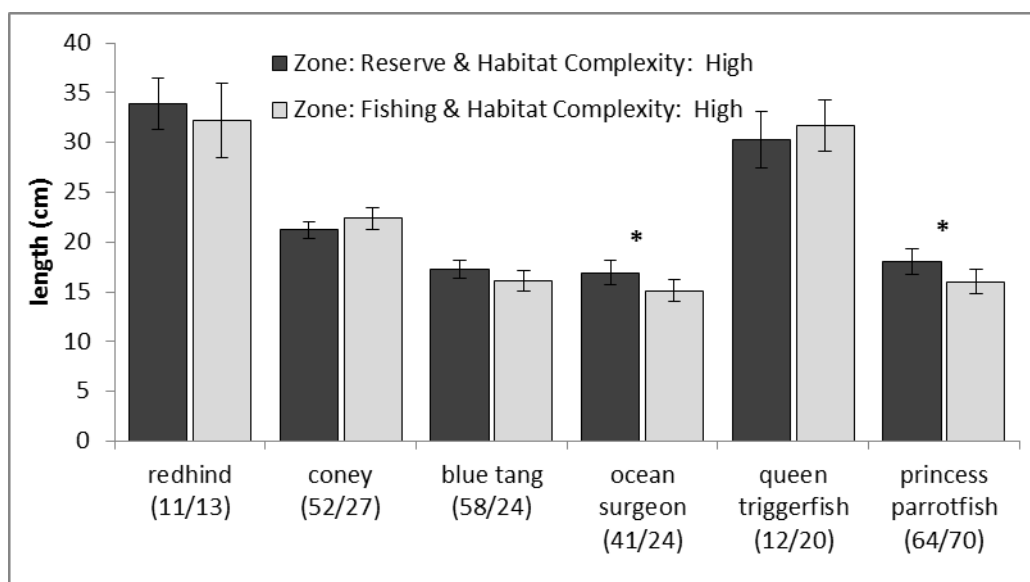


Figure 8. Comparison of mean lengths of principal fishery species from low (habitat category 1) and high complexity habitats (categories 2 and 3).

Table 4. PERMANOVA results for fish numerical abundance, species richness, overall mean length, overall mean fish biomass and total biomass (summed over all fishes), Shannon and Simpson diversity, and mean trophic level calculated both by abundance and biomass for the factors fishing, habitat and their interaction.

Variable	Fishing				Habitat				Fishing * Habitat			
	df	MS	Pseudo-F	P (perm)	df	MS	Pseudo-F	P (perm)	df	MS	Pseudo-F	P (perm)
abundance	1	1.56	6.45	0.001	1	5.11	21.08	0.001	1	0.42	1.73	0.077
number of species	1	0.92	19.36	0.001	1	3.40	71.90	0.001	1	0.02	0.40	0.650
mean length	1	1.53	7.00	0.001	1	4.65	21.31	0.001	1	0.29	1.31	0.199
mean biomass	1	1.28	4.47	0.001	1	3.67	12.78	0.001	1	0.36	1.24	0.235
total biomass	1	1.11	3.87	0.002	1	4.19	14.55	0.001	1	0.42	1.45	0.137
Shannon diversity	1	0.30	16.34	0.001	1	1.30	70.69	0.001	1	0.00	0.15	0.772
Simpson diversity	1	0.08	9.60	0.003	1	0.36	40.42	0.001	1	0.00	0.15	0.749
trophic level (abundance)	1	0.00	0.12	0.755	1	0.02	4.18	0.045	1	0.00	0.42	0.506
trophic level (biomass)	1	0.02	3.30	0.071	1	0.03	4.42	0.042	1	0.00	0.12	0.742

Table 5. ANOVA results for abundance and mean length of the families Acanthuridae, Scaridae and Serranidae for the factors fishing, habitat and their interaction.

Variable	Fishing				Habitat				Fishing * Habitat			
	df	MS	F	P	df	MS	F	P	df	MS	F	P
abundance												
Acanthuridae	1	39.23	0.77	0.384	3	364.66	7.11	0.000	2	2.21	0.04	0.958
Scaridae	1	105.03	5.92	0.017	3	326.82	18.43	0.000	2	31.32	1.77	0.177
Serranidae	1	1.33	0.26	0.611	3	115.22	22.50	0.000	2	0.57	0.11	0.895
mean length												
Acanthuridae	1	2807.99	2.04	0.155	3	3550.09	2.58	0.055	1	4914.20	3.57	0.060
Scaridae	1	10106.29	2.73	0.100	3	18253.67	4.93	0.002	1	10826.68	2.92	0.088
Serranidae	1	525.21	0.07	0.786	3	10482.10	1.48	0.221	1	192.01	0.03	0.869

3.5 Patterns in familial and species representation

Familial representation of fishes differed between habitats having different structural complexity, as shown in figure 9. In the lowest habitat category, the family with the highest relative abundance was the Congridae, accounting for 32% of the total number of fish. The second and third most numerous families were the Carangidae (18%) and Labridae (21%), while all other families accounted for less than 10% of the total fish abundance. Habitat category 1, the second-most barren habitat, showed a completely different picture; as the Malacanthidae, Sphyraenidae and Congridae families were no longer part of the seven most abundant families. Here the Labridae were the most abundant family, accounting for 24% of fish abundance. With 9%, the representation of Carangidae was half lower in habitat 1 compared to habitat 0, while the Acanthuridae were 75% and the Pomacentridae 66% more important in habitat 1 than in habitat 2. For habitat 2, the Pomacentridae and Scaridae were relatively more important than in habitat 1, while the Acanthuridae Labridae and Serranidae were essentially the same (Fig. 9). In habitat 2, the Balistidae (7%) and Chaetodontidae (2%) were notable components of the community (Fig. 9). In habitat 3, the Acanthuridae and Balistidae were relatively more abundant than in habitat 2, whereas the Chaetodontidae and Pomacentridae showed little difference. The Scaridae were 37% less abundant in habitat 3 than in habitat 2 (Fig. 9).

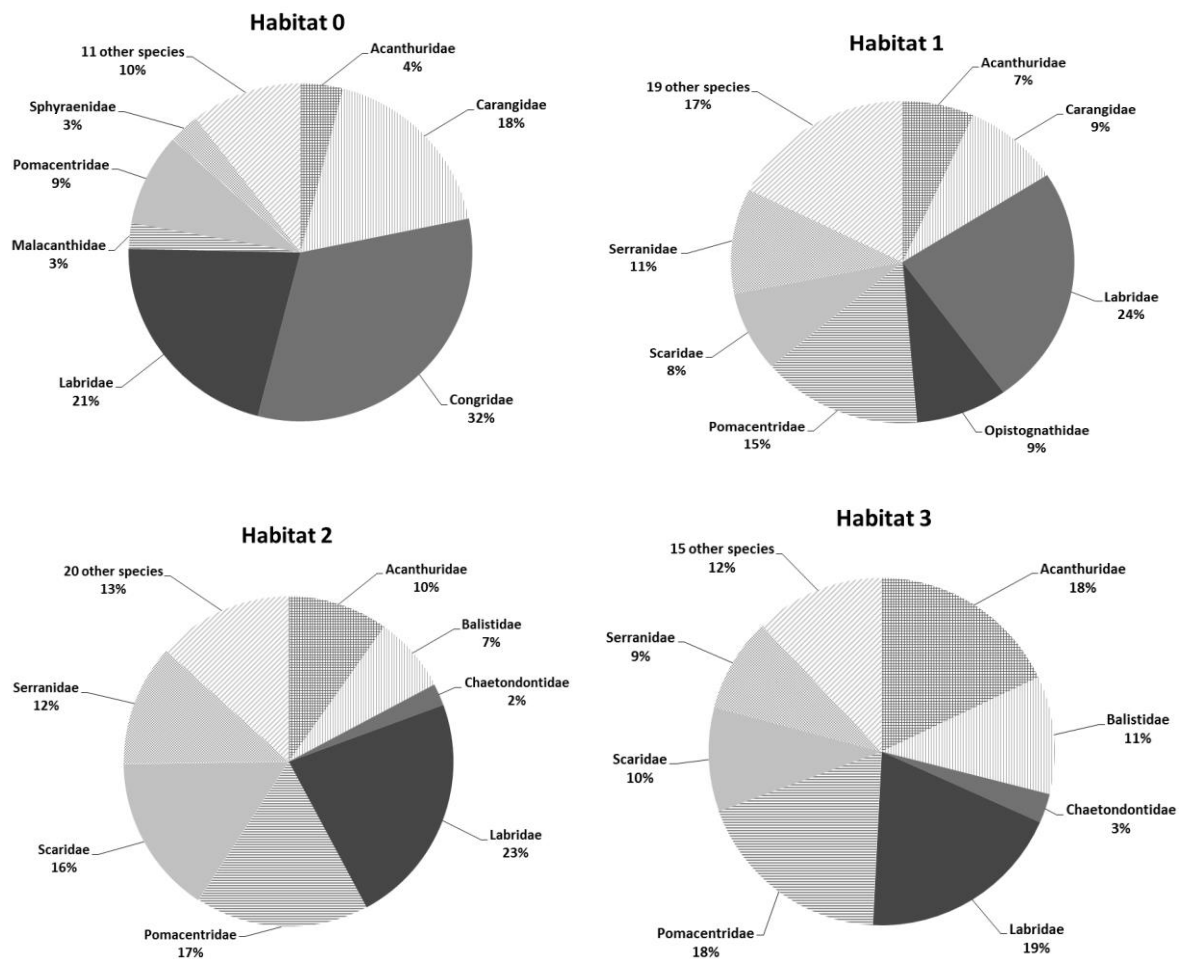


Figure 9. Pie chart showing the proportion of the number of fish for the seven most abundant families in percent for habitat categories 0, 1, 2 and 3. The effects of fishing reserve protection on fish species community structure were not significant and have not been separately plotted.

Many of these differences in familial representation were also statistically significant. For instance, Acanthuridae were significantly more abundant with higher habitat structure ($F=7.11$ and $P=0.000$) but no significant effect was found for reserve status (Fig. 10). For the Scaridae the effect of habitat on abundance was also significant ($F=18.43$ and $P=0.000$) as was the effect of reserve status ($F=5.92$ and $P=0.017$) (Fig. 10). In the case of Serranidae, abundance also correlated positively with habitat structure ($F=22.50$ and $P=0.000$) but no difference could be detected due to reserve status (Fig. 10).

The differences at familial level, translated also into some evident differences at species level. Species whose abundance appeared most notably affected by habitat were *Xyrichtys martinicensis*, *Halichoeres poeyi* and *Serranus baldwini*. These mainly occurred on sand or algae patches in the lowest level of habitat structure, while *Holocentrus adscensionis* and *Thalassoma bifasciatum* were mainly found in the reef habitats of the higher categories. Species associated with high habitat categories in the CAP plot were *Thalassoma bifasciatum* and *Acanthurus coeruleus*, while *Heteroconger longissimus* and *Caranx ruber* were predominantly found in habitats with low spatial structure. as reflecting their known habitat preferences (Humann and Deloach 2002).

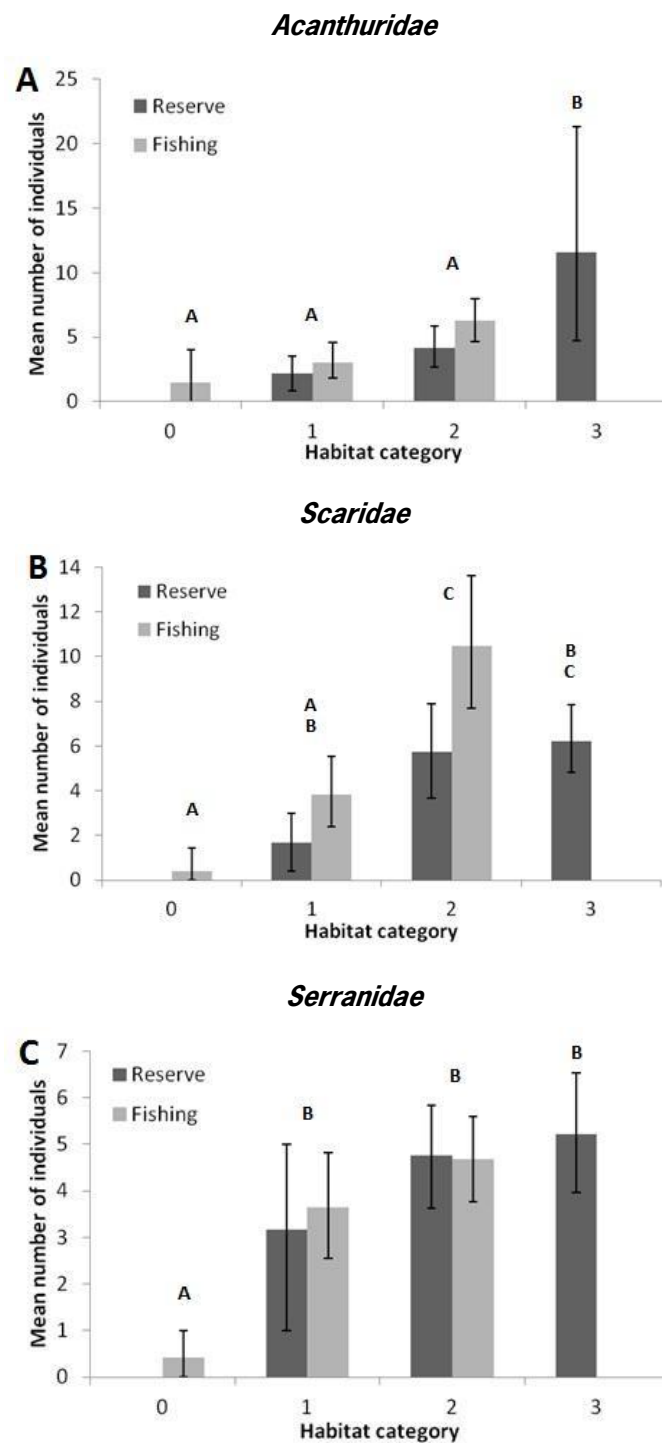


Figure 10. The mean numbers (± 1 SD) of a) *Acanthuridae*, b) *Scaridae* and c) *Serranidae* for all habitat categories in both the reserve and fishing zones. Error bars indicate 95% confidence intervals and different letters (A, B, C) above each bar represent statistically different habitat groups.

3.6 Community biomass distribution

Figure 11 shows the mean biomass per habitat category in the reserve zone. There were large and significant differences in mean biomass with increased habitat structure (pseudo- $F=12.78$ and $P=0.001$, Table 5), namely higher biomass with increased habitat complexity. Variations in biomass were large over all habitats and overall much larger than variations in numerical abundance. The effect of fishing zone on mean biomass was also statistically significant (pseudo- $F=4.47$ and $P=0.001$; Table 5) but was contradictory to fish numbers (Figure 6) and may represent a spurious result.

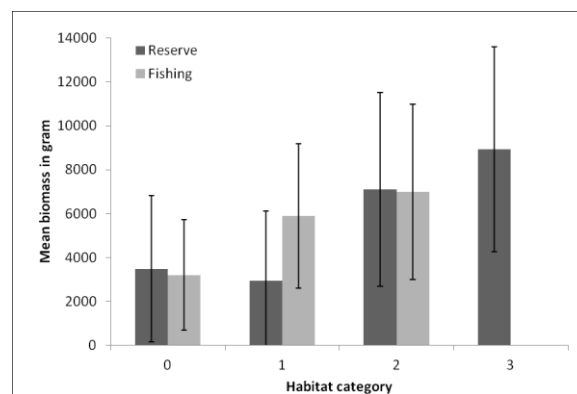


Figure 11. The a) mean biomass ($g \pm 1$ SD) and b) mean fish length ($mm \pm 1$ SD) per BRUV deployment for all habitat categories in both the reserve and fishing zones.

3.7 Higher-order community description

In this section we present three “higher-order” community descriptors. These are:

- a description of trophic guild structure,
- a nonmetric multi-dimensional (NMDS) scaling of community composition along environmental variables to visualize if the community is structured based on habitat or management zone, or both,
- community comparison in terms of widely-used the Shannon-Weaver diversity index H' .

3.7.1 Trophic guild structure

The largest proportion of the total biomass for the most barren habitat (0) is represented by the trophic group of the planktivores (58%; Figure 12). The rest of the assembly’s biomass was represented by invertebrate feeders and piscivores, while both herbivores and omnivores accounted for only a final few percent. For habitat category 1 the biomass distribution was totally different: planktivores only represented 17% of the biomass, while herbivore and invertivores represented 20% and 37% of biomass, respectively. Herbivores were also more abundant in category 2 compared to the low-structure habitats 0 and 1. All other trophic groups showed a lower biomass representation in higher-structure habitats, especially the planktivores. In habitat category 3, the herbivorous biomass amounted to 40%. Piscivores also reached their highest biomass density in this habitat 29%. Omnivores never accounted for more than 2% of the total biomass. The proportion of planktivorous biomass was lowest in habitat category 3 (4%).

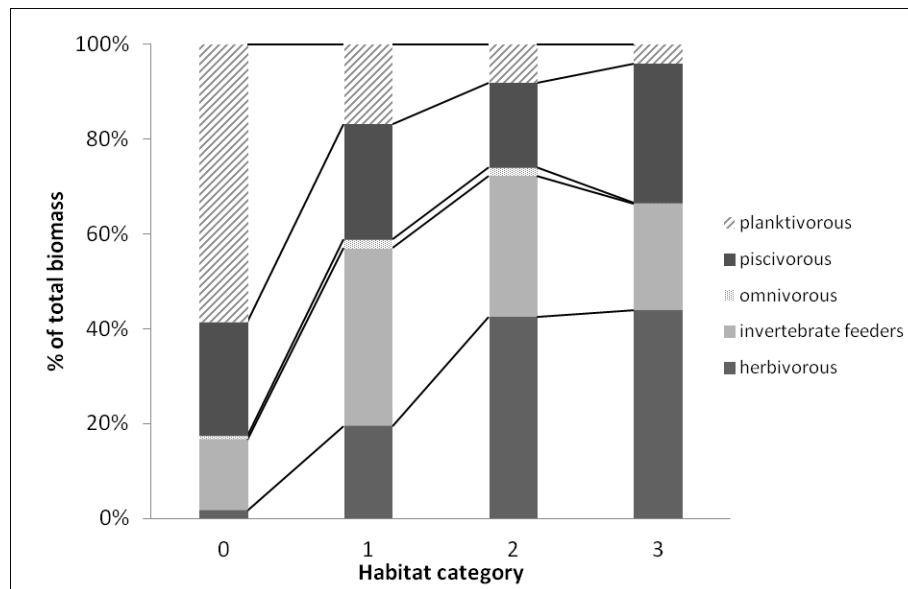


Figure 12. The proportion of the total biomass for the trophic groups of the planktivores, piscivores, omnivores, invertebrate feeders and herbivores in all habitat categories.

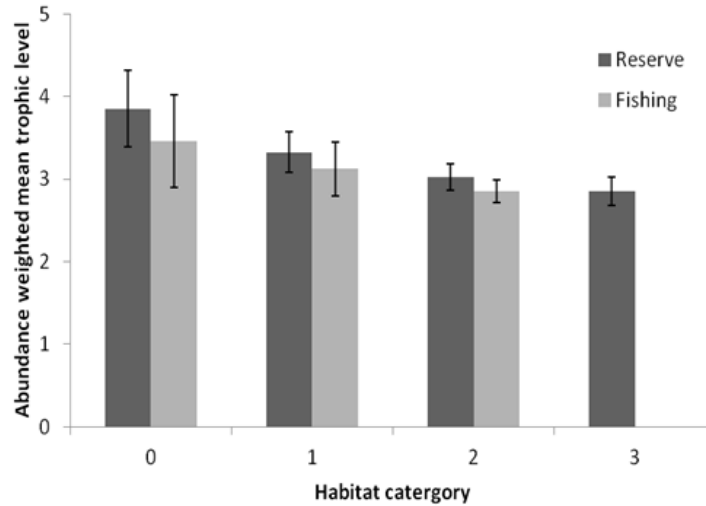


Figure. 13. Mean number-weighted trophic level (± 1 SD) for all habitat categories in both the reserve and fishing zones.

Numbers-weighted mean trophic levels decrease as habitat structure increases for both the fishing and reserve zones with a significant effect of habitat (pseudo- $F=4.42$, $P=0.042$) (Figure. 13). The effects of reserve status and the interaction between reserve status and habitat were not significant. The conclusion here is that habitat and not fishery reserve status affected community trophic structure.

3.7.2 nMDS

Variation in fish community structure between habitat categories was visualized using non-metric multidimensional scaling (nMDS) with abundance data (Figure 14). According to Clarke (1993) a stress value below 0.2 indicates that an nMDS plot is useful for interpreting differences between communities. The stress value for this nMDS plot is 0.17, meaning that our plot provides a useful representation of the variation in fish assemblage structure (Figure 14).

For our St. Eustatius samples, there is a clear separation between fish assemblage structure between the lower and higher habitat types in the nMDS plot, which can be seen by their locations on the two dimensional MDS plot (Figure 14a). Fish assemblages from habitat 0 are located mostly to the upper right of the plot and show the largest variation. Assemblages from habitats with a higher level of spatial structure are found in the upper left corner and show considerable overlap. The spread of fish communities for habitat categories 2, 3 and 4 is also much lower than for the low-structure habitats. Figure 14b shows the same nMDS plot with the same stress of 0.17, this time with a focus on fish assemblage variation between the fishing and reserve zone. Since the points for both zones display the same pattern, fish assemblages appear to have similar variation between the reserve and fishing zones. Fishing reserve status hence has no influence on overall species community structure.

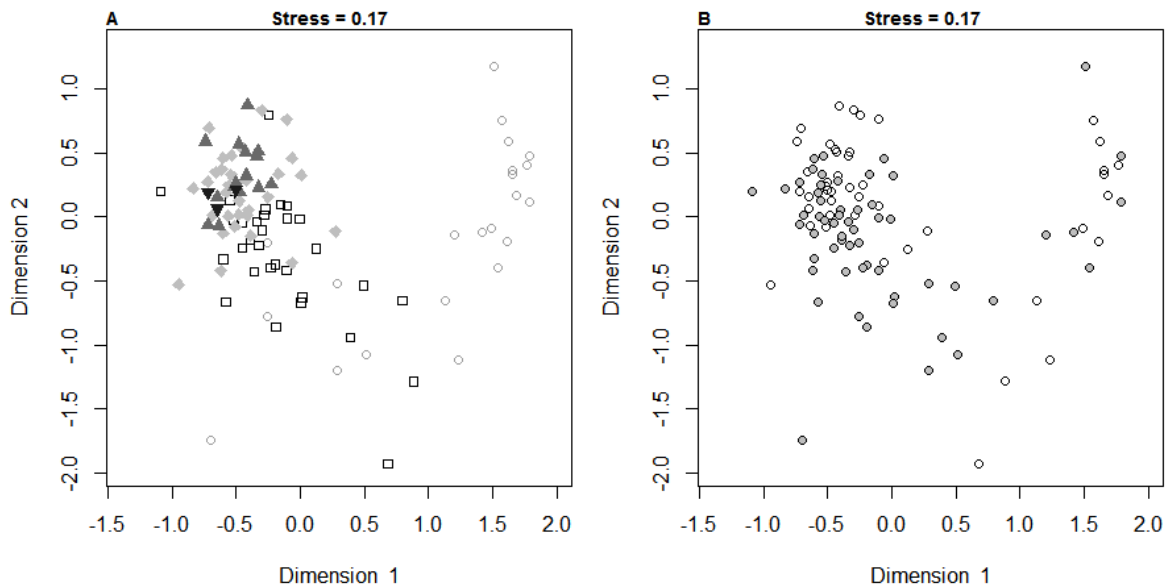


Figure 14. NMDS plot displaying variation in species richness and numerical abundance between drops. (A) shows differences in assemblage structure between the different habitat categories, with habitat category 0 = grey circle, 1 = black squares, 2 = light grey filled diamonds, 3 = dark grey filled triangle pointing up and 4 = black filled triangle pointing down. (B) shows differences in assemblage structure between the management zones, with reserve zone = black circles and fishing zone = grey filled circles.

3.7.3 Shannon-Weaver Diversity Index H'

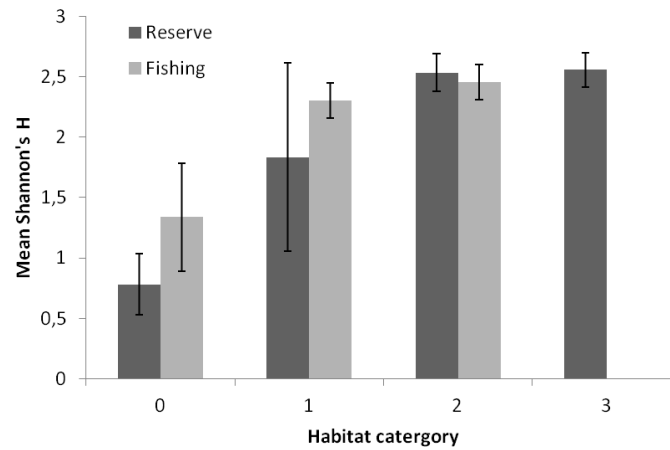


Figure. 15. Mean Shannon diversity H' (± 1 SD) for the different habitat categories in both reserve and fishing zones.

The Shannon-Weaver diversity index is a common and widely-used community descriptor. There was a large and significant increase in diversity with increasing habitat complexity, whereas fishery reserve status had no significant influence on community diversity (Fig. 15).

3.8 Reef shark species

A total of 42 sharks sightings (Blacktip Shark, *Carcharhinus limbatus*: N = 2; Caribbean Reef Shark, *Carcharhinus perezii*: N = 29; and Nurse Shark, *Ginglymostoma cirratum*: N = 11), were recorded from the 104 video-BRUV stations. Fig. 16 shows the distribution of detected sharks around the island. As can be seen, sharks appeared widely distributed around the island. Nevertheless, the two main species appeared to show some contrast in distribution. Most Caribbean reef sharks were found in the fishing reserves whereas most nurse sharks were found outside the reserves (Fig. 17). Due to the small number of sightings involved, more research is needed to assess whether this contrasting pattern is significant and what any underlying causes may be (eg. differences in habitats or fishing pressure).

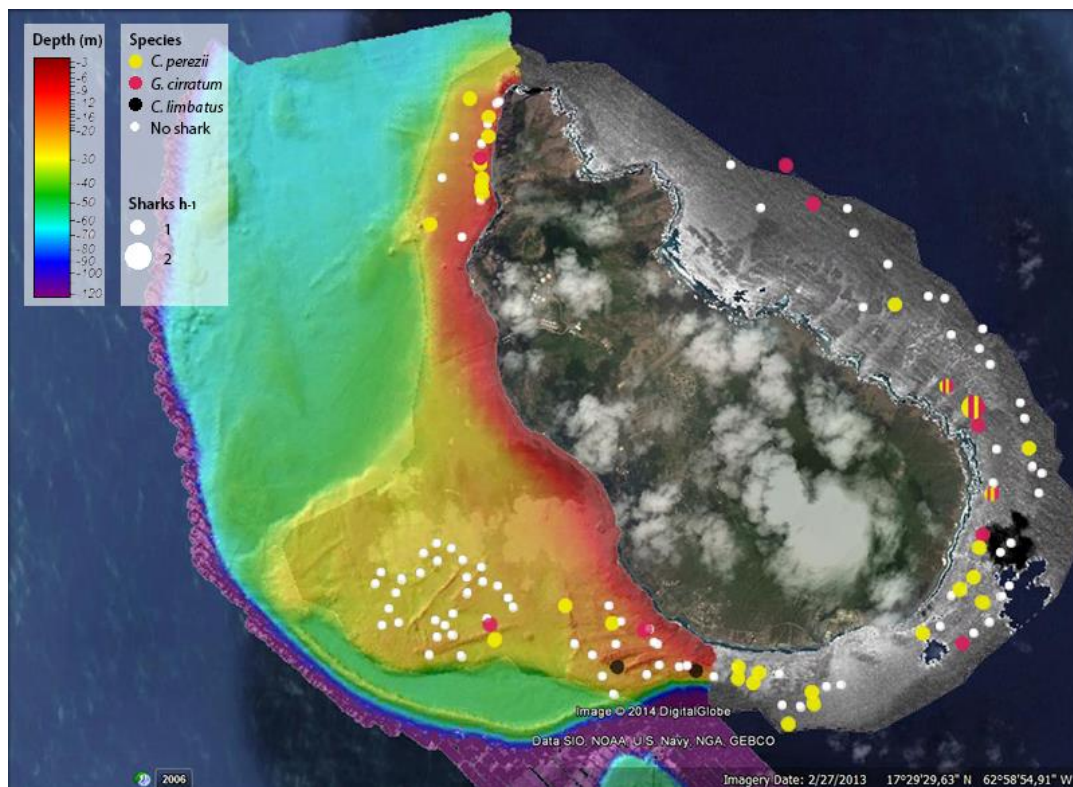


Figure 16. Overview of shark observation of the sBRUV survey.

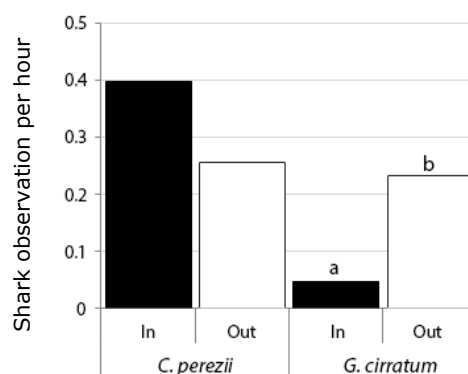


Figure 17. Density of reef shark inside and outside the Reserves.

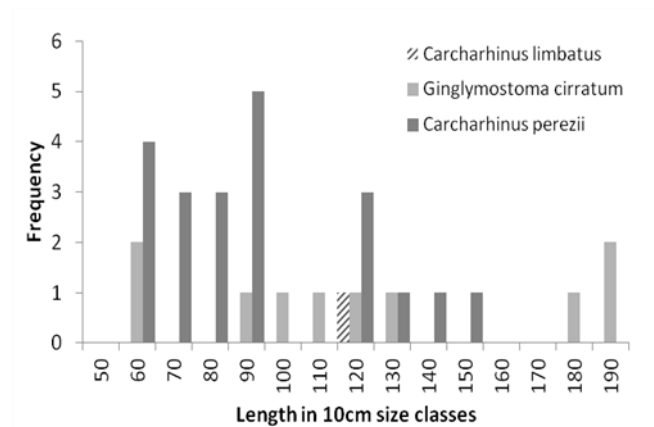


Figure 18. Pooled size-frequencies for all 42 sharks observed.

On a final note, the range of shark lengths found in this study was quite large (Figure. 18). The majority of Caribbean reef sharks were juvenile, as males and females only mature above 1.5 and 2 m respectively (Scharfer 2009). Most nurse sharks had a length between 50 and 130 cm and were also clearly juveniles. All of the largest three sharks of about 180 and 190 cm long were nurse sharks.

4. Discussion

4.1 Patterns in fish assemblage structure, diversity and trophic structure

We here describe the fish communities of shallow sublittoral marine habitats of St. Eustatius and investigate how these differ based on habitat and designated fishing zone. The various analyses show significant differences in the distribution of assemblages but indicate clearly the overriding effect of habitat differences on community structure as opposed to fishing reserve status.

From studies elsewhere (e.g. Watson et al. 2010), and from the outset of this study, it was to be expected that habitat differences would have large effects on fish community structure, so these results are not surprising. Such actual and expected effects of habitat did necessitate controlling for habitat differences when wishing to examine the influence of management measures such as in this case the effects of fishing reserve status. In the case of St. Eustatius possible difference in fishing pressure between fishing reserve fishing zones were minor and only led to a modest difference in fish abundance, size and species richness. No major effects could be detected in terms of community diversity, trophic structure or even size structure of the key fishery “target”-species.

As to be expected based on the lack of mangroves and reduced seagrass beds (Debrot et al. 2014), the species composition of the fish community largely or even fully lacked several species dependent on mangrove and seagrass nursery habitat. For instance, grey snappers and school master snappers which are strongly dependent on mangroves and seagrass habitats were rare or even fully absent as were large parrotfish species such as *S. coeruleus* and *S. guacamaia*, which are highly dependent on mangroves (Dorenbosch et al. 2004; Hylkema et al. 2014).

Our results show clearly that habitat structure plays a major role in diversity of fish assemblages around St Eustatius but that reserve protection did not. Low-structured habitat which offered little critical shelter to small fish had the highest mean fish size of all five habitats. Piscivores were significantly most abundant in higher-structured hard bottoms. Mean trophic levels were significantly affected by habitat but not by difference in marine zoning. These differences expressed themselves in that areas devoid of shelter had markedly lower fish abundance but significantly larger fish of higher trophic levels (predators). Smaller fish that are more vulnerable to predation and which are of lower trophic structure concentrate in higher structured areas with ample shelter and attached macrobenthos.

The use of fishing reserves as a conservation management tool presupposes 1) that measurable fishing effort actually occurs, 2) that it is selective (typically targeting higher trophic prey species), 3) that reserve zones are actually implemented and enforced, and 4) that the spatial segregation between reserve zone and fishing zone is sufficient to overcome the natural tendency of fishes to move and migrate between the designated zones. Research elsewhere (e.g. Roberts 1995) shows that fish abundance effects can occur on relatively small spatial scales due to the overriding tendency of reef fish species to be territorial or otherwise have limited home ranges. Therefore, condition 4 is likely to have been met in this study. As for condition 3, it would seem that in general most fishing by local fishermen takes place outside the boundaries of the marine reserves. In the last five years, marine park enforcement documented only about 15 cases of illegal fishing within the reserve boundaries and prosecuted only two cases of illegal fishing. In both cases the fish and the fishing gear were confiscated and both violators from St Kitts were fined (\$1000) as well (J. Berkel, pers. comm.). Several of the locals have also been caught in the last five years and in some cases their gear was confiscated. More often than not they were just warned by the police or Coast Guard (J. Berkel, pers. comm.). In other words, compliance and enforcement would seem to be sufficient.

Since predatory species with higher trophic levels are the main targets in fishery, an increased trophic level might have been expected in the marine reserves (Jennings and Polunin 1997, Newman et al. 2006; DeMartini et al. 2008). However, this point can be doubted in the case of St. Eustatius as there hardly exists a directed fishery for finfish (annual catch ~5000 kg; Poiesz, 2013). The majority of the small 11 outboard-driven local fishing vessels target lobsters and conch, and finfish landings largely amount to incidental catch resulting from the lobster trap fishery. In 2013, 180 of the 257 fishing trips targeted lobster while the remaining trips consisted of a mix of trolling, handlining and fish-trap fishing (Poiesz 2013). The species thus landed belong to a relatively unselective mix of trophic levels (Poiesz 2013). As a consequence, a large contrast in selective fishing pressure between reserve and fishing zones may not be present in the case of St. Eustatius, which could account for the small yet variously significant effects of fishing reserve designation on fish community parameters as elaborated below.

We suggest that the relatively low quantifiable effect of the present fishing reserves may have to do with either or a combination of a) low finfish fishing effort and/or low fishing selectivity, b) problems in the enforcement of the fishing reserves, or c) geographic scale issues due to the movement of fish between defined zones which act to blur potential effects of stated management regimes.

4.2 Patterns in fish abundance, size, and biomass distributions

In this project we quantify several community parameters typically used to document, compare and monitor developments in fish communities. Habitat complexity had the overriding effect on many community parameters whereas reserve status generally had small, conflicting and/or counter-intuitive comparisons between zones.

For instance, our study found a small but significantly higher numbers of fish in the fishing zone for habitat categories 0, 1 and 2 as compared to the reserve zone. The effect of reserve status was also significant for species richness, with the fishing zone showing a higher species richness for the first three habitat categories than the reserve zone. The results are contra-intuitive to many studies on marine protection, where researchers have found fish abundance and species richness to increase in marine reserves compared to fished areas (Bell 1983, Roberts and Polunin 1991, Rus 1996, Jennings and Polunin 1997, Wantiez et al. 1997, Babcock et al. 1999, Denny and Babcock 2003, Friedlander et al. 2003, García-Charton et al. 2004, Claudet et al. 2006; Ault et al. 2012).

This higher abundance (of smaller) fishes in the fishing zones, could potentially be the result of a selective removal of large predatory fishes due to the higher level of recreational fishing in this zone or caused by incomplete correction for the large habitat differences between the two management zones. Habitat structure was found to have a large and statistically significant effect on both abundance and species richness, with higher habitat categories supporting more individuals and species. The positive effect of increasing habitat structure on abundance and species richness coincides with expectations from studies elsewhere (Chapman and Kramer 1999; García-Charton et al. 2000, Friedlander et al. 2003, García-Charton et al. 2004, Claudet et al. 2006, La Mesa et al. 2010, Ortiz and Tissot 2012). However, for Statia nothing is quantitatively known about either fishing mortality or gear selectivity.

While abundance in number of fish and species richness differed (contra-intuitively) between the fishing and reserve zones, the way that the number of fish is divided over different families was very similar. Not only were six out of the seven most common families the same, their proportions of representation were also highly similar. The only different major families between the zones are the Congridae in the reserve zone and the Carangidae for the fishing zone.

The only species of Congridae observed, *Heteroconger longissimus*, is highly specialized to living in the sand patches near reefs (Humann and Deloach 2002) mainly found in the marine reserves around St Eustatius. This suggests that the shift from Carangidae in the fishing zone to Congridae in the reserve zone is mainly caused by habitat effects.

Fish community structure in terms of familial representation differed significantly between habitats with different structural complexity. Out of the three families compared, a small but statistically significant effect of reserve status on abundance was found only for the Scaridae, which had higher numbers of individuals in the fishing zones than the reserve zones. This was particularly the case for small fish and we speculate that this difference might be due to selective removal of piscivores in the fishing zone or due to uncorrected habitat effects. Several other studies have for instance found increased domination of herbivores associated with fishing pressure (Jennings and Polunin 1996, Friedlander and DeMartini 2002).

Both mean and total biomass was found to be affected by marine reserve protection status but opposite to expectation. The mean value for biomass of habitat category 1 was higher in the fishing zone compared to the reserve zone, but little difference was found for other categories. The overriding effect of habitat structural complexity on biomass was large and significant and matches the results as expected from other studies (Grigg 1994, Jennings et al. 1996, García-Charton et al. 2000, Friedlander et al. 2003, Newman et al. 2006). The distribution of biomass over trophic groups differs between the marine reserve and the fishing zone, with higher percentages of piscivores and planktivores in the reserve, whereas herbivores and invertebrate feeders represented a larger proportion of biomass in the fishing zone. Piscivores are primary targets of fisheries and often have higher densities in MPAs (Jennings and Polunin 1997, García-Charton et al. 2004, Newman et al. 2006, DeMartini et al. 2008).

4.3 Changes in fish communities over time

Notwithstanding quantitative incomparability of the surveys conducted in 1992, 2004, 2008 with the results of this study due to the introduction of a new sampling method (BRUV), some gross trends in the fish communities of St. Eustatius appear justified. For instance, comparing 2008 survey results with the earlier results of White et al. (2004) suggested declines in abundance of harlequin bass, lane snapper, sand tilefish, fairy basslets and trumpetfish in 2008, compared to 2004 (McLellan 2009). McLellan (2009) also suggested declines in abundance of larger snapper and grouper species such as cubera and dog snappers (*L. cyanopterus* and *L. jocu*) and, respectively, rock hind and yellowfin grouper (*Epinephelus adscensionis* and *Mycteroperca venenosa*). In 1992, Sybesma et al. (1993) documented Nassau groupers (*E. striatus*), tiger groupers (*M. tigrinus*) and yellow-mouth groupers at numerous dive sites. These species have been all but totally lacking from later studies. In this study only a single yellow-mouth grouper was documented but no Nassau grouper, rock hinds or tiger groupers were observed. Whereas in 1992 mahogany snappers (*L. mahogani*) and black margate (*Anisotremus surinamensis*) were common at many dive sites in subsequent studies these species have been consistently rare. We conclude that since initial surveys in 1992 (Sybesma et al. 1993), most of the larger piscivorous snappers and groupers appear to have consistently declined or even disappeared from the reefs of St. Eustatius. These large predatory species are also those most vulnerable to overfishing. However, several other fish species less-targeted by fishing activity also appear to have declined in terms of relative abundance suggesting that habitat degradation, which has also taken place (Debrot et al. 2014) are also likely playing an important role aside from potential fishing pressure.

4.4 Reef shark species

Because of its low fecundity in combination with high vulnerability to fishing and local extirpation, the Caribbean reef shark is listed as "Near Threatened" by IUCN (Rosa et al. 2006b). Caribbean Reef sharks were relatively common in the coastal waters of St. Eustatius. The reef shark was observed in 45% of all drops in the marine reserve but only in 30% of all drops in the fishing zone. Bond et al. (2012) reported observations of this species in 27% of BRUV drops in reserve zones and in 8% of drops in fishing zones in Belize, for a much larger contrast between fishing and non-fishing zones. However, the indication of higher abundance of Caribbean reef sharks in the Statia Marine Reserves was not statistically significant.

The mean number of Caribbean reef sharks observed per hour was 0.26 sharks per hour and considerably higher than values reported for the reef shark in the Bahamas ranging from 0.04 to 0.13 sharks per hour (Brooks et al. 2011). Taking into account that Caribbean reef sharks were seen in about twice as many BRUV drops around St Eustatius, our results appear to suggest that the reef shark is more abundant in the waters surrounding St Eustatius compared to a number of other areas in the Caribbean.

The nurse shark is at present the most common shark seen on Caribbean reefs (Ward-Paige et al., 2010) and the second most commonly observed in our study. It is listed by the IUCN as "Data Deficient" (Rosa et al. 2006a). Finally, the blacktip shark was seen only twice. Adults of the latter migratory species are uncommon in nearshore fisheries but are widely impacted by offshore longline fisheries (Tavares 2008). Due to its vulnerability to fishing and its dependence on vulnerable near shore nurseries, this species is listed as "Near Threatened" by IUCN (Burgess and Branstetter 2009).

Sharks are important top predators of healthy reefs (Robbins et al. 2006) and their presence around St. Eustatius might be an indication of a current overall low level of fishing pressure but reef sharks are not specifically targeted by the artisanal fishery in St Eustatius's coastal waters. Their relatively high presence around St. Eustatius is encouraging in the context of conservation, valuable for dive tourism and interesting for research. Based on their importance in the reef ecosystem, a shark protection plan has recently been drafted for the Dutch Caribbean (van Beek et al. 2014).

4.5 Evaluation of the BRUV technique

The BRUV approach can actually not be evaluated without also taking into account the sampling design. Due to the lack of good prior knowledge at the outset of this study we used a basically uniform, evenly-spaced sampling design for BRUV deployment. Our species accumulation curves and power analyses showed that while the net result was largely adequate for broad community description (which was our primary goal), the achieved test power was relatively low to ensure detecting even significant (25%) changes in species richness and numerical abundance for the various habitat categories. For sharper comparison between habitat and management zones a stratified sampling strategy is always preferred but was not used. Our design for BRUV sampling was also not really geared to highly specific or detailed questions regarding management and habitat effects. Our sampling design was meant to give a broad indication of fish communities and distribution, for which it was clearly adequate. In addition it provided significant additional insights.

Nevertheless, both for the purposes of detailed species specific questions as for long-term monitoring of ecological change, sampling strategies should be tailored to each question. Based on our results we can make the following broad recommendations: for periodic broad community description similarly extensive BRUV surveys as conducted here every couple of years should be totally adequate. However for more detailed habitat monitoring we advise a strategy of more frequent (annual?) surveys focussing on a smaller number of permanent monitoring sites of higher habitat complexity and diversity.

4.6 Main management implications

Coastal benthic fish communities are a key natural resource for many small islands that lack other major natural resources or have to deal with the many limitations and constraints that typically characterize Small Island Developing States (SIDS). Therefore, they play a critical role in ecosystem service provision for the island of St. Eustatius, where dive tourism, and small scale fisheries fulfil important role in local economic and socio-cultural well-being and are accorded important roles in future economic development scenarios for the island (ECORYS 2010, Island Government Statia 2010, DLG 2011). Fish populations also fulfil ecosystem-supporting services based principally on the key trophic functions of herbivory and predation. In this case, herbivory particularly regards the role of parrotfishes which we have described and discussed in some detail, while predation especially concerns top predators like the large groupers, snappers and sharks, all of which have also been discussed above.

The low abundance of large commercially interesting reef piscivores (especially groupers and snappers) underscores the need for effective reef fishery management. At the same time, our results suggest limited current management effectiveness of the present fishery reserves. **This clearly calls for additional work to help improve fishery reserve management to the mutual benefit of the ecological resilience, dive tourism, and fishery functions that fish populations serve.**

The reefs of St. Eustatius have been found to have low levels of three-dimensional structure (Debrot et al. 2014), which in this study was found to be the most important local determinant of fish abundance and distribution. This low-level of three-dimensional structure is due to coral reef mortalities which are the cumulative result of both local, regional and global phenomena (Jackson et al. 2014). Debrot and Sybesma (2000) listed overgrazing, soil erosion and coastal erosion as the three most problematic local factors impacting the marine environment of St. Eustatius. At the regional and global scale factor such as marine acidification, global warming, coral bleaching and epidemic coral disease play a critical role at the local level at St. Eustatius (Jackson et al. 2014). The lack of three-dimensional structure caused by coral mortalities and loss in recent decades (Debrot et al. 2014) means that measures to protect and enhance three-dimensional structure may be useful to help increase fish abundance to the benefit of both fishing and biodiversity stakeholders. Many studies have been done on the use of artificial reefs and structures with which to enhance shallow reef fish resources Bohnsack 1989, Bohnsack and Sutherland 1985, Quinn and Kojis 1995) and the most important shallow dive attraction in the St. Eustatius harbour is currently a pile of natural rocks from a collapsed historical structure (Fig. 19). In this regard, Meesters et al. (2015) have recently reviewed promising coral restoration techniques as applicable to the Caribbean Netherlands. **The potential for habitat enhancement to jointly help achieve fishery and conservation goals seems evident.**

5. Conclusions and recommendations

5.1 Conclusions

- A total of 107 species of finfish species were recorded in our survey. Fish communities of St. Eustatius lack many species dependent on mangroves and seagrass beds. (Mangroves have never occurred on St. Eustatius and seagrasses have suffered major degradation in recent decades).
- The most important local determinant of fish community structure was found to be habitat 3-dimensional structure while the measured effect of designated fishing reserve zones was much less pronounced. Community abundance of planktivores and herbivores were notably inversely related, with low-structured sandy habitat being dominated by planktivores and higher-structured hard-bottoms being dominated by herbivores.
- Compared to past surveys, our results show that the large, commercially most-interesting reef piscivores have largely disappeared from the reefs of St. Eustatius.
- Reef and nurse sharks sighting frequencies were relatively high compared to most of the Caribbean.
- Test power for detecting community change over time using the BRUV approach needs to be improved and fine-tuned based on the results of the current work.

5.2 Recommendations

- Monitoring future changes should combine similarly extensive BRUV surveys every 3-5 years with more frequent (annual) surveys focussing on a smaller number of permanent monitoring sites of higher habitat complexity and diversity (e.g. Global Coral Reef Monitoring Network protocol).
- Sharks are important top predators of healthy reefs and their relatively high presence around St. Eustatius is encouraging in the context of conservation, valuable for dive tourism and interesting for research. Considering their ecosystem and conservation importance (van Beek et al. 2014) sharks have been identified as a priority area in the Caribbean Netherlands Nature Plan (MinEZ 2013) and further studies of shark ecology are clearly needed.
- The marine reserves of the island are intended to fulfil key ecosystem services (in terms of resilience, dive tourism development and fishery production). However, our results suggest non-spectacular functioning and effectiveness of the current marine reserves. The causes thereof remain unclear and directed further research will be needed to enhance their functioning.
- Measures to protect and enhance the three-dimensional structure of the reefs of the island (e.g. artificial reefs) may be useful in helping to increase fish abundance to the benefit of fishing, tourism recreation, coastal protection and biological conservation. This calls for practical trials and pilot projects on new ways to enhance these vital ecosystem services.



Figure. 19. Corals and fish using manmade underwater structures from natural quarried rock as habitat and thereby enhancing the nearshore biodiversity.

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Quality Assurance

IMARES utilises an ISO 9001:2008 certified quality management system (certificate number: 124296-2012-AQ-NLD-RvA). This certificate is valid until 15 December 2015. The organisation has been certified since 27 February 2001. The certification was issued by DNV Certification B.V. Furthermore, the chemical laboratory of the Fish Division has NEN-EN-ISO/IEC 17025:2005 accreditation for test laboratories with number L097. This accreditation is valid until 27 March 2017 and was first issued on 27 March 1997. Accreditation was granted by the Council for Accreditation.

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Justification

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Approved: Dr. D.M.E. Slijkerman
Researcher

Signature:



Date: 9 July 2015

Approved: Drs. F.C. Groenendijk
Head of Maritime Department

Signature:



Date: 9 July 2015

Appendix 1: Sampling adequacy

species accumulation curves and power analysis

The rate at which new species were encountered during successive drops differed between habitat categories. While the lowest level of structure shows a steady accumulation of species with an increasing number of drop samples (Figure 6), the species accumulation curves of the other habitat categories rise quickly with the first few but levels off as the number of drops increase. It is also interesting to note that the variation observed around the species accumulation curves is very similar across all habitats, with standard deviations around 10 for all categories, while the total number of species detected within the same number of samples increased with each category. At 14 samples the number of species detected in category 0 is low with about 25 species, in category 1 it is about 50, where in categories 2 and 3 it is highest with about 60 species observed.

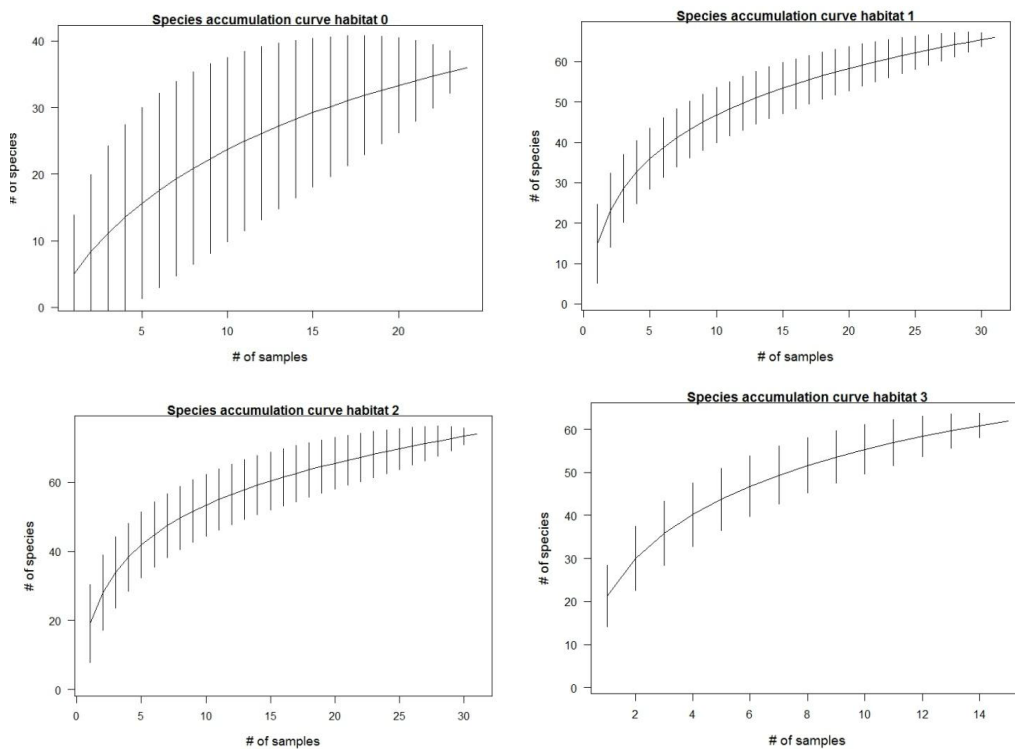


Figure 6. Species accumulation curve constructed for habitat categories 0, 1, 2 and 3, constructed from 24, 31, 31 and 15 drop samples, respectively. Habitat category 4 is excluded from this figure since only 3 drops were made in this type.

The numbers of drop samples necessary to detect changes of 25% in species richness and fish abundance for all habitat categories are shown in figure 7a. As can be seen, when aiming to detect a 25% change in species richness with a power of 0.8, a minimum of 45, 30, 15 and 20 drops are needed for habitat categories 1,2, 3 and 4, respectively, whereas even 95 drops are insufficient to characterize the fish community of habitat category 0.

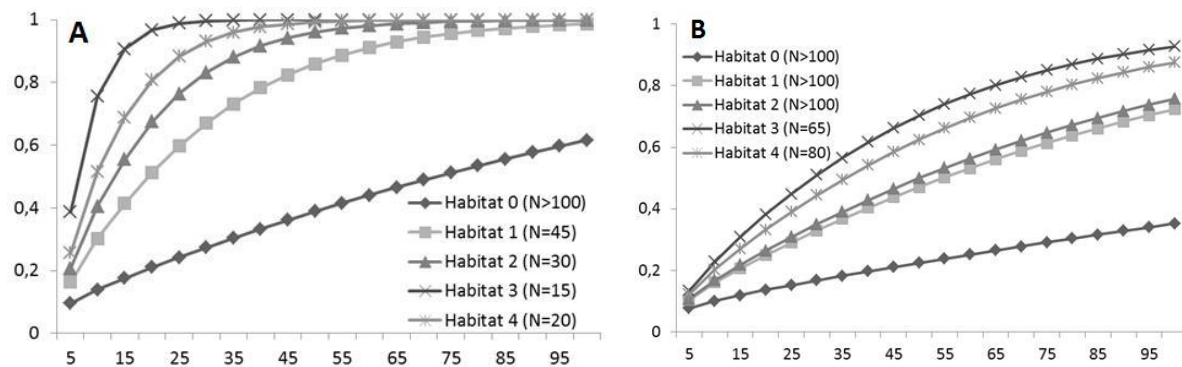


Figure 7. Power curves for the five habitat categories sampled, showing the power to detect a 25% change in a) species richness and b) fish numerical abundance. N is the number of samples required to reach a test power of 0.8.

To detect a change of 25% in total fish numerical abundance for each habitat category an even larger number of samples is required (Figure 7b). Even for categories 3 and 4 the number of drops needed exceeds 65. Clearly the sensitivity to detect changes in species richness is higher than to detect changes in abundance. Our results show that for high certainty in detecting changes of 25% and more, in terms of abundance using our chosen sampling design, much more sampling would be needed than we achieved in this study. However, our sampling design was meant to give a broad indication of fish communities and distribution. Clearly, for long-term monitoring of ecological change, a sampling strategy using repeated measures from a smaller number of selected high-diversity permanent monitoring sites is to be recommended.

Appendix 2: Relative numerical abundance and mean % presence per BRUV drop for all five habitat categories sampled and for all 107 fish species documented during BRUV video recordings.

		Habitat 0			N = 24			Habitat 1			N = 31			Habitat 2			N = 31			Habitat 3			N = 15			Habitat 4			N = 3		
Family	Genus species	Relative abundance	±	SD	Presence in % of drops	Relative abundance	±	SD	Presence in % of drops	Relative abundance	±	SD	Presence in % of drops	Relative abundance	±	SD	Presence in % of drops	Relative abundance	±	SD	Presence in % of drops	Relative abundance	±	SD	Presence in % of drops	Relative abundance	±	SD	Presence in % of drops		
Acanthuridae	<i>Acanthurus chirurgus</i>	0,00	±	-	0	0,03	±	-	3	0,23	±	0,05	10	0,33	±	0,14	13	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Acanthurus coeruleus</i>	0,04	±	-	4	0,71	±	0,03	42	2,06	±	0,05	90	7,93	±	1,02	100	3,00	±	0,67	100	3,00	±	0,67	100	3,00	±	0,67	100		
	<i>Acanthurus tractus</i>	0,71	±	0,23	13	2,13	±	0,12	74	3,16	±	0,08	90	2,73	±	0,18	93	6,00	±	1,76	100	6,00	±	1,76	100	6,00	±	1,76	100		
Aulostomidae	<i>Aulostomus maculatus</i>	0,00	±	-	0	0,16	±	0,00	16	0,39	±	0,01	32	0,20	±	0,00	20	0,67	±	0,00	67	0,67	±	0,00	67	0,67	±	0,00	67		
Balistidae	<i>Balistes vetula</i>	0,29	±	0,02	21	0,77	±	0,01	68	0,77	±	0,01	68	0,60	±	0,02	53	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Cantherhines macrocerus</i>	0,00	±	-	0	0,06	±	-	3	0,48	±	0,02	29	0,33	±	0,03	27	1,00	±	0,24	67	1,00	±	0,24	67	1,00	±	0,24	67		
	<i>Cantherhines pullus</i>	0,04	±	-	4	0,06	±	0,00	6	0,23	±	0,00	23	0,20	±	0,00	20	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Melichthys niger</i>	0,00	±	-	0	0,13	±	0,05	6	2,65	±	0,08	71	5,60	±	0,36	93	0,33	±	-	33	0,33	±	-	33	0,33	±	-	33		
Bothidae	<i>Bothus lunatus</i>	0,00	±	-	0	0,00	±	-	0	0,06	±	0,00	6	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
Carangidae	<i>Caranx latus</i>	0,00	±	-	0	0,19	±	-	3	0,00	±	-	0	0,07	±	-	7	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Caranx lugubris</i>	0,00	±	-	0	0,00	±	-	0	0,03	±	-	3	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Caranx ruber</i>	3,33	±	0,18	92	3,90	±	0,48	74	1,10	±	0,04	68	0,73	±	0,00	73	1,00	±	0,24	67	1,00	±	0,24	67	1,00	±	0,24	67		
	<i>Seriola rivoliana</i>	0,00	±	-	0	0,03	±	-	3	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
Carcharhinidae	<i>Carcharhinus limbatus</i>	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,13	±	0,00	13	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Carcharhinus perezii</i>	0,29	±	0,00	29	0,19	±	0,00	19	0,32	±	0,00	32	0,33	±	0,00	33	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Emblemaria pandionis</i>	0,00	±	-	0	0,10	±	0,02	6	0,03	±	-	3	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
Chaetodontidae	<i>Chaetodon aculeatus</i>	0,00	±	-	0	0,06	±	0,00	6	0,16	±	0,02	10	0,27	±	0,04	20	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Chaetodon capistratus</i>	0,08	±	-	4	0,26	±	0,03	13	0,52	±	0,02	39	1,00	±	0,05	60	2,00	±	0,33	100	2,00	±	0,33	100	2,00	±	0,33	100		
	<i>Chaetodon ocellatus</i>	0,00	±	-	0	0,00	±	-	0	0,03	±	-	3	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Chaetodon sedentarius</i>	0,04	±	-	4	0,16	±	0,02	10	0,00	±	-	0	0,13	±	-	7	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Chaetodon striatus</i>	0,00	±	-	0	0,29	±	0,02	23	0,48	±	0,02	35	0,27	±	0,00	13	0,33	±	-	33	0,33	±	-	33	0,33	±	-	33		
Clinidae	<i>Labrisomus nuchipinnis</i>	0,00	±	-	0	0,03	±	-	3	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
Congridae	<i>Heteroconger longissimus</i>	6,04	±	0,65	21	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
Dactylopteridae	<i>Dactylopterus volitans</i>	0,08	±	-	4	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
Dasyatidae	<i>Dasyatis americana</i>	0,13	±	0,00	13	0,26	±	0,00	26	0,03	±	-	3	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
Echeneidae	<i>Echeneis naucrates</i>	0,04	±	-	4	0,10	±	-	3	0,00	±	-	0	0,07	±	-	7	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Fistularia tabacaria</i>	0,00	±	-	0	0,06	±	0,00	6	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Gramma loreto</i>	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,07	±	-	7	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
Haemulidae	<i>Anisotremus surinamensis</i>	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,07	±	-	7	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Haemulon album</i>	0,00	±	-	0	0,06	±	-	3	0,03	±	-	3	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Haemulon aurolineatum</i>	0,00	±	-	0	0,03	±	-	3	0,13	±	-	3	0,13	±	0,00	13	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Haemulon carbonarium</i>	0,00	±	-	0	0,06	±	0,00	6	0,06	±	0,00	6	0,07	±	-	7	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Haemulon flavolineatum</i>	0,00	±	-	0	0,00	±	-	0	0,29	±	0,06	16	0,27	±	0,00	27	0,33	±	-	33	0,33	±	-	33	0,33	±	-	33		
Holocentridae	<i>Holocentrus adscensionis</i>	0,00	±	-	0	0,03	±	-	3	0,06	±	0,00	6	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Holocentrus rufus</i>	0,00	±	-	0	0,52	±	0,01	42	0,71	±	0,01	58	0,80	±	0,03	67	1,00	±	0,00	100	1,00	±	0,00	100	1,00	±	0,00	100		
	<i>Kyphosus sectatrix-incisor</i>	0,00	±	-	0	1,52	±	0,52	10	0,03	±	-	3	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
Labridae	<i>Bodianus rufus</i>	0,00	±	-	0	0,10	±	0,00	10	0,32	±	0,01	29	0,93	±	0,03	73	0,33	±	-	33	0,33	±	-	33	0,33	±	-	33		
	<i>Clepticus parrae</i>	0,00	±	-	0	0,00	±	-	0	1,03	±	0,25	10	0,33	±	0,05	13	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Halichoeres bivittatus</i>	0,13	±	0,00	13	0,19	±	0,01	16	0,00	±	-	0	0,07	±	-	7	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Halichoeres cyanocephalus</i>	0,13	±	0,00	13	0,65	±	0,02	42	0,32	±	0,01	26	0,13	±	-	7	0,33	±	-	33	0,33	±	-	33	0,33	±	-	33		
	<i>Halichoeres garnoti</i>	0,38	±	0,03	8	2,97	±	0,10	68	1,48	±	0,04	55	1,07	±	0,06	67	2,67	±	0,19	100	2,67	±	0,19	100	2,67	±	0,19	100		
	<i>Halichoeres maculipinna</i>	0,29	±	0,10	13	0,42	±	0,05	23	0,23	±	0,03	16	0,20	±	0,00	20	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Halichoeres poeyi</i>	0,17	±	0,00	8	0,32	±	0,04	13	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0	0,00	±	-	0		
	<i>Halichoeres radiatus</i>	0,00	±	-	0	0,03	±	-	3	0,16	±																				