Research Article

Mediterranean Marine Science
Indexed in WoS (Web of Science, ISI Thomson) and SCOPUS
www.hcmr.gr
DOI: https://doi.org/10.12681/mms.40448

Think FAST:
Applying citizen science to survey the ecological status of fish assemblages in the Côte d'Azur

Isaac BEBER[1,2], Florence LACROSSE[1,2], Patrice FRANCOUR[3], Damien ELOIRE[2], and Samuel JEGLOT[2]

[1] Université Côte d'Azur, Grand Château, 28 Av. Valrose, 06103 Nice, France
[2] NaturDive, 1 Av. des Broussailles, 06400 Cannes, France
[3] ECOSEAS Laboratory, 28 Av. Valrose, 06103 Nice, France

Corresponding author: Isaac BEBER; isaac.beber@doane.edu

Contributing Editor: Stelios SOMARAKIS

Received: 13 February 2025; Accepted: 08 July 2025; Published online: 15 October 2025

Abstract

Fish assemblages are vital for the functioning of Mediterranean coastal ecosystems, yet they have long been overexploited by recreational and professional fishing. In response to this overexploitation, No-take Zones (NTZs) have been applied, resulting in significant conservation benefits. This study, conducted by citizen scientists and members of the scientific community in the north-western (NW) Mediterranean Sea, investigated spatial trends in fish assemblages targeted by small-scale and recreational fisheries using the Fish Assemblage Survey Technique (FAST). A total of 1356 fifteen-minute replicate counts, corresponding to 452 hours in the water over a six-year period (2018 – 2023) were analyzed. Results indicated that the ecological status of fish assemblages were spatially dependent, where *poor* fish assemblage health was observed at dive sites situated in the Lérins Islands, highlighting an area of conservation priority. NTZs were associated with a higher presence of small ($p < 0.001$) and large ($p < 0.001$) size classes of Serranidae fishes. They were also associated with a higher presence of large, threatened carnivorous species: *Dentex dentex* ($p < 0.001$), *Epinephelus marginatus* ($p < 0.001$), and *Sciaena umbra* ($p < 0.001$). Results from this study suggested that data collected by citizen scientists using FAST was robust enough to meet the same conclusions as researchers. We conclude on the positive application of citizen science programs to assess the ecological status of fish assemblages; fill the need for simple, effective monitoring in Marine Protected Areas (MPAs) and understudied areas; and inform conservation policies focused on threatened habitats and associated marine biodiversity in the French region Provence-Alpes-Côte d'Azur (PACA).

Keywords: Fish assemblage; UVC; FAST; Citizen science; Fisheries overexploitation.

Introduction

Fish communities are crucial to Mediterranean coastal ecosystems, yet they have long been overexploited by recreational and professional fishing (Couvray, 2020; Rey *et al.*, 2023). In the Mediterranean Sea, small-scale professional fisheries (SSF, boats less than 12 m) represent more than 80% of the total fishing fleet and provide direct and indirect socio-economic benefits to coastal Mediterranean communities (Cavallé *et al.*, 2020; Penca & Said, 2023). However, poor fishing practices and overexploitation by SSFs have led to a slow degradation of marine biodiversity, and the conservation status of many pelagic fish species (Hussein *et al.*, 2011; Lloret *et al.*, 2020). One of the most recognized conservation tools are Marine Protected Areas (MPAs), specifically designed for enhancing marine biodiversity, safeguarding ecosystem services and providing positive social benefits (Kayal *et al.*, 2020; Grorud-Colvert *et al.*, 2021). Often applied in association with MPAs, No-Take Zones (NTZs) manage overexploitation by artisanal and recreational fisheries reestablishing fish assemblages in the process. Yet, monitoring fish assemblages can be difficult as fish fulfill a wide range of ecological roles in an ecosystem (Villéger *et al.*, 2017) and inhabit a variety of different habitats (Pörtner *et al.*, 2010).

Multiple techniques have been applied to assess fish assemblages in the Mediterranean such as fishery surveys or experimental fishing (Harmelin-Vivien & Francour 1992; Seytre & Francour, 2008), underwater video-based surveys (Nalmpanti *et al.*, 2023), eDNA metabarcoding (Boulanger *et al.*, 2021; Rey *et al.*, 2023), underwater visual census (Seytre & Francour, 2009, 2014; Ben Lamine *et al.*, 2018; Couvray, 2020, Marengo *et al.*, 2021) or a combination herein. One of the most common monitoring techniques is underwater visual census (UVC). UVC is a non-destructive, non-invasive sampling technique used to estimate abundance, size structure, and fish biomass, as well as quantify species richness (Rey *et al.*, 2023). UVC methodologies have different configurations that may include transect lengths, observation ranges, and/or survey times (Pais & Cabral, 2018). UVC methodologies, traditionally used by researchers or members of the scientific community, are now being applied using citizen science.

Citizen science has grown rapidly in popularity as a valuable tool to engage citizens in conservation and research outputs, stimulate environmental education and drive policy changes (Dunkley, 2017; Kelly *et al.*, 2020; Garcia-Soto *et al.*, 2021). In the Mediterranean, citizen science has been applied to species-specific research that aims to model shifts in abundance, spatial distribution, and population (Krželj *et al.*, 2020; Castejón-Silvo *et al.*, 2023; Bosso *et al.*, 2024), as well as in response to ecology-based research. Programs have also been applied in wide-ranging biodiversity assessments that aim to incorporate many species (Rey *et al.*, 2023). This may include research focused on monitoring fish assemblages, a topic widely studied in the Mediterranean (Ghanem & Soussi, 2017; Ben Lamine *et al.*, 2018; Čižmek *et al.*, 2020), and along French coastlines (Seytre & Francour, 2008, 2009, 2014; Couvray, 2020; Marengo *et al.*, 2021; Rey *et al.*, 2023).

One UVC derivative is the Fish Assemblage Survey Technique (FAST), designed for its simplistic data acquisition. Similar to traditional UVC methodologies, indices calculated from FAST can be used to describe the structure and composition of fish assemblages (Ben Lamine *et al.*, 2018). To our understanding, FAST has been applied along continental French coastlines (Seytre & Francour, 2008, 2009; Francour *et al.*, 2013; Francour, 2017), in Corsica (Francour *et al.*, 2011; Marengo *et al.*, 2021), along the Croatian Adriatic coast (Čižmek *et al.*, 2020) and in Tunisian marine waters (Ben Lamine, 2017; Ghanem & Soussi, 2017; Ben Lamine *et al.*, 2018). Ben Lamine *et al.* (2018) found that, through its simplified proxies, FAST is a valuable tool in efficiently monitoring fish assemblages in understudied areas.

This research aimed to assess the ecological status of fish assemblages, and to identify spatial trends in the Côte d'Azur using FAST, adopted by NaturDive in 2018 (naturdive.com /nos-actions/observatoire). In addition, we set out to demonstrate the positive application of FAST to engage citizens in conservation and research outputs, to fill the need for simple and effective monitoring programs in MPAs, and to inform conservation policy in the French region Provence-Alpes-Côte d'Azur (PACA).

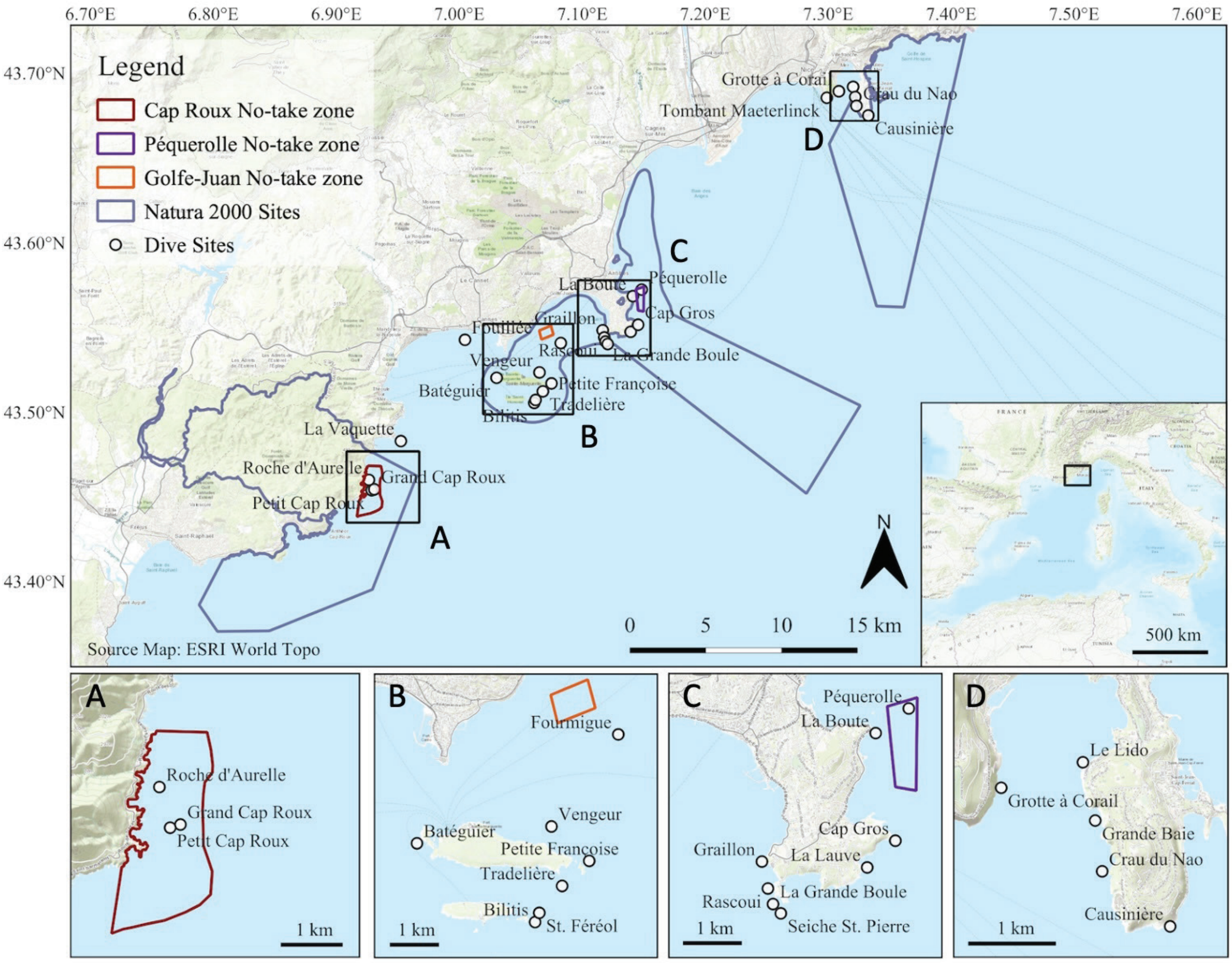

Fig. 1: Map of study region. White dots correspond to the 26 dive sites located along French coastlines. Natura 2000 zones are observed in light blue; the Cap Roux fishing containment in dark red (A), the Golfe-Juan NTZ in orange (B), and the Péquerolle NTZ in violet (C). La Vaquette, Fouillée, Le Lido, Tombant Maeterlinck and Grotte à Corail are located outside of a Natura 2000 zone.

Materials and Methods

Study Location

This study focused on 26 dive sites situated in the north-west (NW) Mediterranean Sea in the French region Provence-Alpes-Côte d'Azur (Fig. 1). The Côte d'Azur, also known as the French Riviera, is in southeastern France, sharing a border with Monaco and Italy to the east. Dive sites in this study were situated off the rocky coastlines of Estérel - Saint Raphaël, Cannes, Golfe-Juan, the Lérins Islands, Antibes, and Nice; sandflats, seagrass beds, and coralligenous assemblages were present at each dive site. Three Natura 2000 sites encompassed the study region starting from the geographical west: 1) Esterél (15,088 ha); 2) Baie and Cap d'Antibes - Lérins Islands (13,597 ha); 3) Cap Ferrat in Nice (8,958 ha); 21 of 26 dive sites were located inside a Natura 2000 site. The study areas also included the Cap Roux NTZ (445 ha, established 2003 – Fig. 1A) located inside Natura 2000 site Estérel, as well as the Golfe-Juan NTZ (50 ha, established 1980 – Fig. 1B), and the Péquerolle NTZ (67 ha, established 2020 – Fig. 1C) located inside Natura 2000 site Cap d'Antibes - Lérins Islands. NTZs were established by fishing tribunals (*prud'homie de pêche)*; recreational and professional fishing is strictly prohibited inside these zones.

Data Collection

FAST was originally developed by Patrice Francour in 1999 to take into account a pre-established list of 23 fish species belonging to 12 fish families, targeted by SSF and recreational fisheries (e.g., spear fishing, net fishing or angling), some of which are patrimonial, protected or rare (Seytre & Francour, 2008, 2009; Francour *et al.*, 2011). Additionally, the protocol considered six 'joker' species,

Table 1. FAST species list, consisting of 23 main species and six 'joker' species (★). The tilde (~) refers to species that are not differentiated during data collection as they are morphologically similar and inhabit similar ecological niches. Trophic level as described by fishbase.org.

Family	Species	French Common	English Common	Trophic Level
Labridea	*Symphodus tinca*	Crénilabre paon	Peacock wrasse	3.3
	Labrus merula	Labre merle	Brown wrasse ~	3.6
	Labrus viridis	Labre vert	Green wrasse ~	3.9
Sparidae	*Sarpa salpa*	Saupe	Salema porgy	2.0
	Diplodus annulis	Sparaillon	Annular seabream	3.6
	Diplodus vulgaris	Sar a tête noire	Two-banded seabream	3.5
	Diplodus sargus	Sar commun	Common seabream	3.4
	Diplodus puntazzo	Sar a museau pointu	Sharpsnout bream	3.2
	Diplodus cervinus	Sar tambour	Zebra seabream	3.0
	Spondyliosoma cantharus	Dorade grise	Black seabream	3.3
	Sparus aurata	Dorade royale	Gilt-head bream	3.7
	Dentex dentex	Denti	Common dentex	4.5
	Lithognathus mormyrus	Marbré	Sand steenbras ★	3.4
	Pagrus pagrus	Pagre	Red porgy ★	3.9
Serranidae	*Serranus cabrilla*	Serran-chèvre	Comber	3.4
	Serranus scriba	Serran écriture	Painted comber	3.8
	Epinephelus marginatus	Mérou brun	Dusky grouper	4.4
Myliobatidae	*Myliobatis aquila*	Raie-aigle	Eagle ray ★	3.6
	Dasyatis pastinaca	Raie-pastenague	Common stingray ★	4.1
Mullidae	*Mullus surmuletus*	Rouget-barbet de roche	Striped red mullet ~	3.5
	Mullus barbatus	Rouget-barbet de vase	Red mullet ~	3.1
Mugilidae	*Mugil* spp. & *Chelon* spp.	Mulet	Mullet	2.5
Scorpaenidae	*Scorpaena scorfa*	Chapon	Red scorpionfish	4.3
Congridae	*Conger conger*	Congre	European conger	4.3
Muraenidae	*Muraena helena*	Murène	Mediterranean moray	4.2
Carangidae	*Seriola dumerilli*	Sériole	Greater amberjack	4.5
Sciaenidae	*Sciaena umbra*	Corb	Brown maegre	3.8
Sphyraenidae	*Sphyraena* spp.	Barracuda	Yellowmouth barracuda	4.0
Moronidae	*Dicentrarchus labrax*	Loup, Bar	European seabass	3.5
Phycidae	*Phycis phycis*	Mostelle	Forkbeard ★	3.9
Syngnathidae	*Hippocampus* spp.	Hippocampe	Seahorse ★	3.5

believed to increase the conservation value of the area (Table 1). Species were chosen to represent the principal habitats in the Mediterranean (e.g., sandflats, seagrass beds, rocky coasts, and coralligenous assemblages) their trophic level chain. Similar to the one conducted by Ben Lamine (2017), our data collection approach required at least six replicates (Table 3) of 15 minutes at each dive site carried out between 5 and 25 m depth along a random pathway using SCUBA diving; each 60-minute dive was independent of each other. The protocol focused on presence/absence of fish species and two size classifications (small/medium & large – Ben Lamine, 2017). Large fish (L) were considered as fish longer than two-thirds of the maximum size for each species, and small/medium fish (SM), shorter than two-thirds of their maximum lengths (Seytre & Francour, 2008, 2009; Ben Lamine, 2017). Maximum lengths of each species were obtained from fishbase.org.

Prior to executing the FAST protocol, citizen scientists participated in one multi-hour theoretical training course and at least two practical trial SCUBA dives. Theoretical sessions were focused on species behavior and included a species identification quiz, while trial dives evaluated citizen scientists' ability to accurately identify fish species and their size classifications under water. Once little discrepancies were noted between new and experienced FAST protocol users, observations were retained in the final database. Data was collected using an underwater diving slate (Fig. App. 1).

Data Analysis

For each replicate (15-minute census), a base index (I) was calculated that considered the number of species observed, their estimated sizes, and the presence of high value species recognized for their commercial or ecological significance (e.g., *Epinephelus marginatus*, *Phycis phycis*, *Sciaena umbra* – Francour, 2017; Ben Lamine, 2017). Large individuals received a coefficient of 2, compared to a score of 1 for small individuals. The base index (I) was calculated by summing the various scores obtained. Conservation coefficients were also applied to species with high conservation value (x2), in consideration of both size classes. These species scores were multiplied by two to account for the heritage or remarkable aspect associated with the species (Francour, 2017). As seen in Seytre & Francour (2008), six indices were calculated as proxies of fish assemblage structure and health.

<u>*Mean Index (MI)*:</u> the average of six base index values (one base index for each 15-minute census); the MI is used as a proxy for species density and size class occurrence (Ben Lamine, 2017).

<u>*Cumulative index (CI)*</u>: calculated following the combination of all six replicates; CI is used as a proxy for species richness and size distribution (Ben Lamine, 2017).

<u>*Relative species richness (RSR)*</u>: the average number of species present during census relative to the total number of species on the predefined list, represented as a percentage.

<u>*Carnivores Proportion (CP)*</u>: the proportion of observed carnivorous species; only large individuals with trophic level > 3.7 were considered as carnivores (Ben Lamine, 2017). Trophic levels were taken from fishbase.org (Table 1).

<u>*Large Proportion (LP)*</u>: the proportion of large individuals among the species listed (Table 1).

<u>*Coefficient of variation (CV)*</u>: variability of the six replicates (one for each 15-minute count); the CV allowed for assessment of fish assemblage variability. If CV was inferior to 30%, this indicated a stable fish assemblage. If the CV was superior to 30%, this either indicated a large variation between replicate counts or between observers (Francour, 2017), or a degradation in fish population structure (Marengo *et al.*, 2021).

Based on the results of the following five calculated proxies: MI, CI, RSR, CP and LP (Table. App. 1), each dive site was assigned a cluster from 1 to 5 indicating the ecological status of fish assemblages (Table 2, Francour, 2017).

Only data collected in 2018-2023 between the months of May-October were included in the analysis (yearly sampling effort found in Table 3). Similarly, dive sites which incorporated less than six replicate counts were excluded from analysis, and fish assemblage proxies, MI, CI, RSR, CP, and LP, were not calculated at these sites. A cluster from 1 to 5 indicating the ecological status of fish

Table 2. Five cluster groups suggesting the ecological status of fish assemblages ranging from 1: "Excellent" to 5: "Very Poor" (Francour, 2017); MI: mean index, CI: cumulative index, RSR: relative species richness, CP: proportion of carnivores, LP: proportion of large individuals.

	MI	CI	RSR	CP	LP	Ecological Status
Cluster 1	>30	>55	>70	>15	>30	Excellent
Cluster 2	<30	40-55	>60	>15	>20	Good
Cluster 3	20-25	35-55	50-60	<15	>20	Average
Cluster 4	20-25	35-55	>65	<12	>20	Poor
Cluster 5	<20	<35	<65	<10	<20	Very Poor

Table 3. FAST sampling effort over the last six years between the months May to October, incorporating 26 dive sites located in Cannes, Cap d'Antibes, Estérel, Lérins Islands and Nice. Here, the number of replicate counts conducted at each site per year, with the total number assumed during the study period, are observed. The last column shows the total number of years over which data was recorded at each dive site. Four sites (✦) fall within a no-take zone (NTZ). Blank spaces signify gaps in data.

City / Area	Dive site	2018	2019	2020	2021	2022	2023	Total	Years
Cannes	Fouillée	18	12	12	12	6	6	66	6
Cap d'Antibes	Cap Gros	18	6	12			12	48	4
	Graillon		12	12	12		12	48	4
	La Boute	12	6	6			6	30	4
	La Lauve	12	6	6	6		6	36	5
	Le Grand Boule	18	12	12	6	12	12	72	6
	Péquerolle ✦	12	6	6			18	42	4
	Rascoui	12	12	12		6	18	60	5
	Seiche St Pierre	12	12	18			12	54	4
Esterel	Grand Cap Roux ✦	6	12	12			12	42	4
	La Vaquette	6	12	24			24	66	4
	Petit Cap Roux ✦	6		18	6		6	36	4
	Roche d'Aurelle ✦	18	12	12			12	54	4
Golfe-Juan	Fourmigue	12	12	24	12	12	6	78	6
Lérins	Bateguier	12	12	18		24	18	84	5
	Bilitis		12	12	12	6	18	60	5
	Petite Françoise	18	12	12	18	6	12	78	6
	St Féréol	12	12	6		12	12	54	5
	Tradelière	12	12	6	12	18	6	66	6
	Vengeur	12	6	18	30	12	6	84	6
Nice	Causinière			18	24	6		48	3
	Crau de Nao	6	12	6	6	6		36	5
	Grande Baie	6	12	6	6	6	6	42	6
	Grotte à Corail		12	6		6		24	3
	Le Lido			6		12		18	2
	Tombant Maeterlinck		6	12	6	6		30	4
		240	240	312	168	156	240	1356	

assemblages was then assigned to each site in consideration of the five calculated proxies. Following an initial analysis in fish assemblage cluster health, statistical analyses were conducted in RStudio (Version 2023.06.0+421) to further understand the differences in fish assemblages at the 26 sampling locations in this study (Fig. 1).

Univariate Kruskal-Wallis tests were performed on each calculated proxy by location (Cannes, Cap d'Antibes, Estérel, Golfe-Juan, Lérins Islands, and Nice). To understand the significance between locations, pairwise comparisons were performed using Dunn's test with Holm's stepwise adjustments. Dunn's tests are an appropriate procedure when Kruskal–Wallis tests are rejected (Dinno, 2015). A similar procedure was performed to assess if clear differences in fish assemblage health existed between dive sites situated inside and outside NTZs using one-way Mann-Whitney tests, also known as Wilcoxon Rank-Sum tests, for statistical significance. Family and species-specific analyses were also conducted; univariate one-way Mann-Whitney tests were realized to understand if the presence (represented as a percentage; the number of presences ÷ total number of observations × 100) of three fish families: Serranidae, Labridae, Sparidae and six fish species: common dentex (*D. dentex),* dusky grouper (*E. marginatus*), brown meagre (*S. umbra*)*,* comber (*S. cabrilla*)*,* barracuda (*Sphyraena* spp.)*,* and the salema porgy (*S. salpa*)*,* differed at dive sites located inside NTZs compared to those situated outside of NTZs.

Results

This research incorporated a total of 1356 fifteen-minute counts, corresponding to 452 cumulative hours (60-minute dive time) in the water, undertaken by 124 observers over the last six years between May to October (Table 3). This enormous effort between citizen scientists, authorities and members of the scientific community contributed to the ecological assessment of fish assemblages at 26 dive sites in Cannes, Antibes, Estérel, Golfe-Juan, the Lérins Islands and Nice (Table 4). Upon an initial analysis, there seemed to be a difference in ecological status of fish assemblages based on sampling location, notably with *excellent* and *good* health observed in Cannes, Estérel and Golfe-Juan, whilst *average* and *poor* health were observed in the Lérins Islands. No evident temporal trends were observed in analysis.

Locational Variation

Results indicated significant differences in both the CI (Fig. 2A; $H(5) = 29.498$, $p < 0.001$) and the MI values (Fig. 2B; $H(5) = 38.495$, $p < 0.001$), with the Lérins Islands showing significantly lower species density and size class occurrence when compared to the rest of the sampling locations. Results also indicated the greatest proportion of carnivorous individuals (CP) in Estérel (Fig. 2C; $H(5) = 27.277$, $p < 0.001$), while the proportion of large individuals (LP) was significantly lower in the Lérins Islands (Fig. 2D; $H(5) = 36.901$, $p < 0.001$). Lastly, results suggested significant differences in relative species richness, with the lowest diversity observed in the Lérins Islands (Fig. 2E; $H(5) = 32.971$, $p < 0.001$).

Table 4. Ecological status of fish assemblages at each site over time, using five cluster groups ranging from 1: "Excellent" to 5: "Very Poor" (Francour, 2017-Table 2). Four sites (✦) fall within a no-take zone (NTZ). Blank spaces signify gaps in data.

City / Area	Dive site	2018	2019	2020	2021	2022	2023
Cannes	Fouillée	1	2	2	2	1	2
Cap d'Antibes	Cap Gros	2	2	2			1
	Graillon		5	2	4		3
	La Boute	3	4	2			2
	La Lauve	2	2	1	2		1
	Le Grand Boule	3	3	3	2	3	3
	Péquerolle ✦	2	2	1			2
	Rascoui	1	2	1		4	3
	Seiche St Pierre	2	2	3			2
Estérel	Grand Cap Roux ✦	1	1	2			5
	La Vaquette	1	1	2			2
	Petit Cap Roux ✦	2		1	2		1
	Roche d'Aurelle ✦	1	1	1			1
Golfe-Juan	Fourmigue	2	2	2	2	2	2
Lérins	Bateguier	3	3	2		2	4
	Bilitis		3	3	4	3	3
	Petite Françoise	3	4	4	3	2	4
	St. Féréol	3	4	3		3	3
	Tradelière	2	2	3	3	1	1
	Vengeur	2	4	4	4	3	3
Nice	Causinière			2	4	1	
	Crau de Nao	2	2	3	2	2	
	Grande Baie	2	2	3	1	2	2
	Grotte à Corail		3	2		1	
	Le Lido			4		3	
	Tombant Maeterlinck		2	2	1	2	

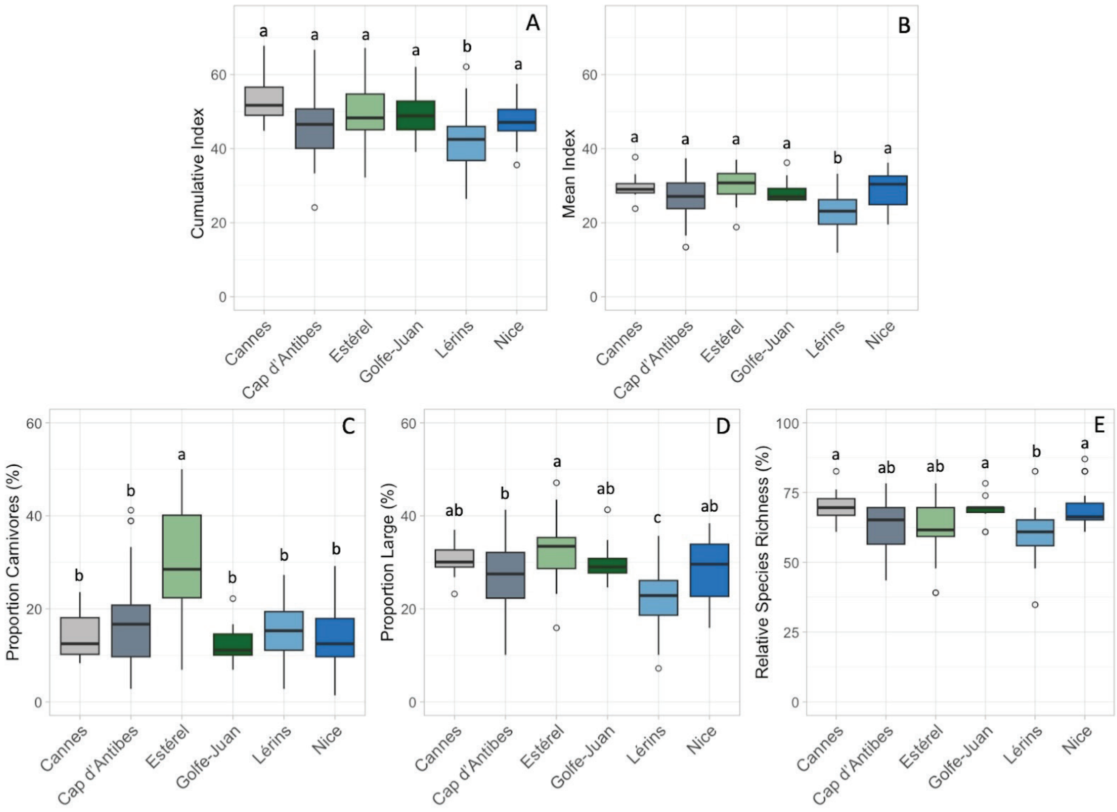

Fig. 2: Boxplots for the five FAST indices ((A) CI: cumulative index, (B) MI: mean index, (C) CP: proportion of carnivores, (D) LP: proportion of large individuals, (E) RSR: relative species richness) during the 2018-2023 summer seasons, segmented by location. Lowercase letters (a, b and c) indicate statistically significant differences between sites.

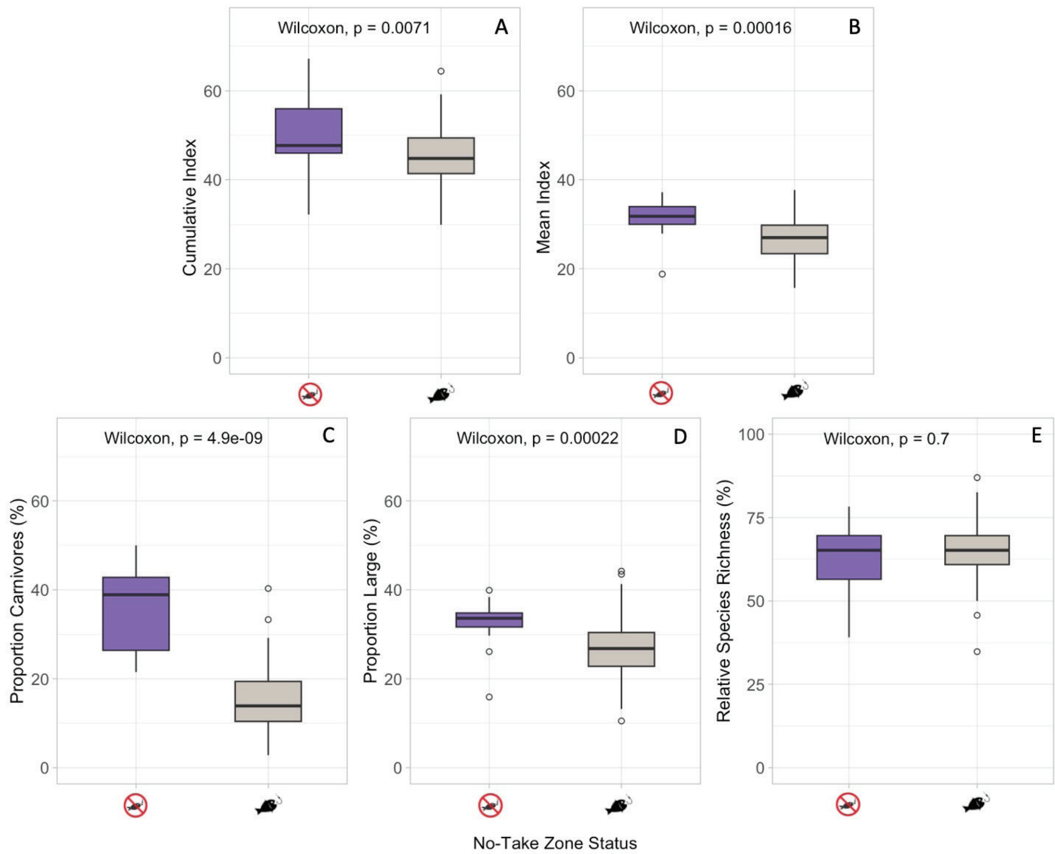

Fig. 3: Boxplots for the five FAST indices (cumulative index, mean index, proportion of carnivores, proportion of large individuals and relative species richness) during the 2018-2023 summer seasons, segmented by dive sites situated inside and outside NTZs. Sites inside NTZs are indicated by no fishing symbols and purple boxplots, while sites outside of NTZs are indicated by fishing symbols and gray boxplots.

No-Take-Zone Status

Greater CI (Fig. 3A; W = 1096.5, p < 0.01) and MI values (Fig. 3B; W = 1241.5, p < 0.001), proxies used for species density and size distribution, were observed at dive sites situated inside NTZ Additionally, results suggested a greater proportion of carnivores (Fig. 3C; W = 1510, p < 0.001) and large individuals (Fig. 3D; W = 1231.5, p < 0.001) at dive sites situated inside NTZs. No significant differences in relative species richness (RSR) were observed (Fig. 3E; W = 738.5, p = 0.70) between dive sites situated inside and outside NTZs.

Responses of Fish Families and Fish Species to NTZ status

NTZ status seemed to have both neutral and positive correlations with the presence of Labridae, Sparidae, and Serranidae fish families (Fig. 4). For both Labridae size classes (SM & L), we observed no significant differences; Labridae individuals were neither more present inside or outside NTZs. For both Sparidae size classes (SM & L), significant differences indicated that fishes were more present outside of NTZs than inside NTZs. In contrast, for both Serranidae size classes (SM & L), results indicated significant differences; Serranidae fishes were more present at dive sites situated inside NTZs.

NTZ status appeared to have a positive correlation with the presence of heavily targeted teleost fishes: *D. dentex, E. marginatus, S. umbra, S. cabrilla* and *Sphy-*

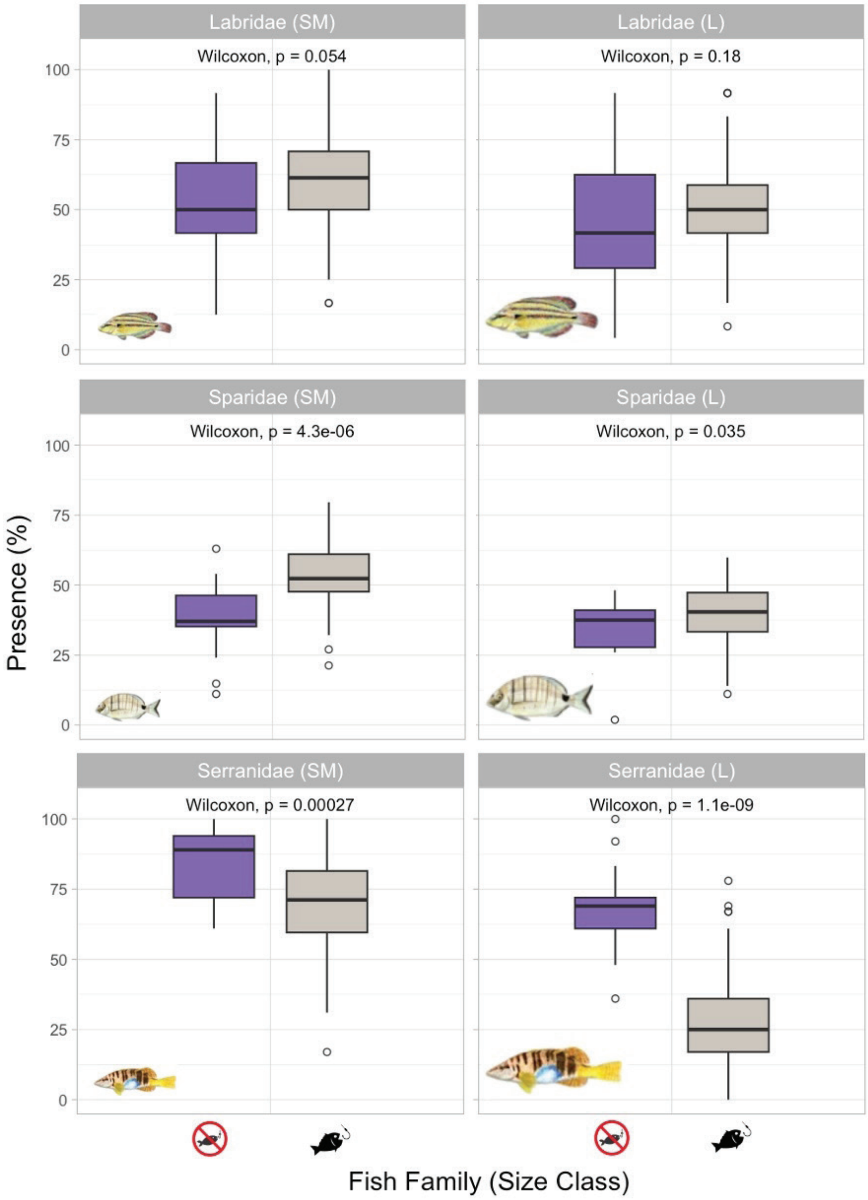

Fig. 4: Boxplots focused on the presence of three heavily targeted fish families by size class (SM–small or medium, L–large) and NTZ status. Sites inside NTZs are indicated by no fishing symbols and purple boxplots, while sites outside of NTZs are indicated by fishing symbols and gray boxplots. Families include: Labridae (top), represented by a rainbow wrasse (*S. tinca*), Sparidae (middle), represented by a common seabream (*D. sargus*) and Serranidae (bottom), represented by a painted comber (*S. scriba*).

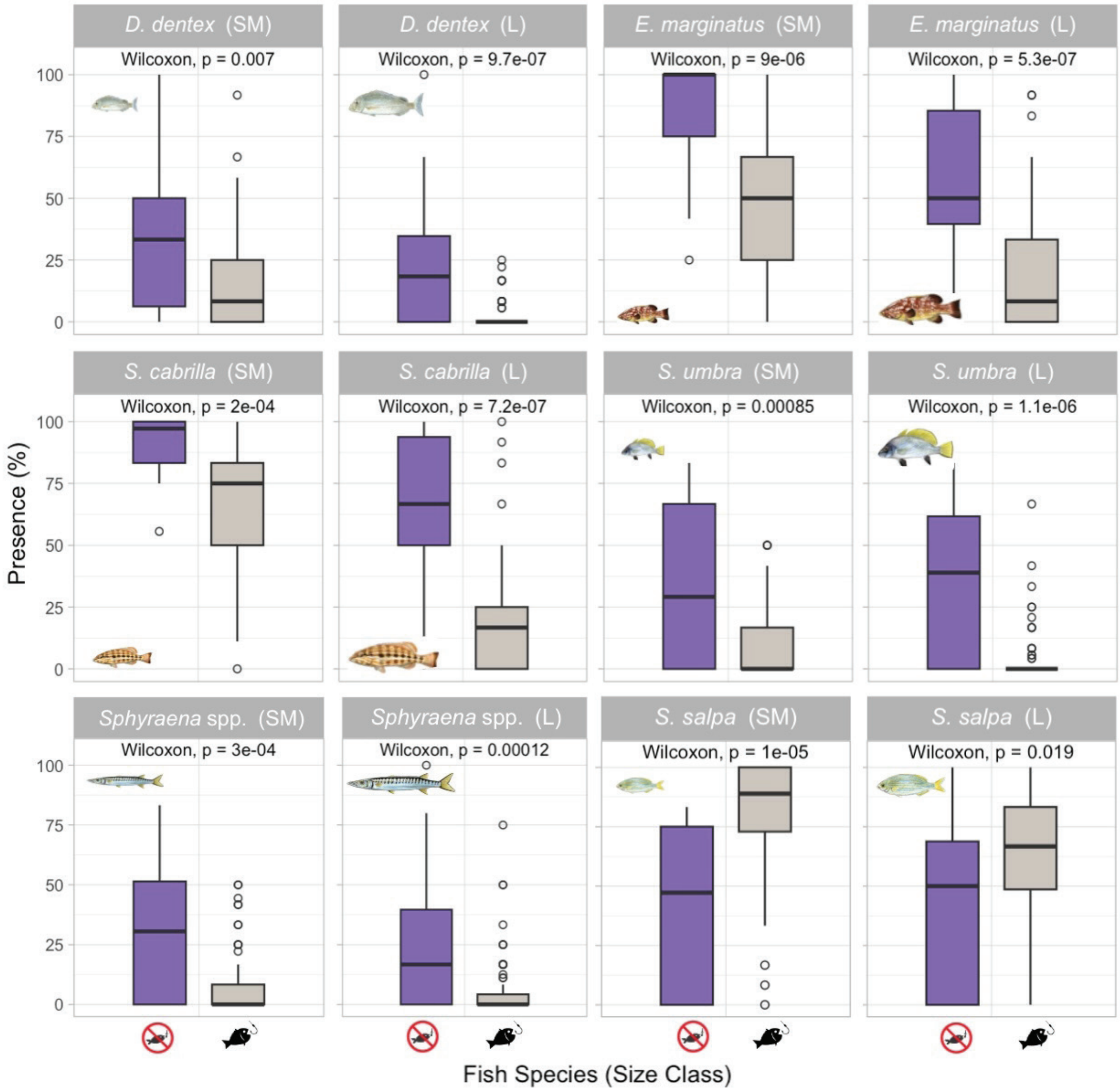

Fig. 5: Boxplots focused on the presence of six fish species by size class (SM–small or medium, L–large) and NTZ status. Sites inside NTZs are indicated by no fishing symbols and purple boxplots, while sites outside of NTZs are indicated by fishing symbols and gray boxplots. Fish species include: common dentex (*D. dentex* - top left), dusky grouper (*E. marginatus* - top right), comber (*S. cabrilla* - middle left), brown meagre (*S. umbra* - middle right), barracuda (*Sphyraena* spp. - bottom left) and the salema porgy (*S. salpa* - bottom right).

raena spp. (Fig. 5). Both size classes (SM & L) of protected species, *E. marginatus* and *S. umbra,* were more present inside NTZs. The same result was observed for *S. cabrilla,* and *D. dentex* individuals*,* who were more present inside NTZs, for both size classes (SM & L). Moreover, significant differences were observed for both size classes (SM & L) of *Sphyraena* spp. In contrast, results indicated that both size classes (SM & L) of herbivore *S. salpa* were more present outside of NTZs in our study region.

Discussion

Fishing Pressure – Lérins Islands

FAST observations confirmed locational variation and spatial dependency, with significantly lower fish densities and smaller fish at dive sites situated in the Lérins Islands (Batéguier, Bilitis, Petite Françoise, St. Feréol, Tradelière and Vengeur). For centuries, even before the establishment of the Cannes *prud'homie de pêche* in 1791, fishing has been a prolific activity along Cannes' coastlines. Today, roughly 30 small-scale fishers continue to deploy their nets along the rocky shores of Sainte-Marguerite, the larger of the two Lérins Islands (Rosati-Marzetti, 2023).

Despite the declining number of SSF in the Mediterranean (Prato *et al.*, 2016; Graner *et al.*, 2023), recreational fishing has increased in popularity along coastlines in the last few decades (Herfaut *et al.*, 2013; Font & Lloret, 2011, 2014). According to a technical report conducted by the *Conseil Scientifique des Iles de Lérins* (CSIL), leisure fishers in the Lérins Islands arrive in traditional *Pointus*, powerful motorboats or kayaks and deploy their fishing lines most commonly over *Posidonia oceanica* (Neptune grass) beds, targeting fish species inhabiting the pelagic zone (Pierre & Loqué, 2023). Even if the total catch of recreational fishing is low when compared to that of SSF,the catch of some target species is very high (e.g., *D. labrex*, *S. aurata* and *S. cantharus* – Herfaut *et al.*, 2013). Studies have also shown that recreational fishers are illegally landing immature fish below the minimum landing size (MLS), thus limiting reproductive potential (Font & Lloret, 2014). Additionally, removing large individuals by spearfishing may also negatively influence the reproductive potential of vulnerable fish populations (Prato *et al.*, 2013), as large females are proportionally more fecund (Font & Lloret, 2014).

Contrary to small scale fisheries, recreational fisheries in France are subject to limited regulation and enforcement (Herfaut *et al.*, 2013; Font & Lloret, 2014), likely contributing to *poor* fish assemblage health observed in the Lérins Islands, as well as other sites along the Cap d'Antibes (e.g., Graillon, Rascoui). To our knowledge, an assessment of recreational fishers' illegal landings below the MLS has not been conducted in the Côte d'Azur, highlighting an area for further research.

Other Anthropogenic Pressures – Lérins Islands

Poor fish assemblage health may also be a result of habitat destruction and degradation in the Lérins Islands, arising from urban development and high levels of tourism (e.g., pollution from sewage outlets, aquaculture farms and shipyards; yachting – Ville d'Antibes Juan-les-Pins, 2012). One of the most impacted habitats include *P. oceanica* meadows, a vital habitat for juvenile fish assemblages, associated with high biodiversity (Francour, 1997; Díaz-Gil *et al.*, 2019). In 1992, a collective 4.8 ha of *P. oceanica* meadows were destroyed, both directly and indirectly, by the laying of an electric cable and a water pipeline (Boudouresque *et al.*, 2009). To add to mounting pressure, it is estimated that the Lérins Islands welcomes more than 1,000 boats per day during the summer months, some greater than 30 m in length. Undeterred by regulations, many of these boats often anchor in areas rich in *P. oceanica,* scarring the seabed and uprooting seagrass shoots in the process. Due to slow growth (2-4 cm yr^{-1}), recolonization of *P. oceanica* is very low (Campagne *et al*., 2015). Therefore, resulting reductions of *P. oceanica* in the area are likely to have long-term impacts on biodiversity.

Given collective anthropogenic pressures, we emphasize the need to implement a NTZ in the Lérins Islands. To address *P. oceanica* degradation more specifically, the implementation of an NTZ could be complemented by the reinforcement and geographical extension of existing ecological mooring zones (*zones de mouillages et d'équipements légers* - ZMEL) and regulations that prohibit anchoring to various extents in different areas around the islands. Efforts to manage mounting pressures in the Lérins Islands would likely ameliorate *poor* fish assemblage health, while also ensuring the socio-economic well-being of fishers in the region.

Fishing Protection – NTZ Status

Fish assemblages benefit directly from protection provided by MPAs, especially those with NTZ status (Guidetti *et al.*, 2008; Stewart *et al.*, 2008; Seytre & Francour, 2008, 2009; Lester *et al.*, 2009; Appolloni, 2014; Sala & Giakoumi 2017; Sköld *et al.*, 2022; Davis & Harasti, 2023). This study found that sampling sites inside NTZs were characterized by high fish densities, larger fish, and a greater proportion of carnivorous individuals. This result was consistent with other studies by trained professionals that have reported positive effects on fish density, and size (Stewart *et al.*, 2008; Lester *et al.*, 2009; Appolloni, 2014; Sköld *et al.*, 2022). High LP and CP index values indicate that large carnivorous individuals occupying higher trophic levels have greater responses to NTZ status, likely because large predators are most often targeted by fisheries (Lester *et al.*, 2009). NTZ status can also have positive effects on surrounding fish assemblages via the spillover effect, or the overflow of fish biomass from protected zones to surrounding areas (Di Lorenzo *et al.*, 2020). This insight likely explains *excellent* fish assemblage health observed in La Vaquette, ~2 km outside the Cap-Roux NTZ, as well as *good* fish assemblage health observed in Fourmigue, ~1 km outside the Golfe-Juan NTZ (Table 4). On the contrary, this spillover phenomenon was not observed in La Boute, situated ~250 m outside of the Péquerolle NTZ, potentially explained by the youthfulness of the NTZ.

Despite overall positive trends inside and around NTZs, protected areas are still susceptible to changing environmental conditions that may have direct or indirect effects on fish assemblages. In August 2023, a mucilage event was recorded inside the Cap Roux NTZ. During which, the ecological status of fish assemblages was *very poor* (Table 4). Mucilage, originating from planktonic or benthic algal species, impacts benthic organisms such as crustaceans, molluscs, coralline algae, sea urchins, and gorgonians by reducing light penetration and causing hypoxic conditions (Giuliani *et al.*, 2005; Schiaparelli *et al.*, 2007; Piazzi *et al.*, 2018). It has also been linked to both direct and indirect effects on benthic and pelagic fish populations (Dalyan *et al.*, 2021; Karadurmuş & Sari, 2022).

Responses of Fish Families to NTZ Status

To further assess the benefits of NTZs, we explored the presence of three of the most important fish families to Mediterranean coastal zones: Labridae, Sparidae and Serranidae (Harmelin-Vivien, 2000) in response to fishing protection. In the Mediterranean, Labridae are primarily type 1 meso-carnivorous teleosts with a variable diet, feeding on annelids, amphipods, small crustaceans, echinoderms and molluscs (Bell & Harmelin-Vivien, 1983; Astruch *et al.*, 2018). In the past, an increase in Labridae abundance was observed in areas with heavy fishing activity, likely due to decreased top-down pressure by macro-carnivores (Harmelin-Vivien, 2000). However, our results indicated no difference in Labridae (*L. viridis, L.*

merula and *S. tinca*) presence for either the SM or L size classes (Fig. 4). Meanwhile, Sparidae individuals were more present outside of NTZs for both size classes (SM & L). Fishes of the Sparidae family include herbivorous *S. salpa,* but are mainly type 2 meso-carnivores (Bell & Harmelin-Vivien, 1983; Astruch *et al.*, 2018), consisting of several mobile species heavily targeted by recreational and small-scale fishers:: *P. pagrus, S. aurata, S. cantharus, L. mormyrus* and *Diplodus* spp. (Rocklin *et al.*, 2011; Marengo *et al.*, 2014; Kayal *et al.*, 2020). With increased Sparidae presence in zones with fishing pressure, this may begin to suggest trophic level imbalance. Contrary to Sparidae, both Serranidae size classes (SM & L) were more present inside NTZs. Serranidae (*E. marginatus, S. cabrilla* and *S. scriba*) are primarily macro-carnivorous feeding on decapod crustaceans, large amphipods, and teleosts fishes (Bell & Harmelin-Vivien, 1983; Astruch *et al.*, 2018). With increased macro-carnivore abundance, as seen in zones absent of fishing pressure, we expect to see a decrease in the abundance of meso-carnivores (e.g., Labridae or Spiridae) simply due to top-down pressure by macro-carnivores (Pauly *et al.*, 1998; Harmelin-Vivien, 2000; Lester *et al.*, 2009, Seytre & Francour, 2009).

Responses of Fish Species to NTZ Status

This study also looked at the effects of fishing protection on the presence/absence of six species with high ecological importance: four macro-carnivores (*D. dentex, Sphyraena* spp.*, E. marginatus, S. cabrilla),* one meso-carnivore (*S. umbra*) and one herbivore (*S. salpa*). All macro-carnivorous individuals were more present inside zones absent of fishing pressure (Fig. 5). It is important to first highlight *E. marginatus*, listed as vulnerable–VU as of the latest IUCN report (Pollard *et al.*, 2018). In France, *E. marginatus* has been protected from spearfishing since 1993 and SSFsince 2003 (Pollard *et al.*, 2018). Our results show a clear positive correlation between NTZ status and *E. marginatus* presence for both size classes (SM & L). This is an observation that is consistent with previous studies in Medes Islands MPA, Spain (Garcia-Rubias *et al.*, 2013), Port-Cros National Park, France (Harmelin & Ruitton, 2010), and Scandola Nature Reserve, Corsica (Cottalorda *et al.*, 2012). We did not observe these strong trends outside of NTZs.

NTZ status has also displayed similar correlations with other macro-carnivore populations (Cottalorda *et al.*, 2012). *D. dentex* is currently the only Sparidae species in the Mediterranean classified as vulnerable–VU by the IUCN (Carpenter & Russell, 2014). Contrary to *E. marginatus, D. dentex* is a highly mobile fish species, lacking protection in France and the Mediterranean. As a macro-carnivorous species at the top of the food chain, it serves a vital role as an indicator species for the structure and functioning of coastal ecosystems (Seytre & Francour, 2009; Marengo *et al.*, 2014). In this study, we observed clear positive correlations between the presence and size structure of *D. dentex* individuals and NTZ status, a common trend observed in many other MPAs (Cottalorda *et al.*, 2012; García-Rubies *et al.*, 2013). In order to reestablish *D. dentex* populations outside of NTZs, we suggest incorporating conservation management policy temporarily banning recreational and/or small-scale fishing, similar to that seen for *E. marginatus* and *S. umbra.*

Sciaena umbra is listed as near threatened–NT as of the latest IUCN report (Chao, 2020). Similar to research conducted previously (Garcia-Rubies *et al.*, 2013; Harmelin-Vivien *et al.*, 2015, Di Iorio *et al.*, 2020), NTZs in our study were positively correlated with the presence and size structure of *S. umbra* individuals. We did not observe the same trend outside of NTZs (Fig. 5), either suggesting the continuation of illegal leisure fishing, or that we have yet to see the full recovery of the species. A full recovery is highly dependent on the size and age of the species, and it may strictly depend on its home range and behavioral patterns (Garcia-Rubies *et al.*, 2013). For species with narrow home ranges, *S. umbra* and *E. marginatus*, it may take up to 50 years to reach carrying capacity and make a full recovery (Harmelin-Vivien *et al.*, 2015). This highlights the need for continued long-term assessments focused on monitoring their recovery in MPAs, to include zones where data is insufficient. The citizen science-based FAST protocol could be a useful and necessary tool used to achieve this.

Future FAST Perspectives

Citizen science has been applied to overcome difficulties associated with field research such as insufficient time, inadequate funding, and large geographical spatial scales that may be impractical without the help of a larger conservation-minded audience. Despite its positive potential, citizen science has been met with data quality concerns (Aceves-Bueno *et al.*, 2017), likely linked to citizen science data that was more variable when compared to data collected by scientists (Moyer-Horner *et al.*, 2012). Yet, applying citizen science to monitor fish assemblages using FAST was proven effective when compared to its application by trained professionals (Ben Lamine *et al.*, 2018), justifying its use in the study. In a similar manner, results from this research appeared robust enough to meet the same conclusions as those collected by scientists.

Nevertheless, we consider future adaptations that may improve the FAST protocol. With the expansion of species distribution ranges under climate change, it may be important to consider including the presence/absence of exotic species in assessment. One of the most common biological invasions is the Lessepsian migration, referring to the transfer of marine organisms from the Red Sea following the opening of the Suez Canal (Azzurro *et al.*, 2021). Two herbivorous fish species belonging to the Siganidae family, the Rabbitfish (*Siganus luridus*) and Marbled spinefoot (*Siganus rivulatus*) are native to the western Red Sea (Boris *et al.*, 2009; Tsirintanis *et al.*, 2022). Currently, both species inhabit much of the eastern Mediterranean as far west as Tunisia (Azzurro & Andaloro, 2005; Castriota & Andaloro, 2008) and are known to compete for habitat and food resources with native Mediterranean herbivores, such as *S. salpa* (Daniel *et al.*, 2009). In the summer of 2008, *S. luridus* was caught by small-scale fishers in Marseille, France (Daniel *et al.*, 2009), and most recently in February 2024 in our study region (CSIL, unpublished data). Other species that may fall under this category include: Bluespotted cornetfish (*Fistularia commersonii*), the Silver Cheeked Toadfish (*Lagocephalus sceleratus*), and the Lionfish (*Pterois miles*), all of which have an ecological and financial impact on native fish assemblages and associated small-scale fisheries (Coro *et al.*, 2018).

Aside from proposed adaptations to FAST, the protocol could also be complemented by incorporating numerical counts for threatened species with high ecological importance (e.g.,

S. umbra, E. marginatus, D. dentex, etc.). This would allow researchers to calculate species biomass, further assess trophic level interactions and evaluate the effectiveness of regional NTZs.

Acknowledgements

The authors would like to thank the late Patrice Francour for developing FAST and for supporting NaturDive in adapting the protocol for this ecological assessment. We would also like to thank the citizen scientists who contributed to data collection (i.e. members of NaturDive). In addition, we express our gratitude to the Université Côte d'Azur and MARRES master's program staff, as well as those who provided their advice and expertise in our manuscript preparation. Lastly, we appreciate those who helped finance this citizen science project: *L'Office Français de la Biodiversité* (OFB) and the *Métropole de Nice*.

References

Aceves-Bueno, E., Adeleye, A.S., Feraud, M., Huang, Y., Tao, M. *et al*., 2017. The Accuracy of Citizen Science Data: A Quantitative Review. *Bulletin of the Ecological Society of America,* 98 (4), 278-290.

Appolloni, L., 2014. Reserve effect on fish assemblages of a no-take zone in the Punta Campanella MPA (Gulf of Naples: Tyrrhenian Sea). In: *International Marine Conservation Congress (IMCC) Conference*, Glasgow, 15 August 2014. University of Naples, Italy.

Astruch, P., Boudouresque C.F., Rouanet, E., Le Direach, L., Bonhomme, P. *et al*., 2018. A quantitative and functional assessment of fish assemblages of the Port-Cros Archipelago (Port-Cros National Park, north-western Mediterranean Sea). *Scientific Reports of the Port-Cros National Park*, 32, 17-82.

Azzurro, E., Andaloro, F., 2005. A new settled population of the Lessepsian migrant *Siganus luridus* (Pisces: Siganidae) in Linosa Island - Sicily Strait. *Journal of the Marine Biological Association of the UK*, 84 (4), 819-821.

Azzurro, E., Nourigat, M., Cohn, F., Ben Souissi, J., Bernardi, G., 2021. Right out of the gate: The genomics of Lessepsian invaders in the vicinity of the Suez Canal. *Biological Invasions*, 24 (4), 1117-1130.

Bell, J.D., Harmelin-Vivien, M.L., 1983. Fish fauna of French Mediterranean Posidonia oceanica seagrass meadows. II: feeding habits. *Tethys,* 11 (1), 1-14.

Ben Lamine, E., 2017. *Fish and fishing monitoring methods and indicators in Marine Protected Areas in Tunisia.* Ph.D Thesis. Université Côte d'Azur/Institut national agronomique de Tunisie, France/Tunisia, 275 pp.

Ben Lamine, E., Di Franco, A., Romdhane, M.S., Francour, P., 2018. Can citizen science contribute to fish assemblages monitoring in understudied areas? The case study of Tunisian Marine Protected Areas. *Estuarine Coastal and Shelf Science*, 200, 420-427.

Boulanger, E., Loiseau, N., Valentini, A., Arnal, V., Boissery, P. *et al.*, 2021. Environmental DNA metabarcoding reveals and unpacks a biodiversity conservation paradox in Mediterranean marine reserves. *Proceedings of the Royal Society B, Biological Sciences*, 288 (1949).

Boudouresque, C.F., Bernard, G., Pergent, G., Shili, A., Verlaque, M., 2009. Regression of Mediterranean seagrasses caused by natural processes and anthropogenic disturbances and stress: a critical review. *Botanica Marina*, 52 (5), 395-418.

Boris, D., Serge, P., Charbonnel, E., Francour, P., Letourneur,Y., 2009. Lessepsian rabbitfish *Siganus luridus* reached the French Mediterranean coasts. *Cybium: International Journal of Ichthyology*, 33 (2), 163-164.

Bosso, L., Panzuto, R., Balestrieri, R., Smeraldo, S., Chiusano, M.L. *et al*., 2024. Integrating citizen science and Spatial Ecology to inform management and conservation of the Italian seahorses. *Ecological Informatics*, 79, 102402.

Campagne, C.S., Salles, J.M., Boissery, P., Deter J., 2015. The seagrass Posidonia oceanica : Ecosystem services identification and economic evaluation of goods and benefits. *Marine Pollution Bulletin* 97, 391-400.

Carpenter, K., Russell, B., 2014. *Dentex dentex. IUCN Red List of Threatened Species*. (Accessed on 25 October 2024).

Castejón-Silvo, I., Terrados, J., Morales-Nin, B., 2023. Citizen Science in the Study of Marine Biodiversity: The case of Iconic and Cryptic Syngnathids. *Thalassas: An International Journal of Marine Sciences*, 39 (2), 679-686.

Castriota, L., Andaloro, F., 2008. First record of the Lessepsian fish Siganus luridus (Osteichthyes: Siganidae) in the Tyrrhenian Sea. *Marine Biodiversity Records*, 1 (e11).

Cavallé, M., Said, A., Peri, I., Molina, M., 2020. *Social and economic aspects of Mediterranean small-scale fisheries: A snapshot of three fishing communities.* Low Impact Fishers of Europe/MAVA Foundation. Mainstreaming Small-scale Fisheries in the Mediterranean report, 76 pp.

Chao, L., 2020. *Sciaena umbra*. *IUCN Red List of Threatened Species*. (Accessed on 25 October 2024).

Čižmek, I.Z., Stipić, S., Čižmek, H., Mokos, M., 2020. Fast, Not Furious – Adaptation of the Species List and Fish Size Classes for Fish Assemblage Survey Technique (FAST) for the Adriatic Sea. *Croatian Journal of Fisheries*, *78* (4), 195-202.

Cottalorda, J.M., Dominici, J.M., Harmelin, J.G., Harmelin-Vivien, M., Louisy, P. *et al.*, 2012. *Etude et synthèse des* *principales données disponibles sur les espèces de « mérous » de la Réserve naturelle de Scandola et de ses environs immédiats*. Parc Naturel Régional de Corse, 48 pp.

Coro, G., Vilas, L.G., Magliozzi, C., Ellenbroek, A., Scarponi, P. *et al.*, 2018. Forecasting the ongoing invasion of *Lagocephalus sceleratus* in the Mediterranean Sea. *Ecological Modelling*, 371, 37-49.

Couvray, S., 2020. Six years of monitoring of fish assemblages in shallow bottoms around the Embiez Islands and Cap Sicié (French Mediterranean Sea). *Vie et Milieu*, 70.

Dalyan, C., Kesici, N.B., Yalgin, F., 2021. Preliminary study on cryptobenthic fish assemblages affected by the mucilage event in the northeastern Aegean Sea. *Journal of the Black Sea / Mediterranean Environment,* 27 (2), 202-213.

Daniel, B., Piro, S., Charbonnel, E., Francour, P., Letourneur, Y., 2009. Lessepsian rabbitfish *Siganus luridus* reached the French Mediterranean coasts. *Cybium*, 33 (2), 163-164.

Davis, T.R., Harasti, D., 2023. Forty years of no-take protection preserves local fish diversity in a small urban Marine Protected Area. *Coasts*, 3 (4), 401-413.

Di Iorio, L., Bonhomme, P., Michez, N., Ferrari, B., Gigou, A. *et al.*, 2020. Spatio-temporal surveys of the brown meagre *Sciaena umbra* using passive acoustics for management and conservation. *BioRxiv*, 131326.

Di Lorenzo, M., Guidetti, P., Di Franco. A., Calò A., Claudet, J., 2020. Assessing spillover from Marine Protected Areas and its drivers: a meta-analytical approach. *Fish and Fisheries*, 21 (5), 906-915.

Díaz-Gil, C., Grau, A., Grau, A.M., Palmer, M., Cabrera, R. *et al*., 2019. Changes in the juvenile fish assemblage of a Mediterranean shallow posidonia oceanica seagrass nursery area after Half Century. *Mediterranean Marine Science*, 20 (3), 603.

Dunkley, R.A., 2017. The role of citizen science in environmental education: A critical exploration of the environmental citizen science experience. p. 213-230. In: *Analyzing the role of citizen science in modern research*. Albini, E., Bardají, R., Claramunt, B., Jurado, E., Latrofa, F. *et al.* (Eds). IGI Global Publishing, Hershey.

Francour, P., 1997. Fish assemblages of Posidonia Oceanica Beds at port-cros (France, NW Mediterranean): Assessment of composition and long-term fluctuations by Visual Census. *Marine Ecology*, 18 (2), 157-173.

Francour, P., 2017. *Evaluation des peuplements de poissons au sein de l'observatoire de la biodiversité du Parc national de Port-Cros: Synthèse des données FAST*. Parc National de Port-Cros/ECOMERS, 23 pp.

Francour, P., Bodilis, P., Cottalorda, J.M., 2011. Évaluation des peuplements de poissons dans la Réserve Naturelle de *Scandola et à proximité de ses limites: Les Apports de la méthode FAST.* Parc Naturel Régional de Corse/ECOMERS, 21 pp.

Francour, P., Bodilis, P., Cottalorda, J.M., 2013. *Suivi des peuplements de poissons dans le cadre de l'Observatoire opérationnel de la Biodiversité du Parc national de Port-Cros.* Parc National de Port-Cros/ECOMERS, 41 pp.

Font, T., Lloret, J., 2011. Biological implications of recreational shore angling and harvest in a marine reserve: The case of Cape Creus. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 21 (2), 210-217.

Font, T., Lloret, J., 2014. Biological and ecological impacts derived from recreational fishing in Mediterranean coastal areas. *Reviews in Fisheries Science & Aquaculture*, 22 (1), 73-85.

Garcia-Rubies, A., Hereu, B., Zabala, M., 2013. Long-term recovery patterns and limited spillover of large predatory fish in a Mediterranean MPA. *PLoS One,* 8 (9), e73922.

Garcia-Soto, C., Seys, J.J.C., Zielinski, O., Busch, J.A., Luna, S.I. *et al*., 2021. Marine citizen science: Current state in Europe and new technological developments. *Frontiers in Marine Science*, 8 (621472).

Ghanem, R., Soussi, J.B., 2017. Fish Assemblages Survey Technique (FAST) in Tunisian marine waters (Central Mediterranean). *Journal of Ichthyology*, 57 (2), 313-320.

Giuliani, S., Virno Lamberti, C., Sonni, C., Pellegrini, D., 2005. Mucilage impact on gorgonians in the Tyrrhenian sea. *The Science of the Total Environment*, 353 (1-3), 340-349.

Graner, E., Dudouet, B., Buchholzer, H., Frangoudes, K., 2023. *Towards a new era of support for small-scale fisheries and aquaculture in Europe*. Too Big To Ignore Research Report, No R02/2023, 34 pp.

Grorud-Colvert, K., Sullivan-Stack, J., Roberts, C., Constant, V., Horta e Costa, B. *et al*., 2021. The MPA Guide: A framework to achieve global goals for the ocean. *Science*, 373 (6560), eabf0861.

Guidetti, P., Milazzo, M., Bussotti, S., Molinari, A., Murenu, M. *et al*., 2008. Italian marine reserve effectiveness: Does enforcement matter? *Biological Conservation*, 141 (3), 699-709.

Harmelin-Vivien, M.L., Francour, P., 1992. Trawling or visual censuses? Methodological bias in the assessment of fish populations in seagrass beds. *Marine Ecology*, 13 (1), 41-51.

Harmelin-Vivien, M.L., 2000. Fishing down the Mediterranean food webs: Influence of fishing on the trophic structure of fish assemblages in Mediterranean seagrass beds. p. 99. In: *CIESM Workshop Monographs, Kerkyra, 26-30 July 2000*. CIESM, Greece.

Harmelin, J.G., Ruitton, S., 2010. Statut du mérou brun (Epinephelus marginatus) dans le Parc national de Port-Cros (France, Méditerranée): État 2008 et évolution en 15 ans. *Scientific Reports of the Port-Cros National Park*, 24, 147-159.

Harmelin-Vivien, M.L., Cottalorda, J.M., Dominici, J.M., Harmelin, J.G., Le Diréach, L. *et al*., 2015. Effects of reserve protection level on the vulnerable fish species *Sciaena umbra* and implications for fishing management and policy. *Global Ecology and Conservation*, 3, 279-287.

Herfaut, J., Levrel, H., Thébaud, O., Véron, G., 2013. The nationwide assessment of marine recreational fishing: A French example. *Ocean & Coastal Management*, 78, 121-131.

Hussein, C., Verdoit-Jarraya, M., Pastor, J., Ibrahim, A., Saragoni, G. *et al*., 2011. Assessing the impact of artisanal and recreational fishing and protection on a white seabream (*Diplodus sargus sargus*) population in the north-western Mediterranean Sea using a simulation model. Part 1: Parameterization and simulations. *Fisheries Research*, 108 (1), 163-173.

Karadurmuş, U., Sari, M., 2022. Marine mucilage in the Sea of Marmara and its effects on the marine ecosystem: Mass Deaths. *Turkish Journal of Zoology*, 46 (1), 93-102.

Kayal, M., Cigala, M., Cambra, E., Soulat, N., Mercader, M. *et al*., 2020. Marine reserve benefits and recreational fishing yields: The winners and the losers. *PLoS ONE*, 15 (12), e0237685.

Kelly, R., Fleming, A., Pecl, G.T., von Gönner, J., Bonn, A., 2020. Citizen science and marine conservation: A global review. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 375 (1814), 20190461.

Krželj, M., Cerrano, C., Di Camillo, C., 2020. Marine citizen science and social platforms: The case of *Hermodice carunculata* (Annelida, Polycheta). *Diversity*, 12 (8), 311.

Lester, S.E., Halpern, B.S., Grorud-Colvert, K., Lubchenco, J., Ruttenberg, B.I. *et al*., 2009. Biological effects within no-take marine reserves: A global synthesis. *Marine Ecology Progress Series*, 384, 33-46.

Lloret, J., Biton-Porsmoguer, S., Carreño, A., Di Franco, A., Sahyoun, R. *et al*., 2020. Recreational and small-scale fisheries may pose a threat to vulnerable species in coastal and offshore waters of the western Mediterranean. *ICES Journal of Marine Science*, 77 (6), 2255-2264.

Marengo, M., Durieux, E.D.H., Marchand, B., Francour, P., 2014. A review of biology, fisheries and population structure of *Dentex dentex* (Sparidae). *Reviews in Fish Biology and Fisheries*, 24, 1065-1088.

Marengo, M., Iborra, L., Leduc, M., Lejeune, P., Boissery, P. *et al*., 2021. Assessing spatial and temporal trends in a Mediterranean fish assemblage structure. *Diversity,* 13 (8), 368.

Moyer-Horner, L., Smith, M.M., Belt, J., 2012. Citizen science and observer variability during American pika surveys. *Journal of Wildlife Management,* 76 (7), 1472-1479.

Nalmpanti, M., Chrysafi, A., Meeuwig, J.J., Tsikliras, A.C., 2023. Monitoring marine fishes using underwater video techniques in the Mediterranean Sea. *Reviews in Fish Biology and Fisheries*, 33 (4), 1291-1310.

Pais, M.P., Cabral, H.N., 2018. Effect of underwater visual survey methodology on bias and precision of fish counts: A simulation approach. *PeerJ*, 6, e5378.

Pauly, D.V., Christensen, V., Dalsgaard, J., Froese, R.M., Torres, F.C., 1998. Fishing down marine food webs. *Science*, 279, 860-863.

Penca, J., Said, A., 2023. Market initiatives of small-scale fisheries in the Mediterranean: Innovation in support of sustainable blue economy. p. 365-384. In: *Ocean Governance: Knowledge Systems, Policy Foundations and Thematic Analyses.* Partelow, S., Hadjimichael, M., Hornidge, A.K. (Eds.). Springer International Publishing, Cham.

Piazzi, L., Atzori, F., Cadoni, N., Cinti, M.F., Frau, F. *et al*., 2018. Benthic mucilage blooms threaten coralligenous reefs. *Marine Environmental Research*, 140, 145-151.

Pierre, S., Loqué, F., 2023. *Enquête sur la pratique de la pêche de loisir en mer: une approche scientifique et participative.* Conseil Scientifique des Iles de Lérins (CSIL), CSIL Report, 46 pp.

Pollard, D.P.A., Bertoncini, Á., Fennessy, S., Francour, P., Barreiros, J., 2018. *Epinephelus marginatus.* IUCN Red List of Threatened Species. (Accessed on 25 October 2024)

Pörtner, H.O., Schulte, P.M., Wood, C.M., Schiemer, F., 2010. Niche dimensions in fishes: An integrative view. *Physiological and Biochemical Zoology*, 83 (5), 808-826.

Prato, G., Guidetti, P., Bartolini, F., Mangialajo, L., Francour, P., 2013. The importance of high-level predators in Marine Protected Area management: Consequences of their decline and their potential recovery in the Mediterranean context. *Advances in Oceanography and Limnology*, 4 (2), 176-193.

Prato, G., Barrier, C., Francour, P., Cappanera, V., Markantonatou, V. *et al*., 2016. Assessing interacting impacts of artisanal and recreational fisheries in a small Marine Protected Area (Portofino, NW Mediterranean Sea). *Ecosphere*, 7 (12), e01601.

Rey, A., Viard, F., Lizé, A., Corre, E., Valentini, A. *et al*., 2023. Coastal rocky reef fish monitoring in the context of the Marine Strategy Framework Directive: Environmental DNA metabarcoding Complements Underwater Visual Census. *Ocean & Coastal Management*, 241, 106625.

Rocklin, D., Tomasini, J.A., Culioli, J.M., Pelletier, D., Mouillot, D., 2011. Spearfishing regulation benefits artisanal fisheries: The ReGS indicator and its application to a multiple-use Mediterranean Marine Protected Area. *PLoS ONE*, 6 (9), e23820.

Rosati-Marzetti, C., 2023. L'évolution des cabanes de pêcheurs de l'île Sainte-Marguerite (Cannes). *Habiter*, 1.

Sala, E., Giakoumi, S., 2017. No-take marine reserves are the most effective protected areas in the ocean. *ICES Journal of Marine Science*, 75 (3), 1166-1168.

Schiaparelli, S., Castellano, M., Povero, P., Sartoni, G., Cattaneo-Vietti, R., 2007. A benthic mucilage event in North-Western Mediterranean Sea and its possible relationships with the summer 2003 European heatwave: Short term effects on littoral rocky assemblages. *Marine Ecology*, 28 (3), 341-353.

Seytre, C., Francour, P., 2008. Is the Cape Roux Marine Protected Area (Saint-Raphaël, Mediterranean Sea) an efficient tool to sustain artisanal fisheries ? First indications from visual censuses and trammel net sampling. *Aquatic Living Resources*, 21, 297-305.

Seytre, C., Francour, P., 2009. The Cap Roux MPA (Saint-Raphaël, French Mediterranean): Changes in fish assemblages within four years of protection. *ICES Journal of Marine Science*, 66, 180-187.

Seytre, C., Francour, P., 2014. A long-term survey of Posidonia oceanica fish assemblages in a Mediterranean Marine Protected Area: Emphasis on stability and no-take area effectiveness. *Marine and Freshwater Research*, 65, 244-254.

Sköld, M., Börjesson, P., Wennhage, H., Hjelm, J., Lövgren, J. *et al*., 2022. A no-take zone and partially protected areas are not enough to save the Kattegat cod, but enhance biomass and abundance of the local fish assemblage. *ICES Journal of Marine Science*, 79 (8), 2231-2246.

Stewart, G., Côté, I.M., Kaiser, M., Halpern, B., Lester, S.E. *et al*., 2008. *Are Marine Protected Areas effective tools for sustainable fisheries management? I. Biodiversity impacts of marine reserves in the temperate zone.* Collaboration for Environmental Evidence (CEE), CEE Systematic Review, No. 23.

Tsirintanis, K., Azzurro, E., Crocetta, F., Dimiza, M., Froglia C. *et al.*, 2022. Bioinvasion impacts on biodiversity, ecosystem services, and human health in the Mediterranean Sea. *Aquatic Invasions*, 17 (3), 308-352.

Ville d'Antibes Juan-les-Pins, 2012. Document d'objectifs du site Natura 2000 FR 9301573 "Baie et Cap d'Antibes – Iles de Lérins" - Tome 1 "Fiches descriptives des habitats et espèces marins d'intérêt communautaire". Document final (Décembre 2013), 96 pp.

Villéger, S., Brosse, S., Mouchet, M., Mouillot, D., Vanni, M.J., 2017. Functional ecology of fish: Current approaches and future challenges. *Aquatic Sciences*, 79 (4), 783-801.

Appendix

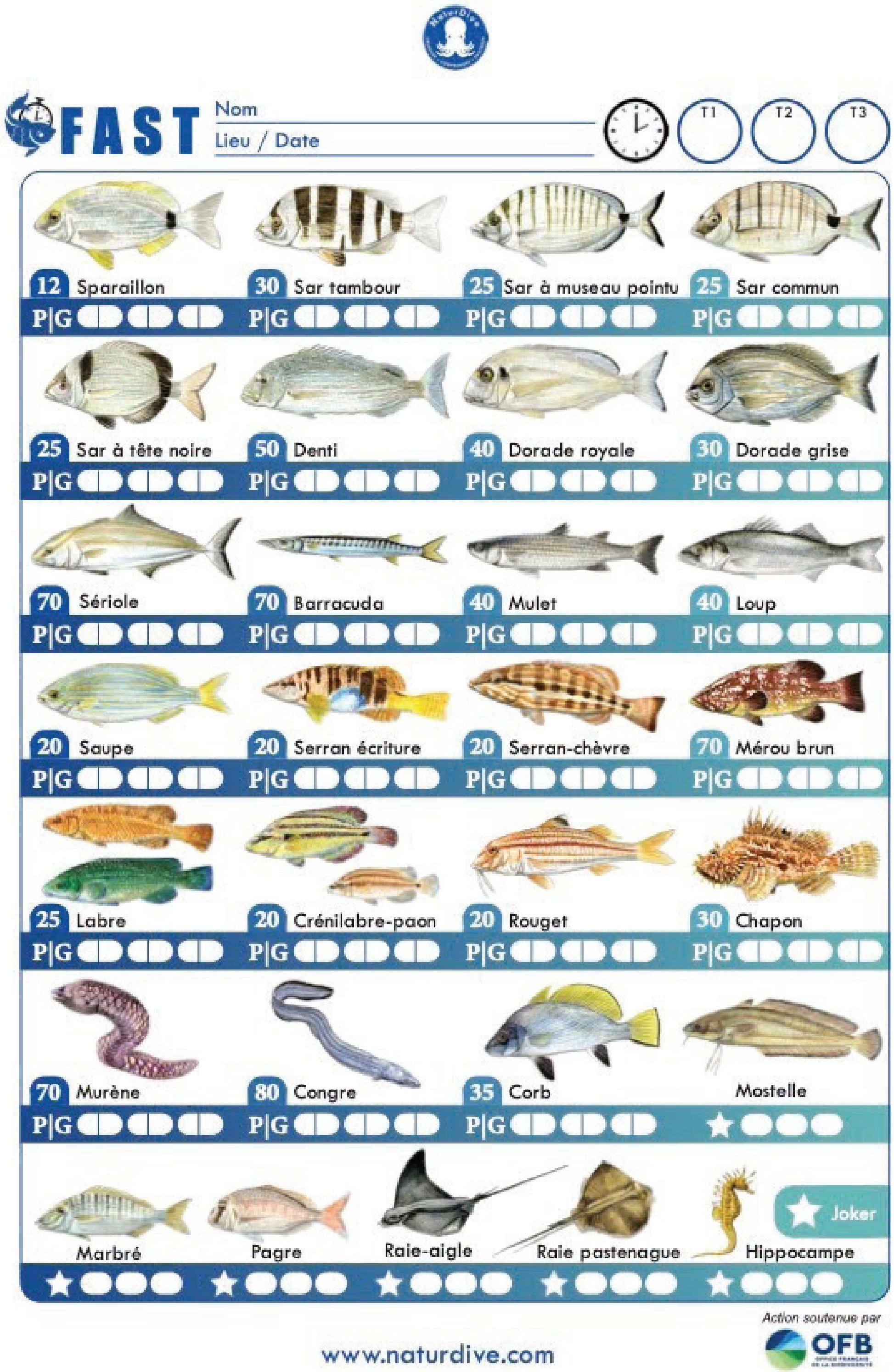

Fig. App.1: Underwater diving slate, consisting of 23 main species and six joker or rare species (★), used during FAST data collection.

Table. App.1. Results of the five calculated FAST indices, mean index (MI), cumulative index (CI), relative species richness (RSR), proportion of carnivores (CP) and proportion of large individuals (LP), at each dive site each year. We also include the coefficient of variance (CV), standard deviation (STDEV), and cluster health, indicating the ecological status of fish assemblages from 1 to 5 (Table 2). If CV was inferior to 30%, this indicated a stable fish assemblage. If the CV was superior to 30% (highlighted in blue), this indicated a large variation between replicate counts, between observers, or a degradation in fish population structure. Four sites (✦) fall within a no-take zone (NTZ).

Location	Dive Site	Year	MI	CI	RSR	CP	LP	CV	STDEV	Cluster
Cannes	Fouillée	2018	29.2	57.1	75.4	18.1	30.2	25.5	7.6	1
Cap d'Antibes	Cap Gros	2018	29.2	48.7	50.0	18.1	29.0	15.9	4.7	2
Cap d'Antibes	La Boute	2018	24.0	39.1	63.0	9.0	22.5	22.2	5.3	3
Cap d'Antibes	La Lauve	2018	28.4	50.0	65.2	20.8	29.0	19.2	5.5	2
Cap d'Antibes	Le Grand Boule	2018	23.8	42.1	60.9	9.7	22.9	22.5	5.3	3
Cap d'Antibes	Péquerolle ✦	2018	27.9	42.0	52.2	27.8	26.1	27.6	7.7	2
Cap d'Antibes	Rascoui	2018	30.7	50.6	67.4	18.8	30.8	18.2	5.5	1
Cap d'Antibes	Seiche St Pierre	2018	28.4	43.7	63.0	20.8	26.1	15.6	4.5	2
Estérel	Grand Cap Roux ✦	2018	34.3	58.6	65.2	50.0	34.8	24.8	8.5	1
Estérel	La Vaquette	2018	33.5	51.7	60.9	27.8	43.5	15.8	5.3	1
Estérel	Petit Cap Roux ✦	2018	31.8	46.0	56.5	40.3	33.3	9.3	3.0	2
Estérel	Roche d'Aurelle ✦	2018	31.4	52.9	73.9	22.7	33.8	19.2	6.1	1
Golfe-Juan	Fourmigue	2018	30.1	47.1	60.9	5.6	31.9	13.5	4.1	2
Lérins	Bateguier	2018	22.4	49.4	67.4	13.9	18.5	19.3	4.4	3
Lérins	Petite Françoise	2018	21.7	43.3	59.4	20.4	19.6	27.2	5.7	3
Lérins	St Féréol	2018	22.6	40.2	58.7	17.4	23.6	24.4	5.5	3
Lérins	Tradelière	2018	26.4	44.3	60.9	17.4	26.4	20.3	5.2	2
Lérins	Vengeur	2018	25.1	41.4	65.2	18.1	25.7	15.3	3.8	2
Nice	Crau de Nao	2018	29.7	50.6	78.3	12.5	29.0	23.5	7.0	2
Nice	Grande Baie	2018	30.3	52.9	69.6	15.3	30.4	21.0	6.4	2
Cannes	Fouillée	2019	23.8	50.0	76.1	10.4	23.2	24.5	5.7	2
Cap d'Antibes	Cap Gros	2019	27.0	48.3	65.2	23.6	26.1	24.5	6.6	2
Cap d'Antibes	Graillon	2019	15.7	29.9	52.2	4.2	10.5	23.8	3.8	5
Cap d'Antibes	La Boute	2019	19.9	34.5	56.5	2.8	20.3	15.8	3.1	4
Cap d'Antibes	La Lauve	2019	31.0	48.3	65.2	19.4	32.6	14.1	4.4	2
Cap d'Antibes	Le Grand Boule	2019	26.1	42.5	58.7	11.8	26.8	17.6	4.6	3
Cap d'Antibes	Péquerolle ✦	2019	28.4	47.1	56.5	22.2	29.7	21.4	6.1	2
Cap d'Antibes	Rascoui	2019	26.9	43.1	67.4	12.5	28.3	14.1	3.8	2
Cap d'Antibes	Seiche St Pierre	2019	26.8	46	56.5	16.7	31.9	15.3	4.1	2
Estérel	Grand Cap Roux ✦	2019	33.9	67.2	78.3	39.6	33.0	30.6	10.3	1
Estérel	La Vaquette	2019	34.0	49.4	60.9	28.5	44.2	11.5	3.8	1
Estérel	Roche d'Aurelle ✦	2019	30.1	46.0	65.2	21.5	34.8	18.5	5.5	1
Golfe-Juan	Fourmigue	2019	29.4	52.3	69.6	16.0	29.7	21.6	6.3	2
Lérins	Bateguier	2019	24.2	41.4	63.0	9.0	23.6	14.2	3.5	3
Lérins	Bilitis	2019	23.0	44.8	65.2	14.6	21.7	34.8	7.4	3
Lérins	Petite Françoise	2019	17.6	37.9	56.5	12.5	15.2	24.3	4.2	4
Lérins	St Féréol	2019	18.7	35.6	58.7	13.2	15.2	27.5	5.4	4
Lérins	Tradelière	2019	28.5	47.1	67.4	22.9	27.9	13.5	3.8	2
Lérins	Vengeur	2019	17.2	37.9	60.9	6.9	15.2	21.5	3.7	4
Nice	Crau de Nao	2019	28.9	45.4	65.2	14.6	27.5	7.8	2.2	2

Continued

Table. App.1. coninued

Location	Dive Site	Year	MI	CI	RSR	CP	LP	CV	STDEV	Cluster
Nice	Grande Baie	2019	29.4	44.8	65.2	6.9	28.3	14.4	4.2	3
Nice	Grotte à Corail	2019	27.6	43.1	63.0	6.9	26.4	13.5	3.8	3
Nice	Tombant Maeterlinck	2019	33.5	44.8	60.9	29.2	38.4	6.7	2.2	2
Cannes	Fouillée	2020	33.1	57.5	67.4	13.9	37.0	34.2	10.8	2
Cap d'Antibes	Cap Gros	2020	29.8	52.9	67.4	19.4	29.0	29.1	8.7	2
Cap d'Antibes	Graillon	2020	28.0	51.7	69.6	17.4	28.6	25.5	8.0	2
Cap d'Antibes	La Boute	2020	24.5	42.5	69.6	6.9	24.6	25.1	6.2	2
Cap d'Antibes	La Lauve	2020	37.0	64.4	73.9	33.3	41.3	15.5	5.7	1
Cap d'Antibes	Le Grand Boule	2020	23.6	44.8	69.6	13.9	19.9	30.2	7.1	3
Cap d'Antibes	Péquerolle ✦	2020	37.2	55.2	65.2	38.9	39.9	17.0	6.3	1
Cap d'Antibes	Rascoui	2020	30.8	59.2	69.6	20.8	33.3	23.5	6.7	1
Cap d'Antibes	Seiche St Pierre	2020	23.4	41.4	56.5	9.7	22.5	20.6	4.8	3
Estérel	Grand Cap Roux ✦	2020	26.3	40.2	45.7	40.3	24.6	19.4	5.2	2
Estérel	La Vaquette	2020	26.5	47.1	65.2	7.6	28.3	15.5	4.1	2
Estérel	Petit Cap Roux ✦	2020	34.0	56.7	62.3	44.4	33.6	22.3	7.6	1
Estérel	Roche d'Aurelle ✦	2020	30.4	60.9	76.1	29.2	35.5	28.0	8.5	1
Golfe-Juan	Fourmigue	2020	29.1	49.1	68.5	15.6	32.2	21.3	6.2	2
Lérins	Bateguier	2020	23.3	43.7	63.0	16.0	23.2	27.6	6.2	2
Lérins	Bilitis	2020	23.6	42.5	54.3	18.1	26.4	27.9	6.6	3
Lérins	Petite Françoise	2020	18.5	36.8	50.0	15.3	18.8	26.8	4.8	4
Lérins	St Féréol	2020	19.2	48.3	69.6	22.2	20.3	17.3	3.3	3
Lérins	Tradelière	2020	21.7	37.9	52.2	12.2	22.5	17.9	6.1	3
Lérins	Vengeur	2020	16.3	36.8	56.5	10.2	15.5	34.4	5.3	4
Nice	Causinière	2020	27.1	50.6	71.0	19.9	26.3	22.9	6.3	2
Nice	Crau de Nao	2020	24.7	44.8	73.9	4.2	22.5	24.4	6.0	3
Nice	Grande Baie	2020	30.8	47.1	65.2	13.9	33.3	23.3	7.2	2
Nice	Grotte à Corail	2020	30.1	44.8	65.2	9.7	32.6	21.1	6.4	2
Nice	Le Lido	2020	19.5	40.2	65.2	13.9	18.8	27.1	5.3	4
Nice	Tombant Maeterlinck	2020	30.7	48.3	65.2	25.0	32.2	14.6	4.5	2
Cannes	Fouillée	2021	28.4	48.9	69.6	10.4	30.4	18.1	5.1	2
Cap d'Antibes	Graillon	2021	16.5	35.6	56.5	6.9	18.1	22.6	3.6	4
Cap d'Antibes	La Lauve	2021	27.2	52.9	73.9	20.8	29.7	20.3	5.5	2
Cap d'Antibes	Le Grand Boule	2021	25.7	46.0	65.2	15.3	24.6	22.1	5.7	2
Estérel	Petit Cap Roux ✦	2021	29.9	44.8	47.8	47.2	31.2	42.6	12.7	2
Golfe-Juan	Fourmigue	2021	26.9	49.4	71.0	10.6	26.8	20.7	5.6	2
Lérins	Bilitis	2021	21.5	43.1	60.9	11.1	19.9	21.4	4.6	4
Lérins	Petite Françoise	2021	25.9	40.5	57.6	18.4	24.8	25	6.3	3
Lérins	Tradelière	2021	22.8	46.0	63.0	11.1	22.8	20.9	4.9	3
Lérins	Vengeur	2021	20.3	35.2	55.7	10.6	19.7	20.0	3.9	4
Nice	Causinière	2021	16.5	36.5	52.2	11.5	13.2	37.8	6.3	4
Nice	Crau de Nao	2021	33.0	44.8	69.6	9.7	34.1	17.7	5.8	2
Nice	Grande Baie	2021	24	46	69.6	9	23.2	27.7	6.5	2
Nice	Tombant Maeterlinck	2021	33.9	52.9	69.6	23.6	33.3	15.3	5.2	1

Continued

Table. App.1. coninued

Location	Dive Site	Year	MI	CI	RSR	CP	LP	CV	STDEV	Cluster
Cannes	Fouillée	2022	37.7	56.3	65.2	23.6	37.0	13.3	5.0	1
Cap d'Antibes	Le Grand Boule	2022	25.3	39.7	52.2	13.2	27.9	16.7	4.3	3
Cap d'Antibes	Rascoui	2022	20.3	34.5	56.5	2.8	20.3	10.5	2.1	4
Golfe-Juan	Fourmigue	2022	28.1	46.6	65.2	11.1	29.7	27.2	7.7	2
Lérins	Bateguier	2022	25.7	43.3	66.7	21.3	27.5	17.6	4.6	2
Lérins	Bilitis	2022	28.0	46.0	56.5	19.4	25.4	28.2	7.9	3
Lérins	Petite Françoise	2022	27.1	44.3	60.9	19.4	26.1	21.4	5.8	2
Lérins	St Féréol	2022	21.0	35.6	60.9	9.7	28.3	15.6	2.8	3
Lérins	Tradelière	2022	33.2	50.6	62.3	27.3	35.7	16.6	5.5	1
Lérins	Vengeur	2022	24.7	48.3	69.6	17.4	25.0	23.5	5.4	3
Nice	Causinière	2022	31.2	51.7	73.9	20.8	37.0	12.6	3.9	1
Nice	Crau de Nao	2022	28.6	47.7	67.4	9.7	29.5	13.4	3.8	2
Nice	Grande Baie	2022	30.8	47.1	73.9	9.7	29.7	11.2	3.4	2
Nice	Grotte à Corail	2022	36.2	55.2	82.6	12.5	34.8	13.7	5.0	1
Nice	Le Lido	2022	27.4	39.1	65.2	6.9	27.5	11.2	3.1	3
Nice	Tombant Maeterlinck	2022	35.1	44.8	60.9	29.2	34.1	15.9	5.6	2
Cannes	Fouillée	2023	28.7	44.8	60.9	8.3	26.8	15.0	4.3	2
Cap d'Antibes	Cap Gros	2023	32.0	51.1	71.7	25.0	32.2	12.6	4.0	1
Cap d'Antibes	Graillon	2023	25	41.4	67.4	4.9	25	21.2	5.3	3
Cap d'Antibes	La Boute	2023	30.7	47.1	69.6	11.1	33.3	18.2	5.6	2
Cap d'Antibes	La Lauve	2023	34.3	48.3	65.2	22.2	37.0	13.6	4.7	1
Cap d'Antibes	Le Grand Boule	2023	22.7	41.0	71.0	12.0	19.3	25.5	5.6	3
Cap d'Antibes	Péquerolle ✦	2023	32.1	47.5	56.5	41.2	32.1	15.1	4.8	2
Cap d'Antibes	Rascoui	2023	28.6	44.3	67.4	10.4	29.3	22.7	6.5	3
Cap d'Antibes	Seiche St Pierre	2023	28.8	54.0	69.6	14.6	29.7	16.1	4.6	2
Estérel	Grand Cap Roux ✦	2023	18.8	32.2	39.1	34.7	15.9	14.3	2.7	5
Estérel	La Vaquette	2023	26.4	44.8	63.0	14.6	26.8	26.7	7.0	2
Estérel	Petit Cap Roux ✦	2023	34.9	55.2	69.6	50.0	38.4	27.4	9.6	1
Estérel	Roche d'Aurelle ✦	2023	32.4	47.7	69.6	25.0	33.7	16.6	5.3	1
Golfe-Juan	Fourmigue	2023	32.8	49.4	69.6	12.5	34.8	24.5	8.0	2
Lérins	Bateguier	2023	17.1	33.9	54.3	13.9	13.4	21.4	3.7	4
Lérins	Bilitis	2023	24.9	36.8	53.6	13.4	27.3	15.6	3.8	3
Lérins	Petite Françoise	2023	18.2	35.6	52.2	12.5	15.9	32.8	6.0	4
Lérins	St Féréol	2023	21.9	34.5	52.2	14.6	22.1	32.1	6.9	3
Lérins	Tradelière	2023	32.4	47.1	60.9	23.6	33.3	13.4	4.3	1
Lérins	Vengeur	2023	27.4	47.1	34.8	13.9	24.6	16.4	4.5	3
Nice	Grande Baie	2023	27.8	57.5	87.0	13.9	23.2	23.5	6.5	2