diversity

MDPI

Article

Diversity and Community Structure of Coastal Elasmobranchs in the French West Indies

Océane Beaufort [1,*], Mark E. Bond [2], Michael R. Heithaus [2], Philip Matich [3] and Karl Questel [4]

[1] Kap Natirel, 97141 Vieux-Fort, France
[2] Institute of Environment, Department of Biological Sciences, Florida International University, Miami, FL 33199, USA
[3] Saving the Blue, Davie, FL 33328, USA
[4] Agence Territoriale de l'Environnement, 97133 Gustavia, Saint-Barthélemy, France
* Correspondence: oceane.beaufort@hotmail.fr

Abstract

Coastal elasmobranchs play key functional roles in Caribbean reef ecosystems, yet their populations have declined markedly across the region. To address major knowledge gaps in the French West Indies (FWI), we characterised recorded coastal elasmobranch assemblages in Guadeloupe, Martinique, Saint-Barthélemy and Saint-Martin using two independent observational datasets analysed separately: 1243 baited remote underwater video system (BRUVS) deployments conducted between 2016 and 2024 and 14,998 dives reported through the INA SCUBA citizen-science programme between 2019 and 2024. Sampling completeness, alpha and beta diversity, dominance patterns and species-specific encounter rates were assessed within each dataset. Across both datasets, 11 taxa were recorded: ten by BRUVS and five by INA SCUBA, with four shared between methods. In both datasets, Saint-Barthélemy had the highest mean richness and diversity with frequent detections of reef-associated sharks, whereas Saint-Martin showed an intermediate pattern. Guadeloupe and Martinique had lower sample-level richness and recorded assemblages dominated by a limited number of taxa, particularly the southern stingray, while reef-associated sharks were rare or not detected. BRUVS and INA SCUBA provided distinct but partly overlapping observational coverage; however, unequal sampling effort, site selection, temporal coverage, and species-specific detectability limit inter-island inference. These results provide a regional baseline for continued monitoring and for testing the ecological and anthropogenic factors that may underlie the observed spatial patterns.

Keywords: sharks; rays; alpha diversity; BRUVS; citizen science; beta diversity; marine protected areas; oceans

Check for updates

Academic Editor: Bert W. Hoeksema

Received: 22 August 2026
Revised: 12 October 2026
Accepted: 13 October 2026
Published: 15 October 2026

Copyright: © 2026 by the authors. Licensee MDPI, Basel, Switzerland. This article is an open access article distributed under the terms and conditions of the Creative Commons Attribution (CC BY) license.

1. Introduction

Elasmobranchs (sharks and rays) can play a critical role in the functioning of marine ecosystems, including tropical reef systems where they often occupy high or intermediate trophic positions [1–4]. As apex or mesopredators, they can couple food webs, transport nutrients, and affect prey behaviour, prey populations and community structure [1,5,6]. However, elasmobranchs generally exhibit life-history traits (including slow growth, late maturity and low fecundity) that make them particularly vulnerable to anthropogenic pressures, notably fishing-induced mortality [7].

Globally, marked declines in coastal and reef shark populations over recent decades have been documented [8,9]. In the Caribbean, this trend is especially concerning; sev-

eral studies report a substantial depletion of reef sharks relative to historical descriptions, suggesting processes of biotic simplification and trophic downgrading [10–12]. The consequences of these declines extend beyond the loss of individual species and may lead to functional erosion within communities, characterised by a reduction in the diversity of ecological roles within reef ecosystems [9,13].

The development of non-destructive monitoring methods has substantially improved the ability to document the presence, occurrence, and distribution of elasmobranchs in reef environments [14–16]. Among these approaches, baited remote underwater video systems (BRUVS) have emerged as a standardised and reproducible method. BRUVS are particularly well suited for assessing coastal elasmobranch communities and can be deployed at large spatial scales. BRUVS have proven effective for detecting rare species, comparing assemblages across sites, and evaluating the influence of environmental gradients or protection levels on reef-associated populations [14,17,18]. In addition to BRUVS surveys, citizen-science programmes conducted with dive clubs have also demonstrated their value in improving knowledge of the spatial and temporal distribution of elasmobranchs through the systematic reporting of sightings made during recreational dives [19,20]. Although each approach has inherent biases, considering observations from multiple approaches can broaden the observational coverage of elasmobranch assemblages [9,10,14,21], which is of great value for addressing knowledge gaps in data-poor regions.

Elasmobranchs have been studied across the Lesser Antilles, particularly within the French territories; however, information relies largely on opportunistic sightings, localised monitoring efforts, or fragmentary fisheries data, which limits the ability to conduct robust comparisons of communities across islands or habitat types. This lack of standardised data hinders the identification of spatial gradients, areas of concentration, or potential ecological specificities associated with individual territories or management regimes. Addressing this gap is crucial for guiding effective conservation strategies, especially in insular systems where anthropogenic pressures (fishing effort, tourism intensity, coastal development) and regulatory frameworks vary markedly from one island to another.

Here we aim to characterise the diversity and structure of coastal elasmobranch communities in the French West Indies (FWI; Guadeloupe, Martinique, Saint-Barthélemy and Saint-Martin) using two independent observational data sources: a standardised BRUVS monitoring programme implemented since 2016 and a citizen-science initiative based on scuba-diving observations (INA SCUBA, for Indice Nautique d'Abondance or "Nautical Abundance Index" for SCUBA diving) operational since 2019. Specifically, this study describes and compares α- and β-diversity patterns across territories within each dataset and analyses recorded assemblage structure and encounter rates for the most commonly recorded species. This comparative assessment at the scale of the FWI addresses a major regional knowledge gap and provides a baseline for long-term monitoring of elasmobranch communities in a Caribbean context.

2. Materials and Methods

2.1. Study Site

The study area encompasses the entire French West Indies (FWI), located in the eastern Caribbean Sea along the Lesser Antilles arc (Figure S1). The FWI comprises the following four territories: Guadeloupe, Martinique, Saint-Martin, and Saint-Barthélemy. Guadeloupe is the largest, with approximately 630 km of coastline, followed by Martinique (approximately 358 km), Saint-Martin (approximately 39 km), and Saint-Barthélemy (approximately 29 km). Guadeloupe and Martinique had populations of 384,160 and 360,630 inhabitants, respectively, in 2023, corresponding to population densities of approximately 236 and 320 inhabitants km^{-2}. The French territories of Saint-Martin and Saint-Barthélemy had

populations of 31,160 and 10,660 inhabitants, respectively, corresponding to substantially higher population densities of approximately 588 and 508 inhabitants km^{-2}, given their much smaller land areas [22,23]. Both Guadeloupe and Martinique host large barrier reef systems, including the 14 km-long Grand Cul-de-sac Marin in northern Guadeloupe and a 30 km-long reef along Martinique's Atlantic coast.

Due to the specific political status of each territory, marine conservation and fisheries are governed by distinct regulatory frameworks across the FWI. In addition to fisheries regulations, several marine protected areas (MPAs) have been established across the FWI in the form of national parks and nature reserves (Figure S2). In Guadeloupe, the national nature reserve (established in 1998) and the strictly protected "core areas" of the national park (designated in 2009) together cover 4152 hectares. This corresponds to 0.68 hectares of no-take area per kilometer of coastline. In Martinique, the entire Exclusive Economic Zone has been designated as a Marine Natural Park since 2017. This designation does not currently impose specific restrictions on fishing, but the island also hosts a 603-hectare marine nature reserve off its Caribbean coast where fishing is prohibited, representing a density ratio of 0.15 hectares of no-take areas per kilometer of coastline. In the French part of Saint-Martin, a 2900-hectare marine nature reserve, established in 1998, protects coastal habitats, corresponding to a density ratio of 72.5 hectares of no-take areas per kilometer of coastline. In Saint-Barthélemy, five coastal marine nature reserves, designated in 1996, cover a total of 1200 hectares in different parts of the island. Within these reserves, human activities are restricted, including a ban on shark and ray fishing, representing a density ratio of 30 hectares of no-take areas per kilometer of coastline.

2.2. Data Collection

Two independent observational datasets were examined: underwater observations collected through the INA SCUBA citizen science programme and observations obtained from standardised BRUVS surveys.

2.2.1. INA SCUBA

The protocol used for monitoring elasmobranchs was adapted from one initially developed in Guadeloupe for marine turtle monitoring [24]. Specific adaptations were introduced following expert recommendations on elasmobranch survey design and species identification to ensure consistency with best practices for coastal shark and ray monitoring [20]. From 2019 to 2024, two to five outreach sessions were held each year (on average three per year) to engage recreational SCUBA diving clubs across the FWI. A total of 51 dive clubs in the FWI were contacted and invited to join the programme; 38 subsequently participated. The dive instructors involved in data collection at each participating club received standardized training on the data collection protocol and on the identification of coastal elasmobranch species likely to be encountered in the region. Follow-up training sessions were also provided upon request to newly hired instructors at participating clubs. In addition, participants received an illustrated sharks and rays identification guide.

After each dive, trained instructors submitted sighting reports that included key information such as the date, location, time of day, dive duration, and, for each elasmobranch species sighted, the maximum number of individuals observed simultaneously during a single dive. This method follows the "best day's catch" approach [25], as recommended by Ward-Paige and Lotze (2011) [20], and helps minimise the risk of overcounting individuals. Overcounting can occur when divers encounter the same individual and/or group multiple times during a dive. Individual-level counts would require reliable identification of each animal, which necessitates complementary techniques such as photo-identification (e.g., refs. [26–28]), an approach beyond the scope of this citizen science program. When

possible, additional data such as estimated body size and sex were also recorded, although these variables were not included in the present analysis. Data were collated monthly either via printed forms or shared online spreadsheets. A trained reviewer checked all entries for quality and internal consistency. Reports containing unusual values, such as rare species, abnormally high counts, or records from new locations, were systematically flagged and followed up with observers to confirm the accuracy of the information. Only data from daytime dives were included in this analysis. The spatial distribution of dive sites was heterogeneous across all four territories (Figure S2). The temporal coverage of INA SCUBA observations differed among territories. Data were collected from 2019 to 2024 in Guadeloupe, from 2021 to 2023 in Martinique and Saint-Barthélemy, and from 2021 to 2024 in Saint-Martin. Both dry and wet seasons were regularly represented in Guadeloupe, whereas sampling effort was more concentrated during the dry season in Martinique, Saint-Martin, and Saint-Barthélemy (approximately 60–80% of sampling effort). Because the study aimed to provide an initial spatial characterisation of recorded elasmobranch assemblages rather than to assess temporal trends, observations were pooled across years and seasons. The term 'INA' hereafter refers to this participatory science programme.

2.2.2. BRUVS

The BRUVS sampling protocol followed the recommendations of Bond et al. (2018) [29]. Baited Remote Underwater Video Systems (BRUVS) consisted of mobile metal frames equipped with a bait cage positioned 1 m in front of the camera (GoPro Hero 3, 5, or 7; GoPro, Inc., San Mateo, CA, USA). Each BRUVS was baited with 1 kg of pilchards (*Sardinops* sp.). Deployment locations were recorded using a GPS, and each unit was deployed for at least 90 min at each station. The distance between consecutive BRUVS deployments was a minimum of 500 m to reduce the probability of sharks moving between sampling sites [30]. Video footage was subsequently analysed to identify species and quantify MaxN, defined as the maximum number of individuals of a given species visible simultaneously within the field of view [31]. Data extraction was conducted by two trained observers, who independently reviewed all videos at real-time speed or in accelerated mode (2× when appropriate). A 5 min acclimation period at the start of each recording was excluded from analyses to account for disturbance associated with camera setup. The spatial distribution of BRUVS deployments was relatively homogeneous in Saint-Martin and Saint-Barthélemy, but more heterogeneous in Guadeloupe and Martinique (Figure S2). The temporal coverage of BRUVS deployments differed among territories. Sampling was conducted in 2016, 2021, 2022, and 2024 in Guadeloupe; in 2016 in Martinique; in 2021 and 2022 in Saint-Martin; and during 2016–2018 and 2022–2024 in Saint-Barthélemy. Wet-season deployments accounted for approximately 60–80% of BRUVS sampling effort in Guadeloupe, Saint-Martin, and Saint-Barthélemy, whereas all BRUVS deployments in Martinique were conducted during the wet season. Because the study aimed to provide an initial spatial characterisation of recorded elasmobranch assemblages rather than to assess temporal trends, BRUVS observations were pooled across years and seasons.

2.3. Statistical Analysis

Statistical analyses examined inter-island variation in recorded elasmobranch assemblages by assessing sampling completeness, sample-level alpha diversity, assemblage composition, and species-specific encounter rates. Within each dataset, island was treated as the main grouping factor. Because BRUVS and INA differed in sampling design, spatial and temporal coverage, and observation processes, the datasets were analysed separately, using an individual deployment as the sampling unit for BRUVS and an individual dive for INA. The datasets were neither combined nor directly compared statistically. Cross-dataset

interpretation was therefore descriptive and focused on differences and overlap in species detections and on broad concordance in inter-island patterns. All analyses were conducted using R version 4.4.2 [32].

To evaluate whether sampling locations covered broadly comparable environmental conditions across territories, and to identify potential differences in habitat representation that could affect inter-island comparisons, separate principal component analyses (PCAs) were conducted for the BRUVS and INA datasets. PCA was performed using continuous environmental variables, including depth (derived from the General Bathymetric Chart of the Oceans GEBCO grid), distance to reef, distance to seagrass beds, and distance to the coastline. The distance variables were calculated from benthic habitat maps available for each territory [33–36]. Bottom rugosity was calculated from a regional bathymetric model at 100 m resolution to quantify seafloor heterogeneity [37]. Variables were centred and scaled to ensure equal weighting in the analysis. The categorical variable ISLAND was included as a supplementary variable in the PCA to visualise site distribution without influencing axis construction. For graphical interpretation, 95% confidence ellipses were added around the centroid of each territory within the space defined by the first two axes.

Sampling completeness with respect to detected species richness was assessed using sample-based rarefaction and extrapolation curves. This analysis examined the relationship between sampling effort and the number of elasmobranch species detected and evaluated whether sampling effort was sufficient to characterise species richness at the territorial scale. Rarefaction curves were used to compare detected species richness at standardised sampling effort, whereas extrapolation curves estimated expected richness under increased sampling effort. This approach reduced the influence of unequal sample sizes among territories and helped assess whether additional sampling under the same survey design was likely to detect additional species. Importantly, these curves evaluate sampling completeness only with respect to detected species richness. They do not assess the precision or representativeness of species-specific relative frequencies, nor do they account for differences in detectability among species, habitats, or survey methods.

Elasmobranch community diversity was described using multiple alpha diversity indices, including species richness, the Shannon index (H'), and the Simpson index ($1 - D$), calculated according to their standard formulations, where $H' = -\sum p_i \ln(p_i)$ and $D = \sum {p_i}^2$, with p_i representing the relative contribution of species 'i' to the total number of individuals recorded within a sample. The Shannon and Simpson indices were calculated only for sampling events in which at least one elasmobranch was detected, because relative species abundances cannot be defined for samples with no detections. Consequently, the resulting values describe assemblage diversity conditional on elasmobranch detection. Because alpha diversity indices did not follow a normal distribution, non-parametric tests were used to assess significant differences among territories. Variation in the diversity indices was examined using Kruskal–Wallis tests. When significant overall differences were detected, pairwise post hoc comparisons were performed using Dunn's test to identify territories that differed significantly from one another. The internal structure of assemblages and the relative importance of dominant taxa were described using dominance curves. For each assemblage, taxa were ranked in decreasing order of observation frequency, then normalised so that their sum equalled 1. These normalised values represent the relative frequency of species among recorded observations within each survey method. They should not be interpreted as population-level species proportions or estimates of true relative abundance, because species-specific detectability was not explicitly modelled. Cumulative frequency was calculated for each taxonomic rank, allowing visualisation of the progressive contribution of the most frequent taxa to the total assemblage and assessment of dominance levels within each community.

Elasmobranch community structure was further examined using beta diversity analyses to characterize variation in species composition among territories. Beta diversity was quantified using the Jaccard dissimilarity index. Dissimilarity matrices were calculated after aggregating observations at the territorial scale, considering a species present if it had been observed at least once within a given territory. Spatial structure of communities was visualized using heatmaps representing Jaccard dissimilarity matrices among territories. These visualizations illustrated levels of assemblage differentiation at the regional scale. Hierarchical dendrograms based on the same dissimilarity matrices were also produced to illustrate similarity relationships among territories (Figure S3). Multivariate ordination methods such as non-metric multidimensional scaling (NMDS) were considered but not retained due to the high proportion of zeros and low species-richness observations in the dataset, which could result in unstable and difficult-to-interpret ordinations. As an exploratory analysis, Bray–Curtis dissimilarity, incorporating species abundance, was calculated to assess whether abundance information provided additional structuring of beta diversity. However, Spearman correlation tests confirmed that Jaccard and Bray–Curtis dissimilarities were highly correlated (Spearman's $\rho = 0.98$ for BRUVS and $\rho = 0.97$ for INA), suggesting that abundance information did not substantially alter patterns of beta diversity. This strong correlation likely reflects the high proportion of low-abundance and low-richness observations.

Encounter rates were used to describe the spatial distribution of elasmobranchs based on BRUVS and INA datasets. For each species and territory, the encounter rate was defined as the proportion of sampling events in which at least one individual of the focal species was detected, irrespective of the number of individuals observed. Binomial 95% confidence intervals were computed using the Wilson method [38,39] to quantify uncertainty around each encounter rate. This approach is appropriate given the structure of the data, characterized by a very high proportion of zero detections (92.6% for BRUVS and 96.7% for INA) and the predominance of single-individual observations among positive detections (87.8% for BRUVs and 73.7% for INA). To facilitate interpretation, encounter rates were classified into five categories following Beaufort and Prieto-Gonzalez (2025) [40]: "absent" when the species was never recorded, "rare" for encounter rates <5%, "occasional" for encounter rates between 5% and 25%, "common" for encounter rates between 25% and 50%, and "very common" for encounter rates >50%. This categorical scale provides an intuitive description of species occurrence patterns across islands.

3. Results

3.1. Sampling Effort and Data Completeness

3.1.1. Sampling Effort and Environmental Coverage

The datasets comprised 14,998 dives and 1243 BRUVS deployments, with an unequal spatial distribution of effort (Table 1, Figure S2). Depth coverage varied among territories and datasets. INA sampling locations ranged from 13 to 40 m, with territory-specific median depths ranging from 16 to 26 m, whereas BRUVS deployments ranged from 7 to 35 m, with median depths ranging from 9 to 20 m (Table S1).

The first two axes of the BRUVS PCA (PC1 = 34%, PC2 = 26.9%) explained 60.9% of the total variance (Figure 1). The first axis primarily represents a horizontal spatial gradient driven mainly by distance from the coast and reef habitats, with an additional depth component. The second axis was structured by seabed rugosity (roughness). The distribution of sampling sites showed substantial overlap of the 95% confidence ellipses among the four territories, indicating broad overlap in the environmental conditions represented by the sampled sites.

Table 1. Sampling Effort and Island Characteristics by Territory.

Island	Sampling Effort		Coastline (km^2)	Human Population Density (km^{-2})	No-Take Area Density (ha/km Coastline)
	BRUV	INA			
GUADELOUPE	491	11333	630	236	0.68
MARTINIQUE	132	2693	358	320	0.15
SAINT-BARTHELEMY	525	301	29	588	30.0
SAINT-MARTIN	95	671	39	508	72.5
TOTAL	1243	14998			

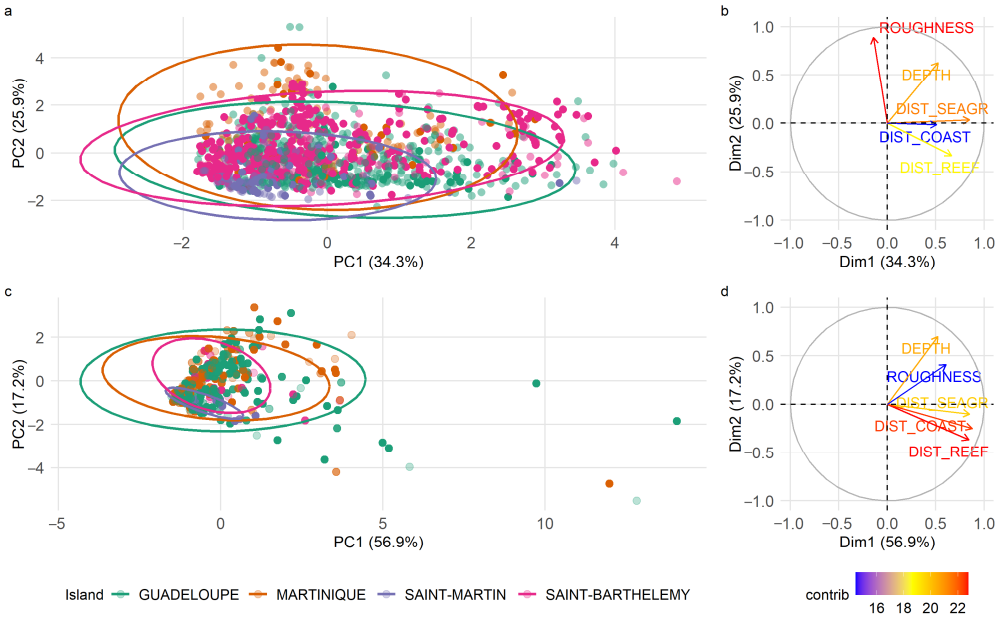


Figure 1. Principal component analyses of environmental characteristics at BRUVS (**a**) and INA (**c**) sampling sites, with contributions of environmental variables for BRUVS (**b**) and INA (**d**).

For INA data, the first two axes (PC1 = 56.9%, PC2 = 17.2%) explained 74.1% of the total variance (Figure 1). The first axis was mainly driven by distance to coast and reef structures, while the second axis was structured by depth and roughness (Figure 1). As observed for BRUVS, the confidence ellipses of the territories largely overlapped, and centroids remained close to one another, indicating comparable environmental conditions among islands. Guadeloupe showed a slightly wider dispersion, driven by a few extreme values, whereas Saint-Martin displayed a more compact cluster, indicating a narrower range of conditions among the sampled sites.

3.1.2. Rarefaction and Extrapolation Curves

Rarefaction and extrapolation curves of species richness derived from BRUVS data indicated a near-saturation pattern in most territories except for Saint-Martin (Figure 2). In Guadeloupe, species richness increased rapidly before approaching an asymptote. At the observed sampling effort (approximately 500 deployments), the slope remained only slightly positive, and extrapolation to 1000 deployments suggested a gain of approximately one to two additional species. In Martinique, the curve reached a plateau rapidly despite the relatively low sampling effort (150 deployments). In Saint-Barthélemy, the curve displayed a marked plateau, and no additional species were predicted to be detected from 500 to 1000 deployments, indicating that the estimated asymptotic richness under this survey design was likely approached. In contrast, in Saint-Martin the curve did not show a clear plateau at 100 deployments. Extrapolation to 180 deployments suggested a potential gain

of one to three additional species. Detectable richness therefore does not appear to have been fully reached in this territory.

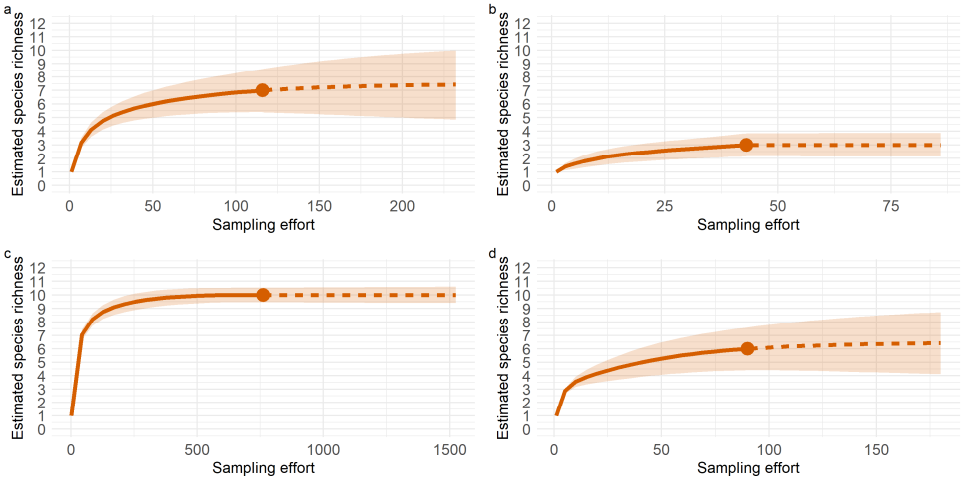


Figure 2. Rarefaction and extrapolation curves of BRUVS species richness across territories: (**a**) Guadeloupe, (**b**) Martinique, (**c**) Saint-Barthélemy, (**d**) Saint-Martin.

Rarefaction and extrapolation curves of species richness based on INA data showed rapid saturation of detected richness across all territories (Figure 3). In Martinique, species richness increased sharply at low sampling effort before quickly reaching a plateau. At the observed effort (approximately 3000 dives), the curve had stabilised, and extrapolation to 5000 dives indicated no additional gain in species richness. In Guadeloupe, a similar pattern was observed: richness increased rapidly before stabilizing at around five species. At the observed effort (approximately 11,000 dives), the curve had already reached a plateau, and extrapolation to 22,000 dives predicted no additional species. In Saint-Barthélemy, saturation was reached very early, at around four species. At the observed effort (approximately 300 dives), extrapolation to 600 dives showed no increase in estimated richness. In Saint-Martin, the curve also exhibited rapid stabilisation at approximately four species. At the observed effort (approximately 600 dives), extrapolation to 1200 dives did not suggest any further increase in species richness.

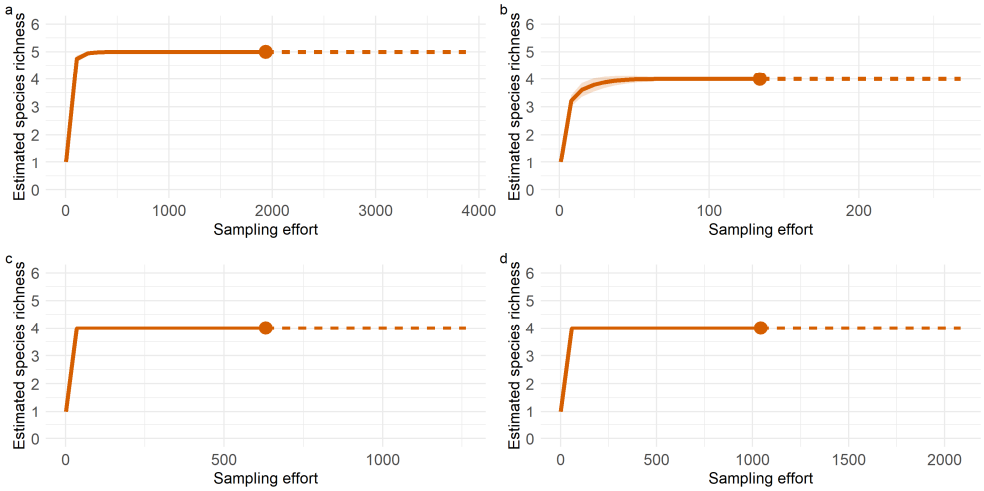


Figure 3. Rarefaction and extrapolation curves of INA species richness across territories: (**a**) Guadeloupe, (**b**) Martinique, (**c**) Saint-Barthélemy, (**d**) Saint-Martin.

3.1.3. Species Coverage Across Observational Approaches

BRUVS and INA provided distinct but partially overlapping species coverage (Figure 4). BRUVS documented a wider range of mobile shark species, whereas INA contributed observations of benthic taxa that were not recorded by BRUVS. BRUVS recorded more elasmobranch taxa, including the following sharks not detected by INA: blacktip shark (*Carcharhinus limbatus*), blacknose shark (*Carcharhinus acronotus*), sharpnose shark (*Rhizoprionodon* spp.), tiger shark (*Galeocerdo cuvier*), great hammerhead (*Sphyrna mokarran*), and lemon shark (*Negaprion brevirostris*). INA primarily detected benthic species and contributed one species not recorded by BRUVS, the lesser electric ray (*Narcine bancroftii*). Four species were detected by both approaches: the spotted eagle ray (*Aetobatus narinari*), southern stingray (*Hypanus americanus*), Caribbean reef shark (*Carcharhinus perezi*), and nurse shark (*Ginglymostoma cirratum*).

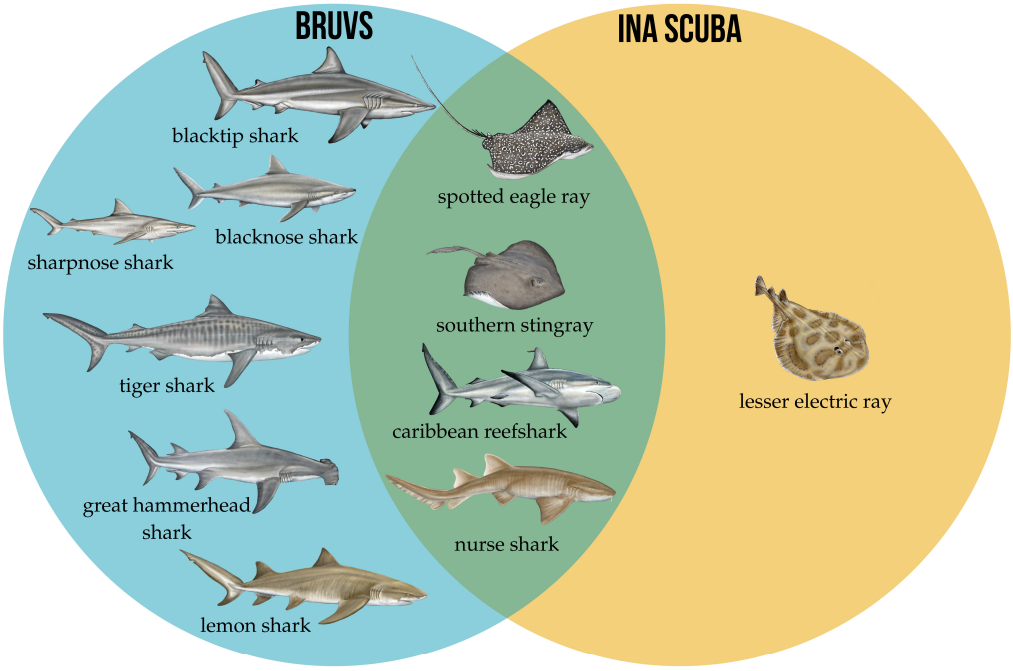


Figure 4. Overlap in elasmobranch species detected by BRUVS and INA.

3.2. Alpha Diversity

3.2.1. Species Richness

Sample-level species richness differed among islands in the BRUVS dataset (Kruskal–Wallis test, H = 368.014, df = 3, $p < 0.001$; Figure 5). Total richness was highest in Saint-Barthélemy (ten species) and lowest in Martinique (three species). Mean sample-level richness was highest in Saint-Barthélemy (1.26), followed by Saint-Martin (0.60), Martinique (0.29), and Guadeloupe (0.22) (Figure 5). Dunn's post hoc comparisons indicated no significant difference between Guadeloupe and Martinique, whereas Saint-Barthélemy and Saint-Martin each formed distinct groups.

Sample-level species richness derived from INA data also differed among islands (Kruskal–Wallis test, H = 2939.379, df = 3, $p < 0.001$; Figure 5). Mean species richness was highest in Saint-Barthélemy (1.38), followed by Saint-Martin (0.86), Guadeloupe (0.12), and Martinique (0.04) (Figure 5). Dunn's post hoc comparisons indicated significant differences among all four islands.

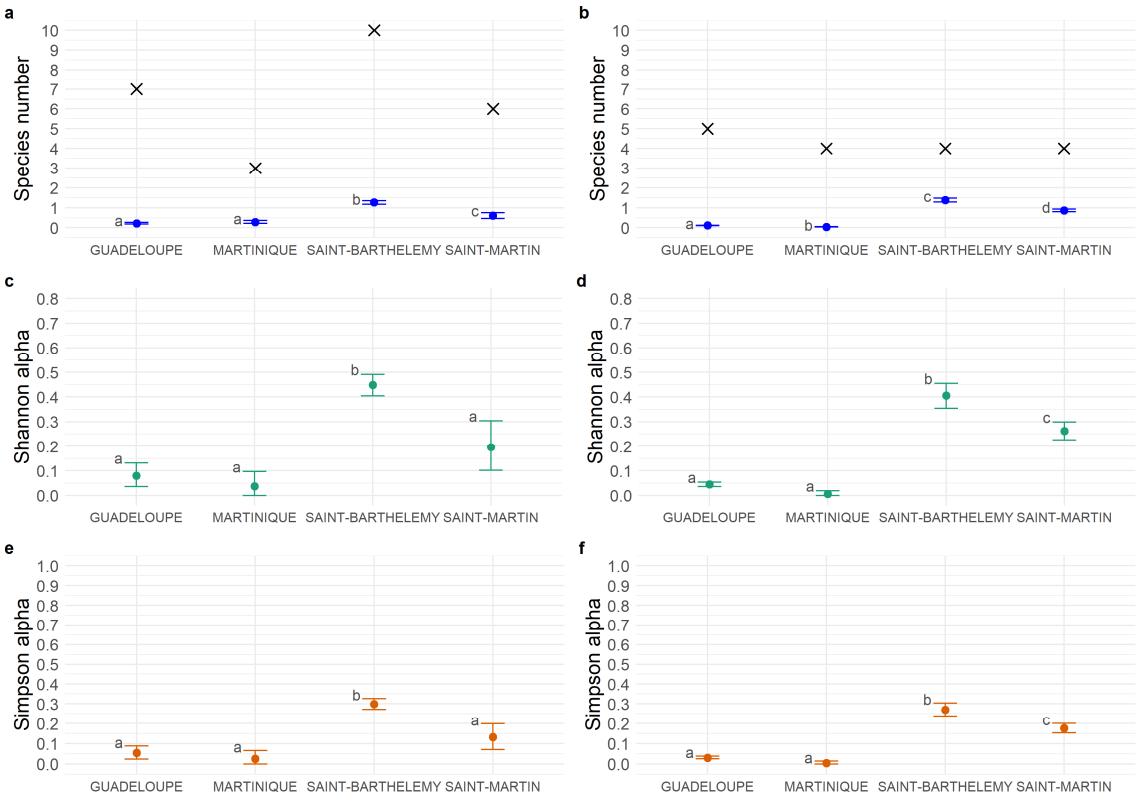


Figure 5. Alpha diversity metrics across islands for BRUVS data (**left** panels) and INA data (**right** panels). (**a**,**b**) Species richness per island, where points represent mean richness per sample, error bars show 95% bootstrap confidence intervals, and crosses indicate total species richness recorded at the island scale. (**c**,**d**) Shannon diversity index (H′), where points represent mean Shannon diversity per island and error bars show 95% bootstrap confidence intervals. (**e**,**f**) Simpson diversity index, where points represent mean Simpson diversity per island and error bars show 95% bootstrap confidence intervals. For all panels, letters next to points indicate significant pairwise differences among islands based on Kruskal–Wallis tests followed by Dunn's post hoc comparisons ($p < 0.05$).

3.2.2. Shannon Diversity Index

Shannon diversity indices estimated from BRUVS data were generally low but differed overall among islands (Kruskal–Wallis test, H = 90.669, df = 3, $p < 0.001$; Figure 5). Mean Shannon diversity was highest in Saint-Barthélemy (H′ = 0.45), followed by Saint-Martin (H′ = 0.20), Guadeloupe (H′ = 0.08), and Martinique (H′ = 0.04). Dunn's post hoc comparisons showed that Saint-Barthélemy had significantly higher diversity than the other three islands.

Shannon diversity indices obtained from INA data followed a similar spatial pattern (Kruskal–Wallis test, H = 436.521, df = 3, $p < 0.001$; Figure 5). Mean Shannon diversity was highest in Saint-Barthélemy (H′ = 0.41), followed by Saint-Martin (H′ = 0.26), Guadeloupe (H′ = 0.04), and Martinique (H′ = 0.01). Dunn's post hoc comparisons showed no significant difference between Guadeloupe and Martinique, whereas Saint-Barthélemy and Saint-Martin each formed distinct groups.

3.2.3. Simpson Diversity Index

Simpson diversity indices calculated from BRUVS data differed overall among islands (Kruskal–Wallis test, H = 90.595, df = 3, $p < 0.001$; Figure 5). Mean Simpson diversity was highest in Saint-Barthélemy (1 − D = 0.29), followed by Saint-Martin (1 − D = 0.13), Guadeloupe (1 − D = 0.05), and Martinique (1 − D = 0.03). Dunn's post hoc comparisons

showed that Saint-Barthélemy differed significantly from the other three islands, whereas Guadeloupe, Martinique, and Saint-Martin did not differ significantly from one another.

Simpson diversity indices calculated from INA data also differed among islands (Kruskal–Wallis test, H = 436.403, df = 3, $p < 0.001$; Figure 5). Mean Simpson diversity was highest in Saint-Barthélemy (1 − D = 0.27), followed by Saint-Martin (1 − D = 0.18), Guadeloupe (1 − D = 0.03), and Martinique (1 − D = 0.004). Dunn's post hoc comparisons showed no significant difference between Guadeloupe and Martinique, whereas Saint-Barthélemy and Saint-Martin each formed distinct groups.

3.3. Relative Composition and Dominance of Recorded Assemblages

BRUVS observations were concentrated among a limited number of frequently recorded taxa, with marked variability among territories (Figure 6). In Guadeloupe and Martinique, the southern stingray (*Hypanus americanus*) was the most frequently recorded species. This pattern was particularly pronounced in Martinique, where the southern stingray accounted for a substantially larger share of observations than the other species. In Saint-Barthélemy, the nurse shark (*Ginglymostoma cirratum*) was the most frequently recorded species, followed by the southern stingray and the Caribbean reef shark (*Carcharhinus perezi*). In Saint-Martin, the southern stingray was recorded slightly more frequently than the nurse shark, while the Caribbean reef shark also contributed substantially to the recorded assemblage. Observations in Saint-Barthélemy and Saint-Martin were therefore more evenly distributed among several taxa than those in Guadeloupe and Martinique. Cumulative dominance curves were consistent with these descriptive patterns (Figure 6). Martinique exhibited the steepest initial slope, indicating a strong concentration of observations within a limited number of species. In contrast, Saint-Barthélemy and Saint-Martin showed more gradual curves, while Guadeloupe displayed an intermediate pattern.

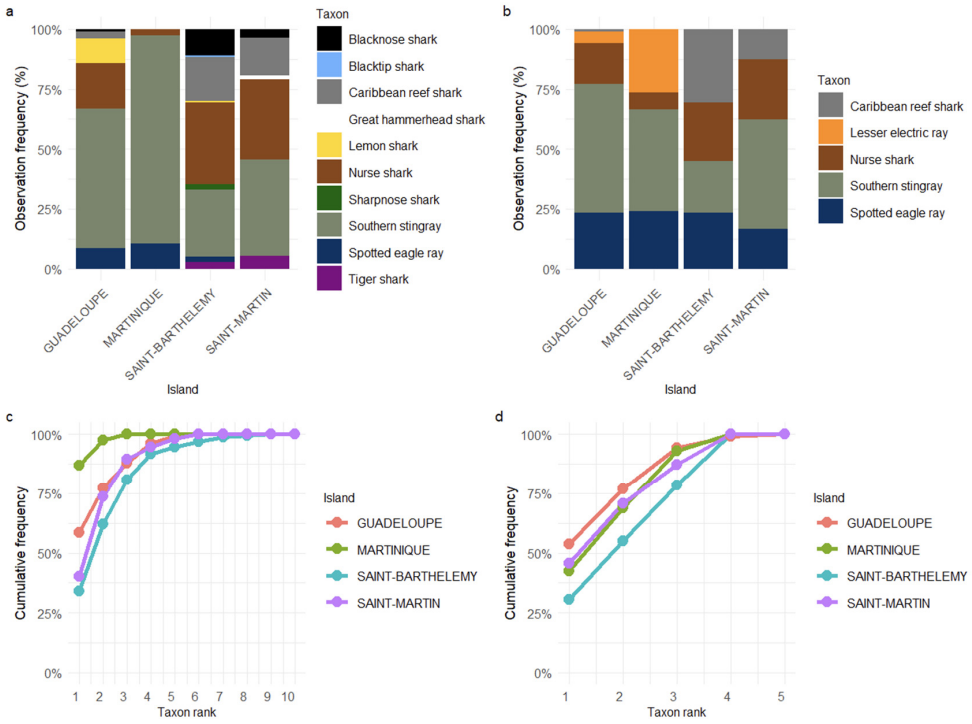


Figure 6. Relative composition and cumulative dominance of recorded elasmobranch assemblages across islands. (**a**) Relative frequency of species among BRUVS observations. (**b**) Relative frequency of species among INA observations. (**c**) Cumulative dominance curves for BRUVS observations. (**d**) Cumulative dominance curves for INA observations.

INA data also showed descriptive differences in the relative composition and dominance of recorded assemblages among territories (Figure 6). The southern stingray was the most frequently recorded species in Guadeloupe, Martinique, and Saint-Martin, whereas the Caribbean reef shark was the most frequently recorded species in Saint-Barthélemy. In Guadeloupe, the spotted eagle ray (*Aetobatus narinari*) and nurse shark also contributed substantially to recorded observations. In Martinique, substantial contributions were also observed for the lesser electric ray (*Narcine bancroftii*) and spotted eagle ray. Observations in Saint-Barthélemy were relatively evenly distributed among the Caribbean reef shark, nurse shark, spotted eagle ray, and southern stingray. These four species also accounted for most observations in Saint-Martin, although the southern stingray was the most frequently recorded. The cumulative dominance curves showed the most gradual increase in Saint-Barthélemy and the steepest initial increase in Guadeloupe, with Martinique and Saint-Martin displaying intermediate profiles (Figure 6).

3.4. Beta Diversity

The Jaccard-based dissimilarity matrices revealed clear variation in elasmobranch assemblage composition among the four territories (Figure 7). The BRUVS heatmap revealed predominantly intermediate to high dissimilarity values, indicating marked contrasts between several territory pairs. A relatively higher similarity was observed between Guadeloupe and Saint-Barthélemy, whereas other pairs, such as Martinique–Saint-Barthélemy, displayed much higher dissimilarity levels. Within the INA dataset, pairwise dissimilarities were generally moderate.

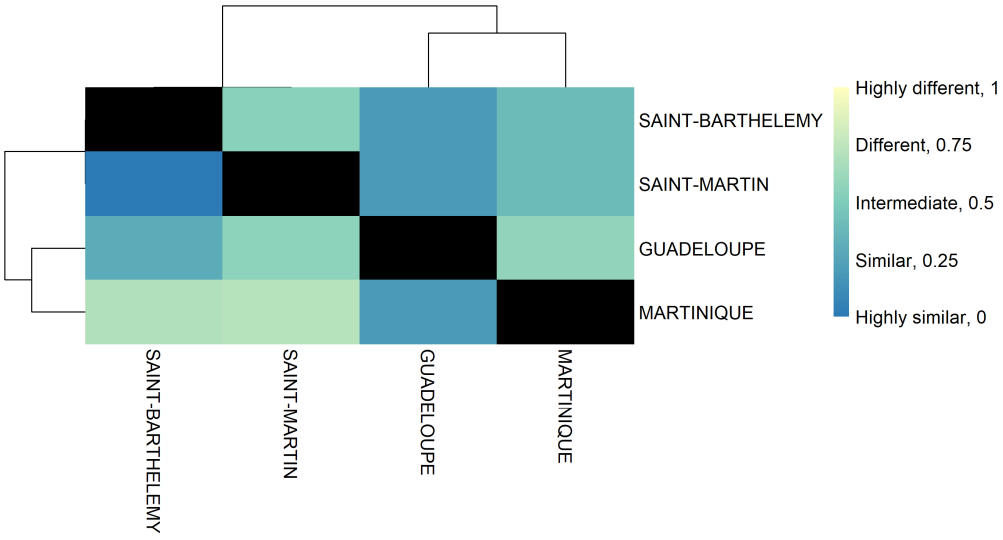


Figure 7. Jaccard dissimilarity heatmaps for the BRUVS dataset (**upper** panel) and the INA dataset (**lower** panel). Black cells indicate that no Jaccard dissimilarity value was calculated for same-island comparisons.

3.5. Frequency of Observation and Dominant Species

The species with the highest encounter rate varied among territories and between datasets (Figures 8–11). Within the BRUVS dataset, the southern stingray (*Hypanus americanus*) had the highest encounter rate in Guadeloupe, Martinique, and Saint-Martin, whereas the nurse shark (*Ginglymostoma cirratum*) had the highest encounter rate in Saint-Barthélemy. Within the INA dataset, the southern stingray had the highest encounter rate in Guadeloupe, Martinique, and Saint-Martin, whereas the Caribbean reef shark (*Carcharhinus perezi*) had the highest encounter rate in Saint-Barthélemy. Detailed species-specific encounter rates are presented below.

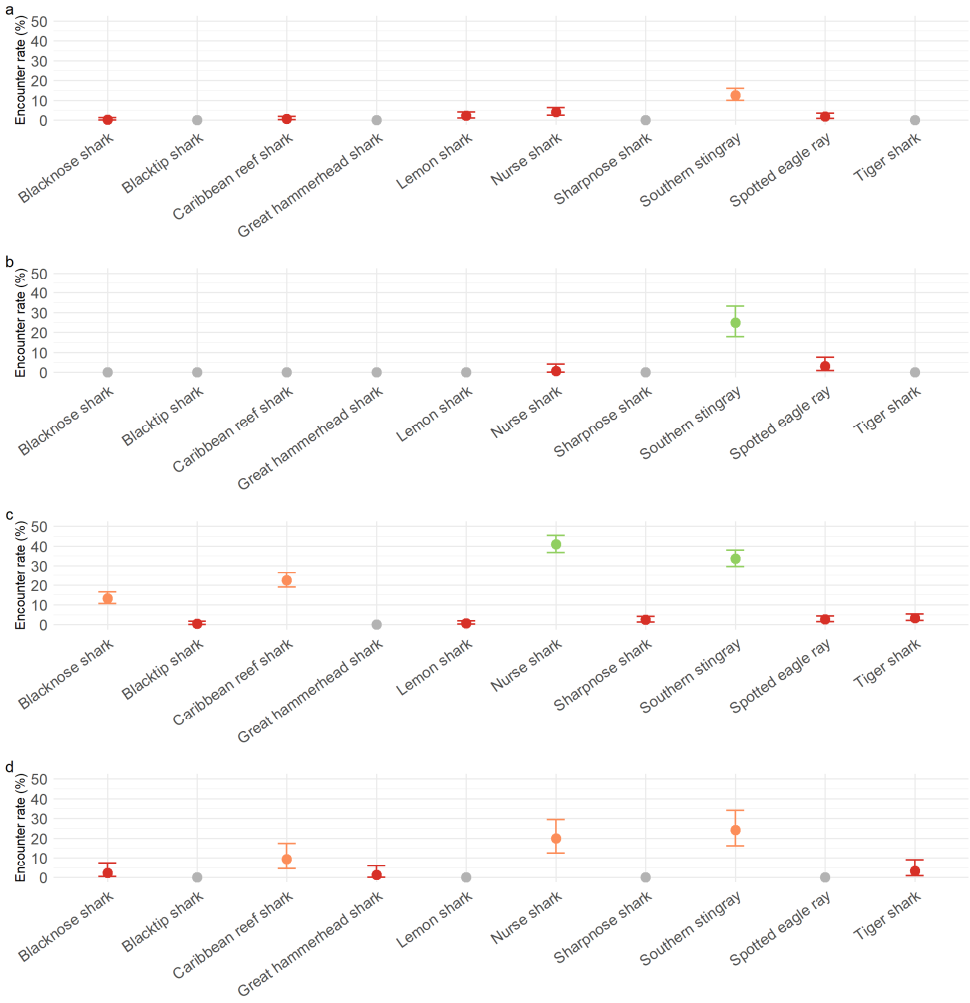


Figure 8. Elasmobranch encounter rates across the French West Indies with BRUVS data. Panels show encounter rates in (**a**) Guadeloupe, (**b**) Martinique, (**c**) Saint-Barthélemy, and (**d**) Saint-Martin. Gray dots indicate species not detected; red, orange, and green dots indicate rare (<5%), occasional (5–25%), and common (>25%) species.

For BRUVS, in Guadeloupe, the southern stingray was the most frequently encountered species (12.63% [9.82–15.89]), followed by the nurse shark (4.07% [2.51–6.22]) and the lemon shark (*Negaprion brevirostris*) (2.24% [1.12–3.97]) (Figures 8 and 9). All other species were infrequently observed (<2%), including the spotted eagle ray (*Aetobatus narinari*), the Caribbean reef shark and the blacknose shark (*Carcharhinus acronotus*). In Martinique, the southern stingray largely dominated the records (25.00% [17.88–33.28]). The spotted eagle ray (3.03% [0.83–7.58]) and the nurse shark (0.76% [0.02–4.15]) showed low encounter rates. In Saint-Barthélemy, the nurse shark (40.95% [36.71–45.30]) and the southern stingray (33.71% [29.68–37.94]) were the most frequently encountered species. They were followed by the Caribbean reef shark (22.29% [18.80–26.09]) and the blacknose shark (13.33% [10.54–16.54]). The spotted eagle ray, sharpnose shark (*Rhizoprionodon* spp.), tiger shark (*Galeocerdo cuvier*), lemon shark and blacktip shark (*Carcharhinus limbatus*) showed comparatively low encounter rates (mean < 3% for all species). In Saint-Martin, the southern stingray (24.21% [16.01–34.08]) and the nurse shark (20.00% [12.49–29.46]) were the most frequently recorded species, followed by the Caribbean reef shark (9.47% [4.42–17.22]). Blacknose shark and tiger shark were recorded at low frequencies (both species had encounter rates below 4%).

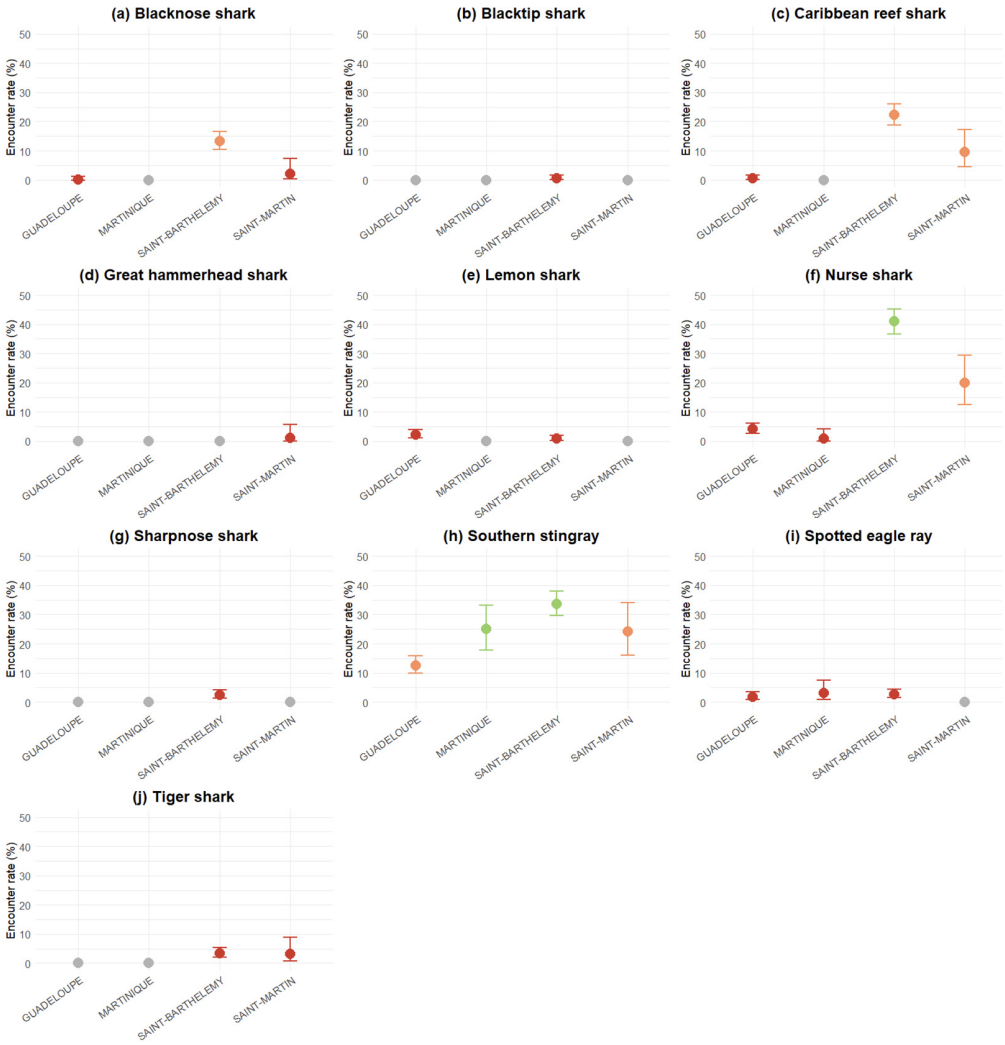


Figure 9. Encounter rates of elasmobranch species across the French West Indies with BRUVS data. Gray dots indicate species not detected; red, orange, and green dots indicate rare (<5%), occasional (5–25%), and common (>25%) species.

Encounter rates differed among species and islands within the INA data (Figures 10 and 11). In Guadeloupe, the southern stingray was the most frequently recorded species (6.28% [5.86–6.72]). It was followed by the spotted eagle ray, the nurse shark, the lesser electric ray (*Narcine bancroftii*) and the Caribbean reef shark (all species had encounter rates below 3%). In Martinique (n = 2190 dives), the southern stingray showed the highest observation encounter rate (2.19% [1.62–2.90]), followed by the lesser electric ray, the spotted eagle ray and the nurse shark. All were, however, observed on <3% of dives. In Saint-Barthélemy, the Caribbean reef shark (41.52% [36.14–47.04]), the nurse shark (33.03% [27.98–38.39]), the spotted eagle ray (31.82% [26.82–37.14]) and the southern stingray (29.09% [24.25–34.32]) were the most frequently recorded species. In Saint-Martin, the southern stingray was the most frequently recorded species (39.56% [35.85–43.36]), followed by the nurse shark (21.48% [18.44–24.78]), the spotted eagle ray (14.37% [11.81–17.25]) and the Caribbean reef shark (10.96% [8.71–13.57]).

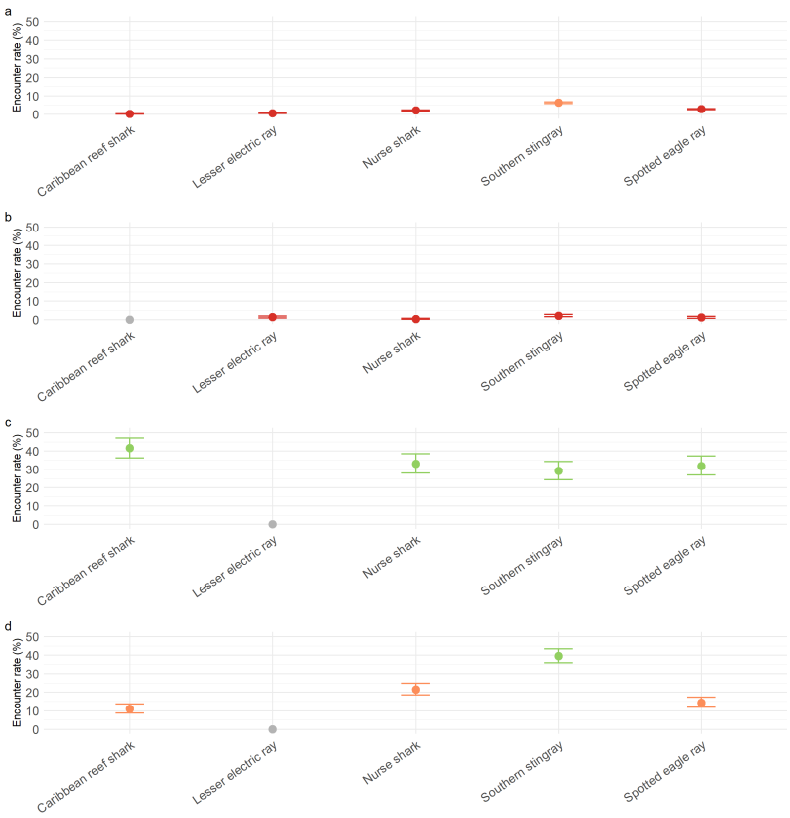


Figure 10. Elasmobranch encounter rates by the French West Indies with INA data. Panels show encounter rates in (**a**) Guadeloupe, (**b**) Martinique, (**c**) Saint-Barthélemy, and (**d**) Saint-Martin. Gray dots indicate species not detected; red, orange, and green dots indicate rare (<5%), occasional (5–25%), and common (>25%) species.

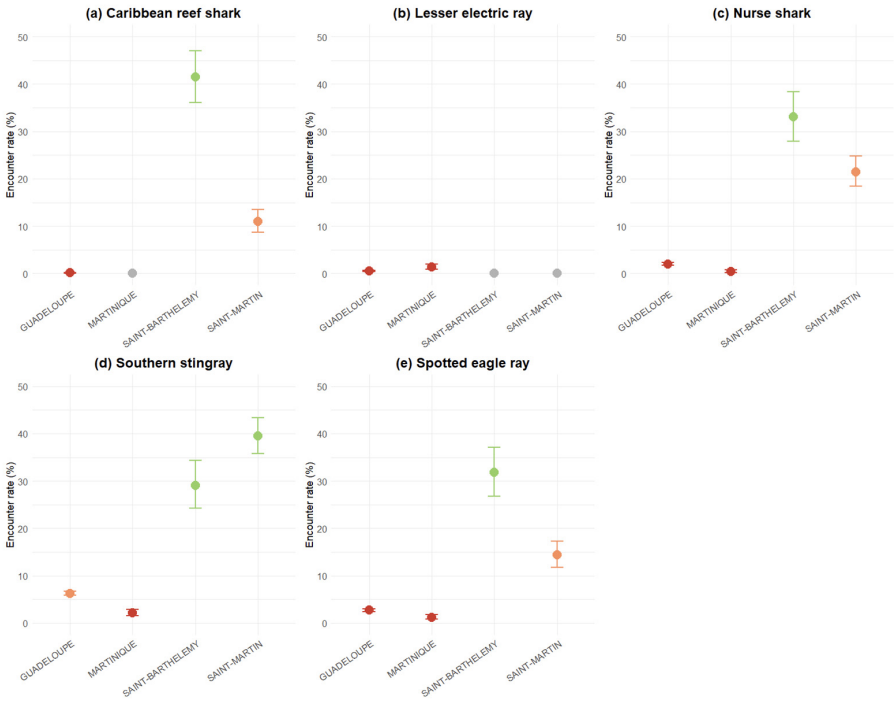


Figure 11. Encounter rates of elasmobranch species across the French West Indies with INA data. Gray dots indicate species not detected; red, orange, and green dots indicate rare (<5%), occasional (5–25%), and common (>25%) species.

4. Discussion

This study provides a comparative assessment of the diversity and structure of recorded coastal elasmobranch assemblages across the four territories of the FWI. Separate analyses of the BRUVS and INA datasets revealed marked inter-island differences in species richness, diversity, assemblage composition and encounter rates. In both datasets, Saint-Barthélemy had the most diverse and evenly represented recorded assemblage, while Guadeloupe and Martinique had the lowest diversity These findings provide a regional baseline for future monitoring, although differences in sampling coverage and species detectability, together with the observational design of the study, limit inferences about the processes responsible for the patterns observed.

4.1. Inter-Island Variation in Elasmobranch Assemblages Within the Caribbean Context

Across the FWI, this study recorded a total of 11 shark and ray taxa: 10 were detected by BRUVS, comprising eight sharks and two rays, and five were recorded by INA (three rays and two sharks). Together, these two methods provided information on nearly 60% of the coastal elasmobranch species whose presence has been confirmed in the FWI [41]. In both datasets, Saint-Barthélemy had the highest mean richness and the highest Shannon and Simpson indices, as well as the most even distribution of observations among species. Saint-Martin occupied an intermediate position, and Guadeloupe and Martinique had lower diversity and stronger dominance by a limited number of species. The Jaccard matrices showed differences in species identity, whereas the descriptive dominance curves showed differences in the relative frequency of recorded taxa. Inter-island differences were therefore not only the number of species recorded, but assemblage composition and structure.

In Saint-Barthélemy, several species contributed regularly to the observations, particularly the Caribbean reef shark (*Carcharhinus perezi*), nurse shark (*Ginglymostoma cirratum*), spotted eagle ray (*Aetobatus narinari*) and southern stingray (*Hypanus americanus*). The same species dominated the records from Saint-Martin, but diversity and encounter rates were lower than in Saint-Barthélemy. Caribbean reef sharks and nurse sharks are also among the most frequently detected reef-associated sharks in several other Caribbean territories, including Saba, Sint Maarten, the Cayman Islands and other islands in the north-eastern Caribbean [8,16,42–46]. The assemblage recorded in Saint-Barthélemy and, to a lesser extent, Saint-Martin was therefore consistent with patterns reported from several reef systems in the region.

In Guadeloupe and Martinique, observations were more strongly concentrated among a limited number of taxa, resulting in less diverse assemblages. Most species were either not detected or classified as rare on the basis of their encounter rates, with the exception of the southern stingray, which accounted for a substantial proportion of detections, particularly in Martinique. Reef-associated sharks, including nurse sharks and Caribbean reef sharks, were rarely recorded. This contrast is notable because both species are frequently detected in several surveys elsewhere in the Caribbean [8,9,16,42–45], with encounter rates exceeding 20% on nearby islands such as Barbuda (Bond, unpublished data). In addition, in the Guadeloupe BRUVS dataset, six of the seven taxa detected had encounter rates below 5%. Few deployments recorded more than one species, and a substantial proportion yielded no elasmobranch detections. Although imperfect detectability cannot be excluded, this pattern appears more consistent with low local occurrence, restricted use of the sampled habitats, or a combination of these factors than with a limitation inherent to the method. Indeed, higher encounter rates for several of these species have been reported from other Caribbean territories using comparable BRUVS protocols [8].

The relative dominance of the southern stingray in Guadeloupe and Martinique might suggest a shift in assemblage composition from sharks towards rays. Such changes

have been proposed in systems where reef-associated shark populations have declined markedly [9]. In the present study, however, southern stingrays were not detected frequently in either territory. Their proportional dominance therefore appears to reflect the rarity of other elasmobranchs in the records rather than a marked increase in their own encounter rates. The observed assemblages are consequently consistent with lower recorded diversity, rather than providing evidence of a shark-to-ray shift [9,13]. Fishing pressure on southern stingrays should also be considered because the species is one of the most frequently caught elasmobranchs in Guadeloupe and Martinique (Beaufort, unpublished data).

Several large, mobile shark species also had low encounter rates in the FWI. Tiger sharks (*Galeocerdo cuvier*), blacktip sharks (*Carcharhinus limbatus*), great hammerheads (*Sphyrna mokarran*) and lemon sharks (*Negaprion brevirostris*) were only recorded occasionally by BRUVS and were not recorded in the INA dataset. Several of these species have been reported only sporadically in BRUVS surveys elsewhere in the Caribbean [17,42–45], suggesting that low encounter rates are not unique to the FWI.

Some taxa had more geographically restricted records within the datasets. Sharpnose sharks (*Rhizoprionodon* spp.) were detected only by Saint-Barthélemy BRUVS. No published records are currently available for Guadeloupe, Martinique or Saint-Martin [41], although occasional observations have been reported from other Caribbean territories [17]. The lesser electric ray (*Narcine bancroftii*) was recorded by INA in Guadeloupe and Martinique, but not in Saint-Martin or Saint-Barthélemy, where it was also absent from the available historical records [41]. The species has nevertheless been reported from the neighboring island of Saint Kitts and Nevis [47]. These apparent distributions may reflect genuine geographical differences but could also arise from the distribution of sampled habitats and time periods, method-specific characteristics or insufficient detectability. More extensive, spatially balanced, and taxon-specific sampling would be required before any regional distribution boundaries could be inferred.

4.2. Factors That May Contribute to Inter-Island Differences

The factors considered below should be treated as non-exclusive hypotheses based on the literature and on known characteristics of the territories. They were not directly evaluated in this study.

The PCAs showed broad overlap in the general environmental conditions covered by sampling sites across the four territories in both the BRUVS and INA datasets. The observed inter-island contrasts are therefore unlikely to arise solely from the sampling of markedly different depth, distances from shore, proximities to benthic habitats or seabed rugosity at the scale considered. The scope of this result is nevertheless limited: the PCAs assessed the general representation of sampled environments rather than testing the effects of these variables on elasmobranch assemblages. Among the physical factors measured, temperature, depth, bathymetric configuration and island geomorphology can influence elasmobranch distribution, movement and habitat use [48–53]. The rugosity measure included in the PCAs was derived from a bathymetric model with a spatial resolution of 100 m. It describes broad variation in seabed relief, but does not capture fine-scale habitat complexity, including micro-rugosity, the three-dimensional structure of reefs, or the availability of small refuges for potential prey. Local differences in habitat complexity that were not represented by the PCA variables could therefore contribute to the variation observed among territories.

Island geomorphology is consequently a relevant avenue for further investigation. Saint-Martin and Saint-Barthélemy lie on the Anguilla Bank, a shallow insular shelf characterised by a relatively extensive and continuous mosaic of reefs, sandy bottoms and

other coastal habitats. This configuration may increase the availability and continuity of habitats used by reef-associated species and benthic taxa. Guadeloupe and Martinique also contain shallow coastal waters, sheltered bays and, locally, extensive lagoon systems, but parts of their coastlines have narrower shelves and steeper bathymetric slopes. Studies in other island systems have shown that shelf width, bathymetric configuration, habitat availability and spatial continuity can be associated with elasmobranch richness, relative abundance and assemblage composition [48,49,54,55]. These findings make geomorphology a plausible hypothesis in the FWI. Testing it would require higher-resolution bathymetric data and habitat maps, together with explicit analyses relating site characteristics to elasmobranch observations.

Biological factors not included in the analyses may also contribute to the observed differences. Coral reef condition, benthic cover, prey availability and the abundance of other predators can affect habitat quality and its use by sharks and rays [3,54,56–61]. In addition, examination of BRUVS footage from reef sites appeared to show more extensive algal cover at several sampled locations in Guadeloupe and Martinique, whereas algal cover appeared more limited at sites in Saint-Martin and Saint-Barthélemy (Figure S4). This qualitative observation may indicate differences in reef condition and provides a useful hypothesis for further study, but reef condition was not quantified using a standardised protocol in the present analysis. Integrating data on prey availability with BRUVS observations would also help to identify factors potentially influencing elasmobranch abundance and diversity [61].

Anthropogenic factors may also be relevant, particularly spatial protection, fishing and other coastal uses. Saint-Barthélemy and Saint-Martin are close to the Yarari Shark Sanctuary and have the highest ratios of no-take area to coastline, at 30.0 and 72.5 ha km^{-1}, respectively, compared with 0.67 ha km^{-1} in Guadeloupe and 0.15 ha km^{-1} in Martinique. Fishing is prohibited in these areas, while other nautical activities are strictly regulated. These measures may create conditions that are more favourable for coastal elasmobranch populations [62,63], although their effects were not tested here. Fishing activity also differs among territories. Gillnets are widely used by coastal fisheries in Guadeloupe and Martinique, where nearly 500 vessels reported using this gear in 2024 [64,65]. Gillnets are relatively non-selective and can result in both targeted and incidental catches of coastal chondrichthyans [66,67]. The available information suggests more limited gillnet use in Saint-Martin and Saint-Barthélemy, mainly during certain periods of the conch fishery, according to oral communications with J. Chalifour and E. Muller [68,69]. Fishing regulations also differ among territories. In Saint-Barthélemy, restrictions introduced in 2015 and strengthened in 2021 extended protection to several shark and ray species and substantially limited the use of gillnets. In Guadeloupe and Martinique, recreational fishing for sharks and rays has been prohibited since 2019, but commercial fishing remains permitted for several species, and some protected or regulated taxa still occur occasionally in the available landing records [41]. Differences in fishing gear, regulation and potentially fishing intensity provide a plausible context for the observed contrasts among territories [8,11]. Further analyses are required to determine whether and how these anthropogenic factors influence coastal elasmobranch populations in the FWI.

Overall, the physical characteristics of the islands, the biological condition of their habitats and anthropogenic pressures represent potential, non-exclusive explanations for the differences observed among territories. Future analyses should examine whether the recorded differences primarily reflect natural island characteristics, human pressures, management measures or interactions among these factors.

4.3. Observational Coverage and Methodological Limitations

BRUVS and INA provided distinct but partly overlapping taxonomic coverage. Four species were detected by both approaches. BRUVS recorded several mobile shark species that were absent from the INA observations, and INA provided the only records of the lesser electric ray. This difference is consistent with other studies that have employed multiple observational sources to document poorly known assemblages [9,14,19,20,42]. At a descriptive level, the two datasets showed a similar pattern across the study territories. Each approach favoured the detection of particular species according to their habitat, behaviour, ecology and response to the sampling protocol. Considered together, the two datasets therefore broadened the observational coverage of coastal assemblages.

Differences in detection between the two approaches, together with their respective limitations, however, should guide interpretation of the results. BRUVS favour the detection of mobile species likely to respond to bait, but may under-represent taxa that are weakly attracted to the equipment or are absent from the bait plume during a deployment [14,17,30,70,71]. INA primarily records species present at recreational dive sites and tolerant of human presence, particularly those associated with reef habitats visited by divers. The spotted eagle ray, for example, was recorded more frequently in the INA data, whereas the lesser electric ray was entirely absent from BRUVS deployments. Conversely, the tiger shark, which is readily recognisable from its characteristic markings, was not recorded by INA, although BRUVS confirmed its presence in areas close to dive sites. Encounter rates also differed between protocols even for species detected by both approaches.

BRUVS follow a standardised protocol that reduces observer-related bias, but their implementation is constrained by equipment costs, the need to mobilise field teams, and the time required to review and analyse video footage [17,18]. These constraints limit spatial and temporal coverage and make continuous, large-scale monitoring difficult. BRUVS may also be more effective for resident, site-associated species than for transient or highly mobile taxa whose wide-ranging movements reduce the probability of encountering the bait plume during a deployment.

INA provides a substantial observational effort, but broad spatial and temporal coverage. The data are nevertheless opportunistic, because they come from recreational dives rather than a randomised design with dedicated sampling effort. The dives do not specifically target elasmobranch detection, and observations are concentrated at accessible and attractive recreational sites, which are often reef habitats. Both features can affect detection probabilities and introduce habitat-selection bias. A further potential source of bias is the identification of morphologically similar species locally grouped as ‘grey sharks’, including the blacknose shark, lemon shark, blacktip shark and Caribbean reef shark. To reduce this bias, participating instructors received training and an illustrated identification guide. Photographs or videos were also collected where possible to confirm identifications.

Unequal sampling effort and spatio-temporal coverage are important limitations of the inter-island comparisons. INA effort ranged from 300 to more than 11,000 dives among territories, and BRUVS effort was also uneven. Sampling periods and seasons differed among territories. These differences may affect the detection of rare species and the recorded encounter rates, particularly where seasonal movements alter local abundance during the year [72,73]. However, rarefaction and extrapolation curves indicated rapid saturation of detectable richness for INA and near-saturation for BRUVS, except in Saint-Martin, where additional species may still be detected. The observed contrasts may therefore be ecologically informative, but they remain subject to sampling effects.

The two approaches provide distinct and partly overlapping observational coverage, but some species remain poorly detected or undetected. Additional targeted surveys could

therefore broaden taxonomic coverage [42,74]. Visual censuses designed for sandy habitats may be particularly useful for inconspicuous benthic species such as the lesser electric ray [75]. Environmental DNA may also contribute to the detection of rarely observed species, although trials conducted in coastal waters of the FWI between 2020 and 2023 did not detect any species additional to those recorded by BRUVS [76].

5. Conclusions

This study documents marked inter-island differences in recorded coastal elasmobranch assemblages across the French West Indies. BRUVS and INA provided distinct but partly overlapping observational coverage. Analyses conducted separately showed the same broad geographical pattern: Saint-Barthélemy had the highest mean richness and diversity and frequent records of reef-associated sharks, Saint-Martin was intermediate in species diversity, and Guadeloupe and Martinique had lower sample-level richness and assemblages dominated by a limited number of taxa. The study provides a regional baseline for the continued monitoring of coastal elasmobranchs in the FWI. Standardised long-term surveys, more balanced spatial and seasonal coverage, and analyses incorporating habitat condition, geomorphology, fishing pressure and management measures will be needed to better understand the factors underlying the observed patterns and to inform territory-specific conservation decisions.

Supplementary Materials: The following supporting information can be downloaded at https://www.mdpi.com/article/10.3390/d18100630/s1, Figure S1: Location of the French West Indies; Figure S2: Sampling Effort across the French West Indies; Figure S3: Jaccard Similarity; Figure S4: Examples screenshots showing advanced brown algal development observed in Guadeloupe and Martinique; Table S1: Depth (m) by Survey Method and Island.

Author Contributions: Conceptualization, O.B.; methodology, O.B.; validation, O.B., M.E.B., M.R.H., P.M. and K.Q.; formal analysis, O.B.; investigation, O.B., M.E.B., M.R.H., P.M. and K.Q.; resources, O.B., M.E.B., M.R.H., P.M. and K.Q.; data curation, O.B.; writing—original draft preparation, O.B.; writing—review and editing, K.Q., M.E.B., M.R.H. and P.M.; visualization, O.B.; supervision, O.B.; project administration, O.B.; funding acquisition, O.B., M.E.B., M.R.H. and P.M. All authors have read and agreed to the published version of the manuscript.

Funding: This study was conducted as part of the 2PRK2 project, carried out by Kap Natirel NGO within the French West Indies Shark Network (Reguar) and supported by the French Biodiversity Agency (OFB) (Convention de subvention OFB-24-1118), the DEAL Guadeloupe (Convention n°DEAL/RN n°2024-237) and the DEAL Martinique (Arrêté de subvention n°202604-003). The 2016 and 2024 BRUVS surveys conducted in Guadeloupe and Martinique were carried out as part of the Global FinPrint project, with financial support from the Paul G. Allen Foundation and in collaboration with Florida International University (FIU). The others data used in this study were collected through several projects funded by DEAL Guadeloupe, DEAL Martinique, the OFB, the SPAW-RAC, the Agence Territoriale de l'Environnement de Saint-Barthélemy (ATE), and the European Union.

Data Availability Statement: All data supporting the findings of this study are available within the article and its Supplementary Materials. Requests for additional datasets can be directed to the corresponding author and will be evaluated on a case-by-case basis.

Acknowledgments: We extend our gratitude to all our partners, including the Association Titè, the Office National des Forêts, the Parc National de Guadeloupe, the ATE, the Université des Antilles, the BOREA laboratory, the AGRNSM, the SCUBA diving clubs that contributed to the INA SCUBA program (full list available at: https://kapnatirel.org/programme-ina-scuba) (accessed on 3 August 2026), as well as all those who assisted with BRUV fieldwork. We extend our sincere gratitude to all those who contributed to the collection and entry of the data used in this study.

Conflicts of Interest: The authors declare no conflicts of interest.

Abbreviations

The following abbreviations are used in this manuscript:

FWI	French West Indies
BRUVS	Baited Remote Underwater Video Systems
INA SCUBA	Indice Nautique d'Abondance SCUBA
MPA	Marine Protected Area
PCA	Principal Component Analysis
NMDS	Non-metric Multidimensional Scaling

References

1. Dedman, S.; Moxley, J.H.; Papastamatiou, Y.P.; Braccini, M.; Caselle, J.E.; Chapman, D.D.; Cinner, J.E.; Dillon, E.M.; Dulvy, N.K.; Dunn, R.E.; et al. Ecological roles and importance of sharks in the Anthropocene Ocean. *Science* **2024**, *385*, adl2362. [CrossRef] [PubMed]
2. Ferretti, F.; Worm, B.; Britten, G.L.; Heithaus, M.R.; Lotze, H.K. Patterns and ecosystem consequences of shark declines in the ocean. *Ecol. Lett.* **2010**, *13*, 1055–1071. [CrossRef] [PubMed]
3. Heithaus, M.R.; Frid, A.; Wirsing, A.J.; Worm, B. Predicting ecological consequences of marine top predator declines. *Trends Ecol. Evol.* **2008**, *23*, 202–210. [CrossRef] [PubMed]
4. Roff, G.; Doropoulos, C.; Rogers, A.; Bozec, Y.-M.; Krueck, N.C.; Aurellado, E.; Priest, M.; Birrell, C.; Mumby, P.J. The ecological role of sharks on coral reefs. *Trends Ecol. Evol.* **2016**, *31*, 395–407. [CrossRef] [PubMed]
5. Heithaus, M.R.; Dunn, R.E.; Farabaugh, N.F.; Lester, E.; Madin, E.; Meekan, M.G.; Papastamatiou, Y.P.; Roff, G.; Vaudo, J.J.; Wirsing, A.J. Advances in our understanding of the ecological importance of sharks and their relatives. In *Biology of Sharks and Their Relatives*, 3rd ed.; Carrier, J.C., Simpfendorfer, C.A., Heithaus, M.R., Yopak, K.E., Eds.; CRC Press: Boca Raton, FL, USA, 2022; pp. 487–521. [CrossRef]
6. Estes, J.A.; Heithaus, M.R.; McCauley, D.J.; Rasher, D.B.; Worm, B. Megafaunal impacts on structure and function of ocean ecosystems. *Annu. Rev. Environ. Resour.* **2016**, *41*, 83–116. [CrossRef]
7. Dulvy, N.K.; Fowler, S.L.; Musick, J.A.; Cavanagh, R.D.; Kyne, P.M.; Harrison, L.R.; Carlson, J.K.; Davidson, L.N.K.; Fordham, S.V.; Francis, M.P.; et al. Extinction risk and conservation of the world's sharks and rays. *eLife* **2014**, *3*, e00590. [CrossRef] [PubMed]
8. MacNeil, M.A.; Chapman, D.D.; Heupel, M.; Simpfendorfer, C.A.; Heithaus, M.R.; Meekan, M.; Harvey, E.; Goetze, J.S.; Kiszka, J.J.; Bond, M.E.; et al. Global status and conservation potential of reef sharks. *Nature* **2020**, *583*, 801–806. Erratum in *Nature* **2020**, *585*, E11. https://doi.org/10.1038/s41586-020-2692-z. [CrossRef] [PubMed]
9. Simpfendorfer, C.A.; Heithaus, M.R.; Heupel, M.R.; MacNeil, M.A.; Meekan, M.; Harvey, E.; Sherman, C.S.; Currey-Randall, L.M.; Goetze, J.S.; Kiszka, J.J.; et al. Widespread diversity deficits of coral reef sharks and rays. *Science* **2023**, *380*, 1155–1160. [CrossRef] [PubMed]
10. Clementi, G.M.; Babcock, E.A.; Valentin-Albanese, J.; Bond, M.E.; Flowers, K.I.; Heithaus, M.R.; Whitman, E.R.; Van Zinnicq Bergmann, M.P.M.; Guttridge, T.L.; O'Shea, O.R.; et al. Anthropogenic pressures on reef-associated sharks in jurisdictions with and without directed shark fishing. *Mar. Ecol. Prog. Ser.* **2021**, *661*, 175–186. [CrossRef]
11. Ward-Paige, C.A.; Mora, C.; Lotze, H.K.; Pattengill-Semmens, C.; McClenachan, L.; Myers, R.A. Large-scale absence of sharks on reefs in the Greater-Caribbean: A footprint of human pressures. *PLoS ONE* **2010**, *5*, e11968. [CrossRef] [PubMed]
12. Estes, J.A.; Terborgh, J.; Brashares, J.S.; Power, M.E.; Berger, J.; Bond, W.J.; Carpenter, S.R.; Essington, T.E.; Holt, R.D.; Jackson, J.B.C.; et al. Trophic downgrading of planet Earth. *Science* **2011**, *333*, 301–306. [CrossRef] [PubMed]
13. Dulvy, N.K.; Pacoureau, N.; Matsushiba, J.H.; Yan, H.F.; VanderWright, W.J.; Rigby, C.L.; Finucci, B.; Sherman, C.S.; Jabado, R.W.; Carlson, J.K.; et al. Ecological erosion and expanding extinction risk of sharks and rays. *Science* **2024**, *386*, eadn1477. [CrossRef] [PubMed]
14. Whitmarsh, S.K.; Fairweather, P.G.; Huveneers, C. What is Big BRUVver up to? Methods and uses of baited underwater video. *Rev. Fish Biol. Fish.* **2017**, *27*, 53–73. [CrossRef]
15. Ward-Paige, C.A.; Pattengill-Semmens, C.; Myers, R.A.; Lotze, H.K. Spatial and temporal trends in yellow stingray abundance: Evidence from diver surveys. *Environ. Biol. Fishes* **2011**, *90*, 263–276. [CrossRef]
16. Kohler, J.; Gore, M.; Ormond, R.; Austin, T.; Olynik, J. The Sharklogger Network—Monitoring Cayman Islands shark populations through an innovative citizen science program. *PLoS ONE* **2025**, *20*, e0319637. [CrossRef] [PubMed]
17. Brooks, E.J.; Sloman, K.A.; Sims, D.W.; Danylchuk, A.J. Validating the use of baited remote underwater video surveys for assessing the diversity, distribution and abundance of sharks in the Bahamas. *Endanger. Species Res.* **2011**, *13*, 231–243. [CrossRef]

18. Goetze, J.S.; Heithaus, M.R.; MacNeil, M.A.; Harvey, E.; Simpfendorfer, C.A.; Heupel, M.R.; Meekan, M.; Wilson, S.; Bond, M.E.; Speed, C.W.; et al. Directed conservation of the world's reef sharks and rays. *Nat. Ecol. Evol.* **2024**, *8*, 1118–1128. [CrossRef] [PubMed]
19. Blanco-Parra, M.-d.-P.; Argaez Gasca, A.; Reyes Rincón, C.A.; Gutiérrez Martínez, N.H.; Niño-Torres, C.A. Citizen science as a tool to get baseline ecological and biological data on sharks and rays in a data-poor region. *Sustainability* **2022**, *14*, 6490. [CrossRef]
20. Ward-Paige, C.A.; Lotze, H.K. Assessing the value of recreational divers for censusing elasmobranchs. *PLoS ONE* **2011**, *6*, e25609. [CrossRef] [PubMed]
21. Goetze, J.S.; Bond, T.; McLean, D.L.; Saunders, B.J.; Langlois, T.J.; Lindfield, S.; Fullwood, L.A.F.; Driessen, D.; Shedrawi, G.; Harvey, E.S. A field and video analysis guide for diver operated stereo-video. *Methods Ecol. Evol.* **2019**, *10*, 1083–1090. [CrossRef]
22. Institut National de la Statistique et des Études Économiques (INSEE). *Populations des Collectivités d'Outre-Mer en 2023*; INSEE: Paris, France, 2025. Available online: https://www.insee.fr/fr/statistiques/8680667 (accessed on 19 August 2026).
23. Office Français de la Biodiversité. L'Office Français de la Biodiversité aux Antilles. Available online: https://ofb.gouv.fr/antilles (accessed on 19 August 2026).
24. Delcroix, E. *Le Protocole INA Scuba: Analyse des Données d'Observation des Tortues Marines en Plongée—Année 2005—Rapport de Mission*; Association Kap Natirel: Vieux-Habitants, France, 2005.
25. Sáenz-Arroyo, A.; Roberts, C.M.; Torre, J.; Cariño-Olvera, M. Using fishers' anecdotes, naturalists' observations and grey literature to reassess marine species at risk: The case of the Gulf grouper in the Gulf of California, Mexico. *Fish Fish.* **2005**, *6*, 121–133. [CrossRef]
26. Flowers, K.I.; Henderson, A.C.; Lupton, J.L.; Chapman, D.D. Site affinity of white-spotted eagle rays *Aetobatus narinari* assessed using photographic identification. *J. Fish Biol.* **2017**, *91*, 1337–1349. [CrossRef] [PubMed]
27. Conan, A.; Dennis, M.M.; Gilbert, K.; Lenain, E.; Bruns, S.; Henderson, A.C. Occurrence of the endangered whitespotted eagle ray *Aetobatus narinari* around the Lesser Antilles island of Saint Kitts: A photo-identification study. *Environ. Biol. Fishes* **2023**, *106*, 1529–1538. [CrossRef]
28. Beber, I.; Hazard, S.; del Puerto, E.H. Spatial ecology and philopatric tendencies of whitespotted eagle ray, *Aetobatus narinari*, in Cozumel, Mexico. *Environ. Biol. Fishes* **2024**, *108*, 591–609. [CrossRef]
29. Bond, M.E.; Valentin-Albanese, J.; Babcock, E.A.; Hussey, N.E.; Heithaus, M.R.; Chapman, D.D. The trophic ecology of Caribbean reef sharks (*Carcharhinus perezi*) relative to other large teleost predators on an isolated coral atoll. *Mar. Biol.* **2018**, *165*, 67. [CrossRef]
30. Harvey, E.S.; Cappo, M.; Butler, J.J.; Hall, N.; Kendrick, G.A. Bait attraction affects the performance of remote underwater video stations in assessment of demersal fish community structure. *Mar. Ecol. Prog. Ser.* **2007**, *350*, 245–254. [CrossRef]
31. Priede, I.G.; Bagley, P.M.; Smith, A.; Creasey, S.; Merrett, N.R. Scavenging deep demersal fishes of the Porcupine Seabight, North-east Atlantic: Observations by baited camera, trap and trawl. *J. Mar. Biol. Assoc. UK* **1994**, *74*, 481–498. [CrossRef]
32. R Core Team. *R: A Language and Environment for Statistical Computing*; R Foundation for Statistical Computing: Vienna, Austria, 2024.
33. MAREX; Aquasearch; Parc Naturel Marin de Martinique; Office Français de la Biodiversité (OFB). *Carte des Habitats Marins Côtiers 0–40 m de la Martinique*; Parc Naturel Marin de Martinique: Martinique, France, 2024; 101p. Available online: https://www.biodiversite-martinique.fr/sites/default/files/carte_des_habitats_marins_cotiers_0-40m_de_la_martinique_pnmm_marex_aquasearch_2024.pdf (accessed on 12 January 2026).
34. Créocéan. *Cartographie des Biocénoses Marines de Saint-Barthélemy*; Dossier 230540; Agence Territoriale de l'Environnement (ATE) de Saint-Barthélemy: Gustavia, France, 2024; 70p. Available online: https://www.agencedelenvironnement.fr/upload/Cartographie-des-biocenose-marines-68c45d3445e42.pdf (accessed on 12 January 2026).
35. DIREN Guadeloupe; CAREX Environnement. *Cartographie du Milieu Marin de la Guadeloupe et de ses Dépendances*; Rapport et Atlas Cartographique; DIREN Guadeloupe: Basse-Terre, France, 1999.
36. i-Sea. *Projet de Cartographie Multi-Thématique de la Réserve Naturelle Nationale de Saint-Martin*; Rapport d'Étude pour l'Association de Gestion de la Réserve Naturelle de Saint-Martin (AGRNSM); AGRNSM: Saint-Martin, France, 2025.
37. GEBCO Compilation Group. *GEBCO 2024 Grid*; British Oceanographic Data Centre: Liverpool, UK, 2024. [CrossRef]
38. Wilson, E.B. Probable inference, the law of succession, and statistical inference. *J. Am. Stat. Assoc.* **1927**, *22*, 209–212. [CrossRef]
39. Brown, L.D.; Cai, T.T.; DasGupta, A. Interval estimation for a binomial proportion. *Stat. Sci.* **2001**, *16*, 101–133. [CrossRef]
40. Beaufort, O.; Prieto-Gonzalez, R. INA Scuba: Mobilization of French West Indies scuba diving clubs to rapidly improve knowledge on threatened marine species. Case of the white-spotted eagle ray (*Aetobatus narinari*). *Gulf Caribb. Res.* **2025**, *36*, GCFI19–GCFI35. [CrossRef]
41. Beaufort, O. Sharks, rays, and chimeras of the French West Indies: Diversity, status and conservation needs assessment. *Caribb. J. Sci.* **2025**, *55*, 271–302. [CrossRef]

42. Leurs, G.; de Graaf, M.; Hassell-Knijff, D.; Kuramae Izioka, L.A.; van Looijengoed, W.; Meijer zu Schlochtern, M.P.; Bervoets, T.; Nagelkerke, L.A.J.; Winter, H.V. An integrated baseline assessment of reef-associated sharks around Saba (Dutch Caribbean), combining three methods: Stereo-BRUVs, telemetry and citizen science. *R. Soc. Open Sci.* **2025**, *12*, 241754. [CrossRef] [PubMed]
43. Kramer, O.; Odinga, J. Baited Remote Underwater Stereo-Video (Stereo-BRUV) Survey as a Basis for Elasmobranch Conservation and Management on Sint Maarten, Dutch Caribbean. Bachelor's Thesis, Hogeschool Van Hall Larenstein, Leeuwarden, The Netherlands, 2015.
44. Stoffers, T.; de Graaf, M.; Winter, H.V.; Nagelkerke, L.A.J. Distribution and ontogenetic habitat shifts of reef-associated shark species in the northeastern Caribbean. *Mar. Ecol. Prog. Ser.* **2021**, *665*, 145–158. [CrossRef]
45. Bruns, S.; Henderson, A.C. A baited remote underwater video system (BRUVS) assessment of elasmobranch diversity and abundance on the eastern Caicos Bank (Turks and Caicos Islands); an environment in transition. *Environ. Biol. Fishes* **2020**, *103*, 1001–1012. [CrossRef]
46. Simpfendorfer, C.A.; Heupel, M.R.; Kendal, D. Complex human-shark conflicts confound conservation action. *Front. Conserv. Sci.* **2021**, *2*, 692767. [CrossRef]
47. Driggers, W.B., III; Carlson, J. *Narcine bancroftii*. The IUCN Red List of Threatened Species 2019, e.T63142A3121523. Available online: https://www.iucnredlist.org/species/63142/3121523 (accessed on 19 August 2026).
48. Farabaugh, N.F.; Bond, M.E.; Chapman, D.; Clua, E.; Harborne, A.R.; Heupel, M.; Kiszka, J.J.; Heithaus, M.R. Incorporating environmental factors is critical for determining conservation baselines for relative abundance of sharks on coral reefs. *Mar. Ecol. Prog. Ser.* **2024**, *736*, 93–105. [CrossRef]
49. Farabaugh, N.F.; Bond, M.E.; Chapman, D.; Clua, E.; Dedman, S.; Harborne, A.R.; Kiszka, J.J.; Heupel, M.; Heithaus, M.R. Elasmobranch species richness and assemblage composition in the world's largest shark sanctuary. *Mar. Ecol. Prog. Ser.* **2025**, *753*, 155–173. [CrossRef]
50. Pörtner, H.O.; Knust, R. Climate change affects marine fishes through the oxygen limitation of thermal tolerance. *Science* **2007**, *315*, 95–97. [CrossRef] [PubMed]
51. Sims, D.W.; Wearmouth, V.J.; Southall, E.J.; Hill, J.M.; Moore, P.; Rawlinson, K.; Hutchinson, N.; Budd, G.C.; Righton, D.; Metcalfe, J.D.; et al. Hunt warm, rest cool: Bioenergetic strategy underlying diel vertical migration of a benthic shark. *J. Anim. Ecol.* **2006**, *75*, 176–190. [CrossRef] [PubMed]
52. Hight, B.V.; Lowe, C.G. Elevated body temperatures of adult female leopard sharks, *Triakis semifasciata*, while aggregating in shallow nearshore embayments: Evidence for behavioral thermoregulation? *J. Exp. Mar. Biol. Ecol.* **2007**, *352*, 114–128. [CrossRef]
53. Cartamil, D.; Wegner, N.C.; Aalbers, S.; Sepulveda, C.A.; Baquero, A.; Graham, J.B. Diel movement patterns and habitat preferences of the common thresher shark (*Alopias vulpinus*) in the Southern California Bight. *Mar. Freshw. Res.* **2010**, *61*, 596–604. [CrossRef]
54. Espinoza, M.; Cappo, M.; Heupel, M.R.; Tobin, A.J.; Simpfendorfer, C.A. Quantifying shark distribution patterns and species-habitat associations: Implications of marine park zoning. *PLoS ONE* **2014**, *9*, e106885. [CrossRef] [PubMed]
55. Schlaff, A.M.; Heupel, M.R.; Simpfendorfer, C.A. Influence of environmental factors on shark and ray movement, behaviour and habitat use: A review. *Rev. Fish Biol. Fish.* **2014**, *24*, 1089–1103. [CrossRef]
56. Sandin, S.A.; Smith, J.E.; DeMartini, E.E.; Dinsdale, E.A.; Donner, S.D.; Friedlander, A.M.; Konotchick, T.; Malay, M.; Maragos, J.E.; Obura, D.; et al. Baselines and degradation of coral reefs in the Northern Line Islands. *PLoS ONE* **2008**, *3*, e1548. [CrossRef] [PubMed]
57. Rizzari, J.R.; Frisch, A.J.; Connolly, S.R. How robust are estimates of coral reef shark depletion? *Biol. Conserv.* **2014**, *176*, 39–47. [CrossRef]
58. Collins, A.B.; Heupel, M.R.; Motta, P.J. Residence and movement patterns of cownose rays *Rhinoptera bonasus* within a south-west Florida estuary. *J. Fish Biol.* **2007**, *71*, 1159–1178. [CrossRef]
59. Jaine, F.R.A.; Couturier, L.I.E.; Weeks, S.J.; Townsend, K.A.; Bennett, M.B.; Fiora, K.; Richardson, A.J. When giants turn up: Sighting trends, environmental influences and habitat use of the manta ray *Manta alfredi* at a coral reef. *PLoS ONE* **2012**, *7*, e46170. [CrossRef] [PubMed]
60. Berthe, C.; Lecchini, D. Influence of boat noises on escape behaviour of white-spotted eagle ray *Aetobatus ocellatus* at Moorea Island (French Polynesia). *Comptes Rendus Biol.* **2016**, *339*, 99–103. [CrossRef] [PubMed]
61. Harborne, A.R.; Zuercher, R.; Heithaus, M.R.; Clementi, G.M.; Whitman, E.R.; Chapman, D.D. The need to manage prey fish biomass to support shark conservation. *Anim. Conserv.* **2026**, 1–12. [CrossRef]
62. Robbins, W.D.; Hisano, M.; Connolly, S.R.; Choat, J.H. Ongoing collapse of coral-reef shark populations. *Curr. Biol.* **2006**, *16*, 2314–2319. [CrossRef] [PubMed]
63. Bond, M.E.; Babcock, E.A.; Pikitch, E.K.; Abercrombie, D.L.; Lamb, N.F.; Chapman, D.D. Reef sharks exhibit site-fidelity and higher relative abundance in marine reserves on the Mesoamerican Barrier Reef. *PLoS ONE* **2012**, *7*, e32983. [CrossRef] [PubMed]
64. Institut Français de Recherche pour l'Exploitation de la Mer (IFREMER). *Activité des Navires de Pêche 2024—Lieu d'Immatriculation: Pointe-à-Pitre*; Système d'Informations Halieutiques, Ifremer: Plouzané, France, 2024.

65. Institut Français de Recherche pour l'Exploitation de la Mer (IFREMER). *Activité des Navires de Pêche 2024—Lieu d'Immatriculation: Fort-de-France*; Système d'Informations Halieutiques, Ifremer: Plouzané, France, 2024.
66. Pérez-Jiménez, J.C.; Sosa-Nishizaki, O. Reproductive biology of the brown smoothhound shark *Mustelus henlei*, in the northern Gulf of California, México. *J. Fish Biol.* **2008**, *73*, 1244–1261. [CrossRef]
67. Stevens, J.D.; Bonfil, R.; Dulvy, N.K.; Walker, P.A. The effects of fishing on sharks, rays and chimaeras (chondrichthyans), and the implications for marine ecosystems. *ICES J. Mar. Sci.* **2000**, *57*, 476–494. [CrossRef]
68. Chalifour, J. Personal communication (oral conversation), 5 May 2024.
69. Muller, E. Personal communication (oral conversation), 5 May 2024.
70. Chao, A.; Gotelli, N.J.; Hsieh, T.C.; Sander, E.L.; Ma, K.H.; Colwell, R.K.; Ellison, A.M. Rarefaction and extrapolation with Hill numbers: A framework for sampling and estimation in species diversity studies. *Ecol. Monogr.* **2014**, *84*, 45–67. [CrossRef]
71. Cappo, M.; Speare, P.; De'ath, G. Comparison of baited remote underwater video stations (BRUVS) and prawn (shrimp) trawls for assessments of fish biodiversity in inter-reefal areas of the Great Barrier Reef Marine Park. *J. Exp. Mar. Biol. Ecol.* **2004**, *302*, 123–152. [CrossRef]
72. Cook, N.D.; Clementi, G.M.; Flowers, K.I.; Fanovich, L.; Cable, J.; Perkins, S.E.; Wothke, A.; Mohammed, R.S.; Chapman, D.D. Elasmobranch diversity around the southern Caribbean island of Tobago: Opportunities for conservation in a regional trade hub. *J. Mar. Biol. Assoc. UK* **2024**, *104*, e8. [CrossRef]
73. Brooks, E.J.; Sims, D.W.; Danylchuk, A.J.; Sloman, K.A. Seasonal abundance, philopatry and demographic structure of Caribbean reef shark (*Carcharhinus perezi*) assemblages in the north-east Exuma Sound, The Bahamas. *Mar. Biol.* **2013**, *160*, 2535–2546. [CrossRef]
74. Boussarie, G.; Bakker, J.; Wangensteen, O.S.; Mariani, S.; Bonnin, L.; Juhel, J.-B.; Kiszka, J.J.; Kulbicki, M.; Manel, S.; Robbins, W.D.; et al. Environmental DNA illuminates the dark diversity of sharks. *Sci. Adv.* **2018**, *4*, eaap9661. [CrossRef] [PubMed]
75. Garzon, F.; Graham, R.T.; Baremore, I.; Castellanos, D.; Salazar, H.; Xiu, C.; Seymour, Z.; Witt, M.J.; Hawkes, L.A. Nation-wide assessment of the distribution and population size of the data-deficient nurse shark (*Ginglymostoma cirratum*). *PLoS ONE* **2021**, *16*, e0256532. [CrossRef] [PubMed]
76. Beaufort, O.; Kap Natirel. *Suivi des Populations Côtières de Requins et de Raies Dans les Antilles Françaises par la Méthode de l'ADN Environnemental, Phase de Test*; Association Kap Natirel: Guadeloupe, France, 2023; 23p.

Disclaimer/Publisher's Note: The statements, opinions and data contained in all publications are solely those of the individual author(s) and contributor(s) and not of MDPI and/or the editor(s). MDPI and/or the editor(s) disclaim responsibility for any injury to people or property resulting from any ideas, methods, instructions or products referred to in the content.