Small coastal marine protected areas offer recurring, seasonal protection to the common stingray (*Dasyatis pastinaca*)

S. Kraft [a,*,1], A.C. Winkler [a,b], D. Abecasis [a]

[a] *Centro de Ciências do Mar (CCMAR), Universidade do Algarve, Faro, Portugal*
[b] *Department of Ichthyology and Fisheries Science, Rhodes University, Makhanda, South Africa*

A R T I C L E   I N F O

A B S T R A C T

Marine protected areas (MPAs) are a crucial tool in safeguarding marine biodiversity. However, elasmobranchs are often not the primary protection target of MPAs, and their contribution to protect these species remains to be better understood. In this study we examine the movement patterns of common stingrays in the Professor Luiz Saldanha marine park, a Portuguese temperate coastal MPA. Using acoustic telemetry, we tagged 31 common stingrays and analyzed their spatial and temporal distribution within the MPA and adjacent areas using a long-term data set. Our findings indicate that this species exhibits seasonal site fidelity, with greater presence during the colder months and reduced presence during warmer months. Space use areas did not exceed the size of the fully protected area, and nocturnal and crepuscular activity was significantly higher than during daytime. Additionally, we observed that most individuals seasonally migrated between this MPA and the nearby Sado estuary, likely to reproduce in the latter. These results demonstrate the site fidelity of common stingrays to an area within the marine park, however the protection provided is only seasonal. Seasonal protection of the movement corridor between the marine park and the estuary would improve the management of this species.

1. Introduction

Elasmobranchs (sharks, skates and rays) are among the most susceptible groups to overfishing, due to the long-lived life strategies of most species. These are characterized by slow growth, late maturity and low fecundity (Musick, 1999), and given their inherent vulnerability to overfishing, it is regarded as the main reason for their population declines and the major threat these animals are facing (Dulvy et al., 2021). Among the strategies implemented to counter the consequences of overfishing is the establishment of marine protected areas (MPAs) (Gell and Roberts, 2003). MPAs are described as "clearly defined geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ecosystem services and cultural values" (Dudley, 2008). MPAs have positively impacted fisheries and several other areas, including human well-being, economics, knowledge development, and conservation and management actions of marine species and habitats (Ban et al., 2019; Lester et al., 2009). However, their implementation can be complex, and their contribution to the protection of elasmobranchs is not as explored and well understood compared to other groups like bony fishes. Although evidence of swift MPA-induced improvements for elasmobranchs exists (Le Port et al., 2012; Speed et al., 2018), the magnitude of this effect is variable and often considered moderate, at least when isolated MPAs are implemented (Dwyer et al., 2020; MacKeracher et al., 2019). Protecting movement corridors and creating networks of connected MPAs have also been mentioned as effective steps to improve the protection of elasmobranchs (Chapman et al., 2005; Lédée et al., 2015). Most MPAs might not properly encompass their movements, because many MPAs are not established considering elasmobranchs as one of their conservation targets and these are highly mobile species that often undergo long migrations (Dwyer et al., 2020; MacKeracher et al., 2019). Therefore, to evaluate and eventually improve the protection given by established MPAs to elasmobranchs, a better understanding of their movement patterns is needed (Davidson and Dulvy, 2017; MacKeracher et al., 2019).

Acoustic telemetry is a well-suited method to study the movements of marine animals in an area of interest, like an MPA, and evaluate its protection efficiency (Abecasis et al., 2014a). It allows to monitor the long-term movements of multiple individuals in a continuous and mostly automated way, and collect large amounts of data (Hussey et al.,

---

2015). This way, acoustic telemetry can improve our understanding of animal movement patterns, which are important sources of information to evaluate how protection can change over time (Heupel and Simpfendorfer, 2005) and evaluate and optimize protected areas (Chevis et al., 2017). Nonetheless, there is a general scarcity in studies on the spatial ecology of batoids (Myliobatiformes, Rajiformes, Rhinopristiformes and Torpediniformes) (Matley et al., 2021).

The Professor Luís Saldanha Marine Park (LSMP), part of the Arrábida Natural Park, is a 53 km^2 coastal MPA off the Setúbal peninsula designated in 1998 and completely established by 2009 (Portuguese legislation, Council of Ministers Resolution 141/2005). The geographic area where this MPA is established is important because of its high biodiversity (Cunha et al., 2014; Gonçalves et al., 2002; Henriques et al., 2009), its role as a transition area between colder water species of higher latitudes and warmer water species of lower latitudes (Henriques et al., 2007), and because of a nearby upwelling hotspot (Fiúza, 1983). After its establishment, the LSMP's spatial protection was explored for commercially important species of fish (Abecasis et al., 2014b, 2015) and cuttlefish (Abecasis et al., 2013a) using acoustic telemetry. Additionally, some batoids have also been studied (Cabral, 2014; Sousa et al., 2019a). However, because of the low number of individuals in these studies and their short duration, the benefits this marine park is providing to them remains to be further evaluated.

The common stingray (*Dasyatis pastinaca*) is a medium-sized batoid found in the northeastern Atlantic, from the British Isles to Mauritania, and the Mediterranean (Last et al., 2016). It occurs to depths of at least 160 m (Ellis et al., 2004). In Portugal, common stingrays are landed along most of the country's coast, although generally not labeled at the species level (Alves et al., 2020). Landings average 0.15 tonnes per year, while an additional 2.66 tonnes are landed as *Dasyatis* spp. (Alves et al., 2020). Declines in abundance of this species have been reported in the Central Mediterranean (Colloca et al., 2020) and the Atlantic coast of France (Quero, 1998). Coastal and shallow-occurring elasmobranchs are more threatened due to greater exposure to human activities like fishing, coastal development, and habitat degradation (Dulvy et al., 2021). In turn, the family Dasyatidae is one of the most endangered elasmobranch families (Dulvy et al., 2021) and common stingrays are currently catalogued as vulnerable by the IUCN Red List in this species' global, European and Mediterranean assessments (IUCN et al., 2021).

In this study, we tracked common stingrays using acoustic telemetry to understand their presence and movement patterns in the LSMP and surrounding area. Then we aimed at relating this to the marine park's design and regulations to find possible improvements. Therefore, our objectives in this study were to 1) estimate residency and space use of common stingrays within the LSMP; 2) study their small- and large-scale movement patterns and the influence of environmental variables (i.e., diel phases, temperature); and 3) evaluate the marine park's contribution to protecting this species under the current regulations and propose changes to improve them.

2. Materials and methods

2.1. Study area

The LSMP's 53 km^2 are divided into three protection levels: a total protection area -or marine reserve-of 4.3 km^2 to which access and any kind of fishing are prohibited; four partial protection areas totalling 21 km^2 where only octopus traps and jigs are allowed beyond 200 m from the coastline; and three complementary protection areas or buffer areas of a total 28 km^2 where only local licensed fishing boats under 7 m are allowed to operate. To the east is the Sado estuary (hereafter Sado) in which a natural reserve is in place, the Reserva Natural do Estuário do Sado. Additionally, the Tejo estuary (hereafter, Tejo) is found to the north, with its mouth at around 30 km in a straight line from the LSMP. Inside Tejo, another natural reserve is established, the Reserva Natural do Estuário do Tejo, which covers the innermost area of the estuary and

the land surrounding it (Fig. 1). It was implemented with the goal of protecting marine biodiversity and sustaining the livelihoods of the local fishers. The Tejo and Sado natural reserves are RAMSAR sites (wetland sites designated of international importance under the Ramsar Convention) due to their importance for many aquatic birds and as a nursery area for several fish and invertebrate species.

2.2. Study species

Common stingrays are viviparous, giving birth to litters of around 10 pups (Saadaoui et al., 2015). Pups are born after a gestation period estimated to last between four to six months (Ismen, 2003; Saadaoui et al., 2015), suggesting a yearly reproductive cycle (Saadaoui et al., 2015). Studies from various localities show that the reproductive season of common stingrays takes place during the warmer months of the year, in spring and summer (Chaikin et al., 2020; Ismen, 2003; Morey et al., 2006; Saadaoui et al., 2015). Estimates of maturity sizes have been obtained mostly from the central and eastern Mediterranean (Tunisia, north Aegean Sea, Turkey) and one study presented some data from Senegal. These estimations show regional variability; males mature at sizes between 22 and 33 cm in disc width and females between 24 and 42 cm disc width (Saadaoui et al., 2015). Common stingrays are mostly captured as bycatch, and depending on the location can be discarded (Tiralongo et al., 2018) or retained (Yaglioglu et al., 2015). Because of their venomous tail stinger, it is not uncommon to kill them before removing them from fishing gear (Tiralongo et al., 2018).

2.3. Tagging and tracking

We used monofilament trammel nets and longlines to capture stingrays inside the LSMP in April (n = 10) and October (n = 10) 2019 and in April (n = 10) and October (n = 1) 2021. The trammel nets were 500 m in length, 1.6 m in height, with inner panels of stretched mesh of 100 mm and outer panels of 600 mm. We deployed trammel nets in the morning (6–7 AM) and left them for 24 h. We mounted one longline with 120 hooks (size 4/0), baited them with chopped sardines and deployed it in the afternoon of October 15th, 2021 for 12 h. Thirty stingrays were captured with trammel nets and a single individual was captured with the longline.

We placed each captured specimen in an onboard tank with water and used a hydrophone to detect previously tagged individuals. We measured individuals (disc width, total length, and clasper length for males) and evaluated sex by looking for the presence of claspers. We made a small incision of approximately 2 cm with a scalpel in the peritoneal cavity to insert an acoustic transmitter and stitched it using absorbable suture. We selected the tag (Innovasea, either V9, V13, or V9P) based on stingray size and tag availability. Tags had a frequency of 69 kHz and an emission interval of 60–120 s, with an average of 90 s. Additionally, we fitted either a disc or t-tag (www.floytag.com) with a unique identifier and contact information to the pectoral fin. Before release, we checked that the inserted tag was operational using the hydrophone.

We monitored the study area using an array of 33 Innovasea VR2W acoustic receivers (Fig. 1). Not all receivers were active throughout the study period, and a few stations had receivers for shorter times than the study duration. We had four active receivers at different times in Sado and obtained additional detections from an array located in Tejo by accessing data available through the European Tracking Network (ETN). Previous evaluations of detection efficiency in the LSMP have shown no major variations due to environmental variables, diel patterns and/or background noises (Abecasis et al., 2014b; Sousa et al., 2019b).

2.4. Data analysis

Before analyzing the data, we visually examined individual detection plots to determine fish fates following Villegas-Ríos et al. (2020). We
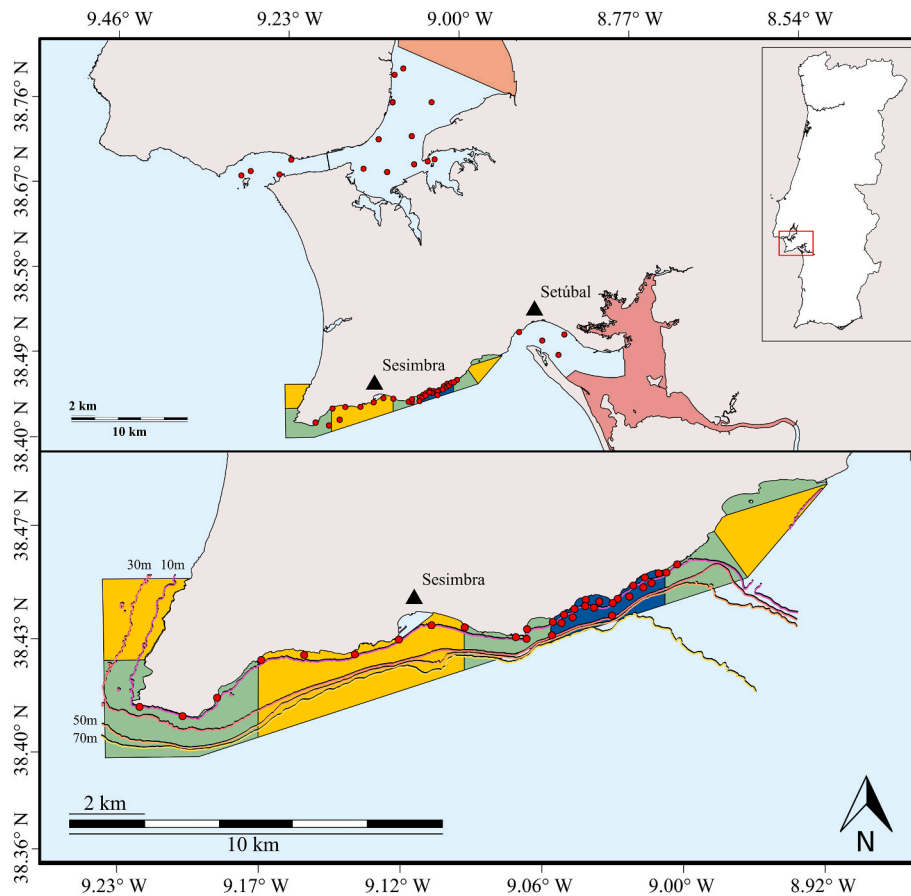
Fig. 1. Top panel shows a general overview of the Setúbal peninsula with the Tejo estuary and its Natural Reserve (orange) to the north and the Sado estuary and its Natural reserve (red) to the East. Also to the south is the Professor Luis Saldanha Marine Park (bottom panel). The three different protection levels are shown: the total protection area (blue), complementary protection areas (green), and buffer areas (yellow). Only the aquatic portions of the Natural Reserves are shown. The acoustic receiver stations are indicated by red circles and bathymetry lines are colour coded.

classified individuals that were detected throughout the study period, i. e., until or near the battery expiration date/last data download, as "survived". We calssified individuals for which detections were not recorded for a long time before the aforementioned dates, and whose last detections were in a peripheral station of the acoustic array as "dispersed". We classified individuals that were last detected in Sado as "dispersed" and "fishing mortality" because the array design did not allow distinguishing these two classes. We excluded dead individuals from further analysis (Table 1).

We estimated two residency indices (Kraft et al., 2023) separately for the LSMP and Sado: I_R, obtained as total number of days detected by at least one receiver (D_d) divided by the duration of the monitoring period (D_t, time between release and last data download or tag expiration dates): $I_R = D_d/D_t$, and as a weighted residency index (Iw_R), obtained by multiplying I_R by the days between first and last detection (D_i) divided by D_t: $I_{WR} = (D_d/D_t) \times (D_i/D_t)$. We replaced D_t with tag lifetime in the equations if the latter was shorter than the total monitoring period (Abecasis et al., 2013b).

We calculated centers of activity (COAs; Simpfendorfer et al., 2002) every 30 min to estimate space use in the LSMP using the dynamic Brownian bridge movement model (dBBMM) (Kranstauber et al., 2012). We selected this probability distribution-based occurrence estimator because the acoustic array only partially covered part of the stingrays' movements in the marine park and not their entire home range area. Occurrence estimators are a better fit in these situations because the former look into the space use during the monitoring period, while home range estimators assume the entire space use is surveyed (Fleming et al., 2015; Kraft et al., 2023).

We calculated minimum distance covered every 30 min as a proxy of activity in the LSMP, calculating a straight line between consecutive COAs (step length) using the R package adehabitatLT (Calenge, 2006). We estimated activity per diel phase by assigning them it into daytime, night, and twilight using the R package suncalc (Thieurmel and Elmarhraoui, 2019). "Daytime" included time between sunrise and sunset (appearance and disappearance of the sun over the horizon), "twilight" refers to the astronomical twilight (dawn and dusk), and "night" was the period between morning twilight and evening twilight. We only retained step lengths between consecutive COAs (i.e., every 30 min) and discarded longer step lengths (i.e., calculated between COAs that were apart 60 min or more). This is because, the individual could have left the area and then returned during the time with no detections, covering a much greater distance than the straight line obtained for those 60 min. We evaluated differences in step length and depth across these three categories (daytime, night, and twilight) with a repeated measures ANOVA (RM-ANOVA) using the R package stats (R Core Team, 2020). The average number of detections per hour per individual was calculated to investigate diel variations in the presence of individuals in the array, removing simultaneous detections first.

We investigated depth by calculating average depth every 10 min using data from two individuals. To standardize values and remove the influence of tides on depth estimations in the LSMP, we obtained local daily high and low tides for the study period using the R-package PTtidaltools (Martins, 2021) and subtracted tide height from the corresponding depth value. We conducted a *t*-test to compare depth differences among diel phases.

To evaluate the drivers of MPA use, we used a Generalized Additive

Table 1

Biological data, tagging data, and residency and space use values of the 31 tagged common stingrays. Tag durations marked with an asterisk indicate those that completed their battery life during the study duration. Tag delay was 90 s. Individuals that were considered as deceased or presented insufficient tracking data were not used in residency and space use estimations (Dp02 in the LSMP was the only exception). D_d: Number of days detected by at least one receiver, D_i number of days between first and last detection or detection interval, and D_t: monitoring period defined as the number of days between release and last data download/tag expiration date. Fish fate categories based on Villegas-Ríos et al. (2020).

Fish ID	Sex	Disc width (cm)	Release date	Tag type (Innovasea)	Tag duration (days)	Dt	LSMP				Sado				LSMP		Fish fate
							Dd	Di	IR	IWR	Dd	Di	IR	IWR	Core area (50%, km2)	Occurrence area (95%, km2)	
Dp 01	F	39.5	February 4, 2019	V9	651*	652	173	535	–	–	–	–	–	–	–	–	Deceased
Dp 02	M	29.6	March 4, 2019	V9	651*	652	3	3	–	–	96	282	–	–	0.80	3.33	Deceased
Dp 03	F	30.8	March 4, 2019	V9	651*	652	21	343	0.03	0.02	7	239	0.01	0.00	0.45	3.54	Fishing mortality/ dispersed
Dp 04	F	30.4	March 4, 2019	V9	651*	652	2	2	0.00	0.00	1	1	0.00	0.00	–	–	Fishing mortality/ dispersed
Dp 05	M	35.7	March 4, 2019	V13	1317	1130	331	1063	0.29	0.28	102	969	0.09	0.08	0.23	1.41	Survived
Dp 06	M	35.5	March 4, 2019	V13	1317	1130	155	366	0.14	0.04	9	207	0.01	0.00	0.22	1.02	Fishing mortality/ dispersed
Dp 07	M	31.1	March 4, 2019	V9	651*	652	191	368	0.29	0.17	17	346	0.03	0.01	0.30	1.88	Fishing mortality/ dispersed
Dp 08	F	30.2	March 4, 2019	V9	651*	652	106	329	0.16	0.08	5	131	0.01	0.00	0.35	2.10	Fishing mortality/ dispersed
Dp 09	M	28.5	March 4, 2019	V9	651*	652	2	2	0.00	0.00	–	–	–	–	–	–	Dispersed
Dp 10	F	35.5	March 4, 2019	V13	1317	1130	975	1127	–	–	–	–	–	–	–	–	Deceased
Dp 11	F	36.5	10/20/2019	V13	1317	930	113	738	0.12	0.10	53	760	0.06	0.05	0.61	2.78	Fishing mortality/ dispersed
Dp 12	M	36.0	10/20/2019	V9	651*	652	76	126	0.12	0.02	24	300	0.04	0.02	0.36	2.31	Fishing mortality/ dispersed
Dp 13	M	39.0	10/20/2019	V9	651*	652	128	454	0.20	0.14	76	576	0.12	0.10	0.15	1.13	Fishing mortality/ dispersed
Dp 14	M	34.5	10/20/2019	V9	651*	652	491	496	–	–	–	–	–	–	–	–	Deceased
Dp 15	M	39.0	10/20/2019	V9	651*	652	195	474	0.30	0.22	70	657	0.11	0.11	0.52	2.93	Survived
Dp 16	M	37.5	10/20/2019	V9	651*	652	271	505	0.42	0.32	40	655	0.06	0.06	0.37	1.88	Survived
Dp 17	F	39.5	10/20/2019	V9	651*	652	135	487	0.21	0.15	13	377	0.02	0.01	0.36	2.30	Dispersed
Dp 18	M	41.0	10/20/2019	V13	1317	930	136	166	0.15	0.03	13	26	0.01	0.00	0.19	0.99	Dispersed
Dp 19	F	38.5	10/22/2019	V9	651*	652	1	1	0.00	0.00	2	2	0.00	0.00	–	–	Fishing mortality/ dispersed
Dp 20	M	42.0	10/22/2019	V9P	404*	405	179	410	0.44	0.45	96	367	0.24	0.21	0.39	1.63	Survived
Dp 21	M	38.0	June 4, 2021	V9	651	396	163	375	0.41	0.39	7	376	0.02	0.02	0.16	1.38	Survived
Dp 22	F	25.0	July 4, 2021	V9	651	395	–	–	–	–	–	–	–	–	–	–	Not detected
Dp 23	F	26.0	July 4, 2021	V9	651	395	117	342	0.30	0.26	7	340	0.02	0.02	0.33	1.77	Survived
Dp 24	M	33.0	July 4, 2021	V9	651	395	154	354	0.39	0.35	2	154	0.01	0.00	0.16	1.17	Survived
Dp 25	F	34.0	July 4, 2021	V9	651	395	4	5	0.01	0.00	–	–	–	–	–	–	Dispersed
Dp 26	F	30.0	July 4, 2021	V9	651	395	1	1	0.00	0.00	1	1	0.00	0.00	–	–	Fishing mortality/ dispersed
Dp 27	M	31.0	July 4, 2021	V9	651	395	137	338	0.35	0.30	36	337	0.09	0.08	0.26	2.04	Survived
Dp 28	F	39.0	September 4, 2021	V9P	404	393	–	–	–	–	–	–	–	–	–	–	Not detected
Dp 29	M	25.0	September 4, 2021	V9	651	393	287	391	–	–	–	–	–	–	–	–	Deceased
Dp 30	F	26.0	September 4, 2021	V9	651	393	121	338	0.31	0.26	6	337	0.02	0.01	0.39	1.84	Survived
Dp 31	M	37.0	10/15/2021	V9P	404	204	120	151	0.59	0.44	5	151	0	0.02	0.20	1.37	Survived
Male avg.		34.9			758	636	160	368	0.29	0.22	38	394	0.06	0.05	0.27	1.63	
Female avg.		32.9			712	577	62	259	0.11	0.09	11	243	0.02	0.01	0.42	2.39	
Total avg.		34.0			738	611	119	322	0.22	0.17	27	332	0.04	0.04	0.32	1.87	

Model via the R package mgcv (Wood, 2011) to study the relationship between the presence of stingrays in the LSMP (as daily presence/absence) and sea surface temperature (SST), day of the year (doy), stingray size, sex, and individual. We used a cyclic cubic spline to model the effect of day of the year and temperature and treated individual-level effects as random. We obtained daily SST using the Copernicus Marine Service (https://marine.copernicus.eu) and evaluated the level of correlation between SST and day of the year. We used the function gam because it performs better with binary data than the gamm function (Wood, 2011). We obtained a correlation of 0.58, which can be considered a moderate degree of correlation. We decided to run the models without SST to favour a more conservative approach (Johnston et al., 2019). Furthermore, coastal SST estimates from satellite data can vary greatly from those obtained in situ (Smit et al., 2013) and SST estimates from large-scale satellite data can often have poor predictive performance (Santos et al., 2021). Finally, we tested different models combining the mentioned explanatory variables using the Akaike Information Criterion (AIC) in the R package AICcmodavg (Mazerolle, 2023). The full model (model 1) was:

Probability of presence ~ s (doy, bs = "cc") + size + sex + s (individual, bs = "re")

We tested variations to this model: model 2 did not consider size (doy + sex + individual), model 3 did not consider sex (doy + size + individual), and model 4 did not consider either (doy + individual).

3. Results

We captured and tagged 31 individuals (17 males, 14 females) between April 2019 and October 2021 (Table 1), thirty individuals using trammel nets and one using the longline (*Dasyatis pastinaca* [Dp] individual 31, released with the hook in). We tagged 14 individuals inside the total protection area and 17 in the partially protected areas adjacent to it. The sex ratio did not differ from an expected 1:1 ratio (two-tailed binomial test p = 0.12). Overall average disc width was 34 cm, 35 cm for males and 33 cm for females (Table 1). According to most estimations of size at maturity, all or most of the individuals qualify as mature (Saadaoui et al., 2015). Based on the criteria proposed by Villegas-Ríos et al. (2020) to identify post-release mortality, we identified at least five mortality events. The detection patterns of Dp 01, 10, 14 and 29 showed that they remained stationary shortly after release, suggesting they either dropped the tag or suffered from post-release mortality

(Villegas-Ríos et al., 2020). Similarly, after Dp 02 moved to Sado it remained there even after the other tagged individuals returned to the LSMP and was only detected by a single receiver. For these reasons we also considered it as potentially deceased. Dp 22 and 28 were never detected, suggesting they left the study area immediately after tagging (Fig. 2). We obtained an average of 611 monitoring days. Using the full dataset (LSMP, Sado, and Tejo), each stingray was detected on average on 143 days and had a detection interval of 379 days (Table 1). Dp 19 and 26 were not included in the calculations of occurrence area and step length because no detections.

3.1. Detection patterns and residency

The detection patterns noted among the 25 individuals (Fig. 2) can be sorted into two broad categories. The dominant pattern among individuals in this study was a consistent seasonal movement between the LSMP and Sado, while a few permanently dispersed from the LSMP at some point during the study, some after tagging (e.g., Dp 04, 09, 25, 26) and some after some months (Dp 18).

Individuals were mostly detected in the fully protected area (FPA) and its adjacent partially protected areas (PPAs), and only Dp 11, 18 and 27 were detected in the buffer area off the port of Sesimbra. Dp 18 was also the only stingray detected in the westernmost PPA at the tip of the peninsula on its way to Tejo, where it was detected over 60 km from its tagging point in two separate detection intervals. These intervals were at approximately the same time the other stingrays were detected in Sado: from April 28th to October 12th, 2020, and April 21st to October 17th, 2021 (Fig. 2). There was a three-month detection gap in the Tejo array during the six months between these two detection periods, from December 23rd, 2020, and March 23rd, 2121. Dp 18 was not detected in the LSMP again after moving to Tejo.

The overall residency in the LSMP was low (I_R = 0.21), ranging between 0.00 and 0.59 (Table 1). Residency was lower for females (I_R = 0.11) than males (I_R = 0.27). Monthly residencies were highest between October/November until March/April of each year, reaching values of I_R = 1 or very close to 1 several times (Fig. 2, supplementary material 1). During these periods the median detection frequency per individual was 154 s (range = 104–2801 s) after removing possible simultaneous detections (all detections under the delay interval of the tags). Only two of the 25 individuals had averages higher than 8 min (Dp 26 with 23.6 min and Dp 30 with 46.7 min). In contrast, from March/April to October/
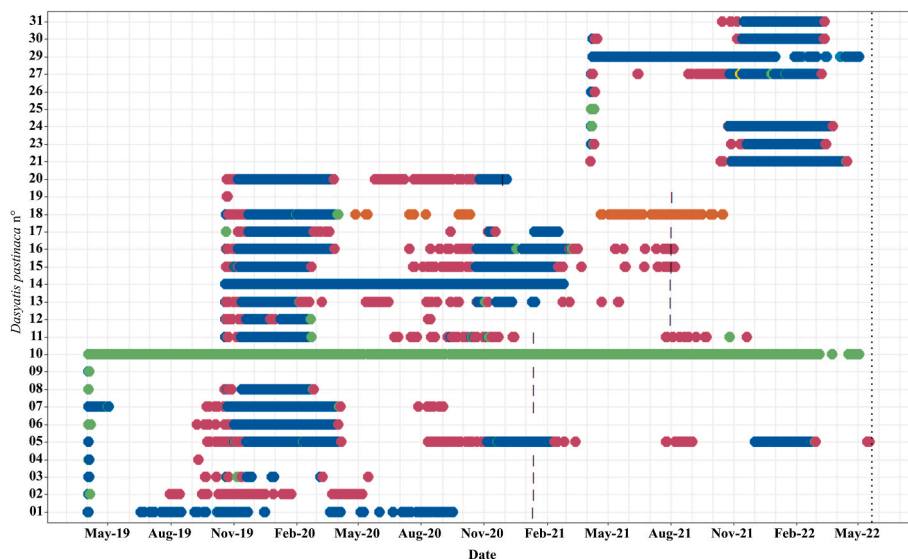

Fig. 2. Abacus plot of the stingrays tagged and monitored in this study, between April 2019 and May 2022. Detections are colour-coded as fully protected area (blue), partially protected area (green), buffer area (yellow), Sado (red) and Tejo (orange). All first detections were on the release date. Tag expiration dates are indicated by a vertical line and the dotted line indicates date of last data download.

November stingrays became virtually absent from the LSMP, as only a few were detected for short periods. A residency value of 0 was obtained between June–September 2019, May–August 2020, and May–October 2021. During these low residency periods in the LSMP, 23 of the 25 valid individuals (92%) travelled to Sado at least once, some in two and even three consecutive years (Fig. 2). Of the individuals detected in Sado, 14 were male and 9 were female. Males were detected in Sado on more days than females (average 42.4 vs. 10.5 days per individual) which resulted in a higher residency index (average $I_{R\ males}$ = 0.07, range 0.01–0.24 vs. $I_{R\ females}$ = 0.02, range 0.00–0.06). Brief ventures returning to the area where they departed from were seen a few times (i.e., a brief period back in Sado after moving to the LSMP and vice versa, e.g., Dp 03, 05, 08, 11, 13, 15 – Fig. 2).

3.2. Space use

We calculated activity space for 19 individuals (13 males and 6 females, Table 1) and excluded Dp 04, 09, 19, 25, and 26 because of an insufficient number of COAs. We also excluded the space use estimate of Dp 02 because of the low number of days (D_d = 3). The average 50% or core area size was 0.32 km^2 (range 0.15–0.80 km^2) and 95% or occurrence area was 1.87 km^2 (range 0.99–3.54 km^2). The locations of the space use estimations were mostly in the fully protected area, and all were of smaller size. Space use estimates were larger for females and showed significant statistical difference (Mann-Whitney *U* Test for 50% area estimates: U-value = 15, p = 0.04, and for 95% area estimates: U-value = 15, p = 0.04). Individual plots are available in supplementary material 2.

3.3. Activity

We obtained a total of 114729 step lengths of 30 min for 20 individuals and removed 3198 steps with intervals between 60 min and 351 days. In addition to the individuals already removed considered dead (Dp 01, 10, 14, 19, 26 and 29), we removed three Dp 04 and Dp 09 were detected only one day during daytime (8 and 15 steps respectively), and Dp 25 yielded insufficient step lengths (n = 7, all equal to zero), suggesting it remained static before leaving the array. Average step length was similar across individuals, except for Dp 02 which also had the lowest number of steps (n = 47) (Fig. 3, supplementary material 3). Between males and females, there was neither a statistically significative difference in the average number of steps (two-tailed *t*-test, t = 1.79, p = 0.09) nor in the average step length (two-tailed *t*-test, t =

−0.07, p = 0.95). Average distance was larger during twilight (64.09 m), followed by night (57.97 m) and lowest during daytime (48.22 m). During daytime a greater proportion of values of 0 or close to 0 was obtained compared to the night and crepuscular hours.

The average number of detections per hour per individual was similar throughout the day (Fig. 4). All individuals shared this pattern of steady detection frequency. The overall hourly averages ranged from 5.97 to 6.56 detections per hour, and average number of detections per hour per individual ranged between 0.05 (Dp 02) and 18.40 (Dp 05).

3.4. Depth

Two V9P tags produced enough information for analysis. We obtained a total of 21974 average depth values every 10 min for Dp 20 (male, 42.0 cm DW): 9284 for daytime, 9886 for night-time and 2804 values for twilight. The total for Dp 31 (male, 37.0 cm DW) was 14206: 6071 for daytime, 6268 for night-time and 1867 values for twilight. Following the correction for tidal height, average depth for Dp 20 was greatest during the night (16.1m ± 2.6m, range 5.4–26.9m), followed by twilight (15.6m ± 2.6m, range 8.4–25.9m) and shallowest during daytime(15.4m ± 2.8m, 3.5–23.3m). For Dp 31 the results were inverted, on average occurring deeper during daytime(12.4m ± 2.5, range 4.6–24.6m), followed by twilight (11.1m ± 3.1m, range 2.5–29.2m) and shallowest at night (10.6m ± 3.3m, range 1.0–27.3m).

3.5. Environmental drivers of MPA use

Of the four models tested, the AIC selected the full model (model 1, AIC = 5769.36), however, the model without size scored only slightly higher (model 2, AIC = 5769.39). The models that scored the highest were the model without sex (model 3, AIC = 5863.77) and the model that only considered individual effect in addition to the day of the year (model 4, AIC = 5863.81). Because of the small difference between models 1 and 2, we selected the latter (day of the year, sex and individual effect as variables) because it was simpler and gave similar results. For model 2 we obtained statistically significant relationships with the probability of presence for day of the year and individual effect (both p < 0.001), while sex was non-significant (p = 0.08) (Table 2).

Throughout the year, the highest probability of presence was in the autumn and winter months, i.e., the days at the start and end of the year. Contrastingly, the probability of presence fell to near zero or zero between these days, declining rapidly before the 100th day of the year and increased again, also rapidly, after the 300th day of the year (Fig. 5),
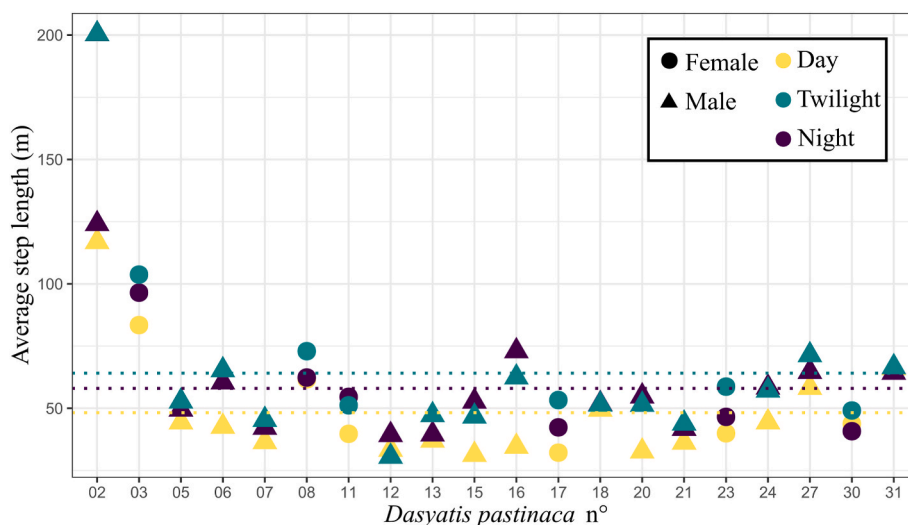

Fig. 3. Average step length per diel phase per individual: day (yellow), twilight (green), and night (purple). Females are shown with a circle and males with a triangle. Horizontal dotted lines indicate overall average. Detailed data on step length are available in the supplementary material.
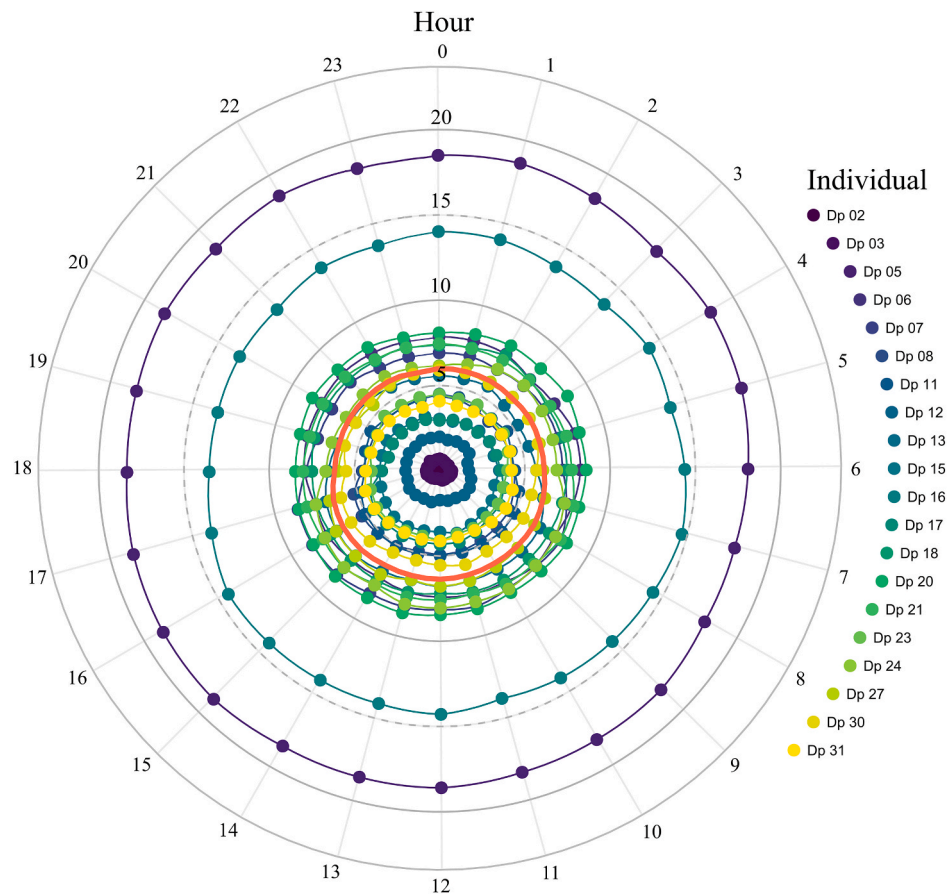
Fig. 4. Average number of detections per hour per individual over a 24-h period. Red line shows the overall hourly average.

Table 2
Summary of the model output investigating the probability of presence in the LSMP, showing the included variables.

Parametric coefficients	Estimate	Std. Error	z value	Pr (>|z|)
(Intercept)	−5.1468	0.7576	−6.794	0.4789
sex (male)	1.2844	0.7259	1.769	0.0768
Approximate significance of smooth terms:	edf	Ref.df	Chi.sq	p-value
s (doy)	6.418	8	51635	<0.001
s (individual)	16.794	17	1114	<0.001
R-sq. (adj) = 0.582, deviance explained = 55.6%, -REML = 2977.6, scale est. = 1, n = 11922				

roughly coinciding with the transition from winter to spring and from summer to autumn, respectively.

4. Discussion

In this study, we obtained long-term movement data of common stingrays and investigated diel and seasonal movements. Diel activity was higher during nocturnal and crepuscular hours compared to daytime hours. At a larger scale, most individuals showed a markedly seasonal presence in the marine park and appeared to be permanent/year-round residents of the southern coast of the Setúbal peninsula, moving seasonally between the LSMP and Sado. A few individuals dispersed after tagging, possibly to other areas (e.g., Dp 18 to Tejo). We were able to obtain these results because different sites were monitored during the study (LSMP, Sado, and Tejo), allowing us to identify other areas that

were visited and the time of the year in which they were visited. This is, to our knowledge, the first acoustic telemetry study on common stingrays in the world, through which we were able to enhance the understanding of their spatial ecology in this area, which will help in assessing and improving their protection.

4.1. Residency in the LSMP and seasonal pattern

During their presence in the LSMP, they moved mostly inside the FPA. The reasonably stable average number of detections per hour and among individuals suggests they do not engage in significant daily on/offshore movements, and likely stay close to the array throughout the entire day. If individuals were to make considerable movements towards deeper waters (and thus away from the acoustic array), as seen in other batoids (Humphries et al., 2017), we would expect greater intervals between detections. The depth data indicates something similar, as we only found small differences between diel phases. Despite the statistical significance, these depth differences might not be biologically significant. Although based on only two individuals, these results suggest that they might stay at similar depths throughout the day. Contrastingly, activity levels did vary throughout the day, in a pattern that is common among elasmobranchs (Hammerschlag et al., 2016). Greater nocturnal and crepuscular activity compared to daytime is usually associated with foraging (Hammerschlag et al., 2016), while the shorter average step length during daytime suggests that stingrays are less active and/or spend more time resting during these hours. Other species in the family Dasyatidae show similar patterns (Cartamil et al., 2003; Ward et al., 2019). It must be noted that net displacements of zero metres could be obtained if during successive COAs a stingray moved within the detection range of the same receiver. This underestimates activity because, despite there being movement, net displacement is zero because only
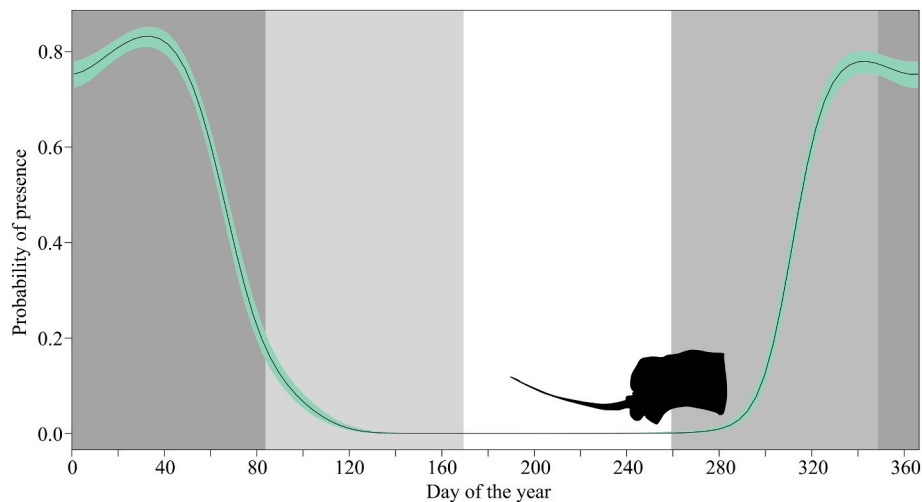
Fig. 5. Predicted probability of presence of common stingrays (n = 19) in the LSMP as a function of the day of the year. The area in green represents the 95% confidence interval. Grey shading represents seasons of the year (from left to right: winter, spring, summer, autumn, winter).

one pair of coordinates is included in the estimation of COAs. None-theless, increases in the distances covered were large enough to over-come this and note diel differences in activity.

Considering the design of the acoustic array, which was denser in the fully protected area, additional support for their preference can be found in previous work. During a biannual experimental fishing study con-ducted in the LSMP, sampling in autumn and spring from 2007 to 2015 and 2018–2019, no stingrays were captured in the buffer area (Martí-nez-Ramírez et al., 2021). One possibility is that this preference relates to the influence of bottom type on the distribution of batoids (Santos et al., 2021; Serra-Pereira et al., 2014). In the LSMP, most protection levels represent similar substrata (Henriques et al., 2015), however, the part closest to Sado is dominated by coarse sand, while in the part where the array is found (fully protected area and part of the adjacent partially protected areas) other bottom types first appear (fine, medium, and mixed sand, muddy to sandy mud) which these individuals seem to be favouring. In addition to closeness to Sado, this preference might be influenced by predator and/or prey distributions (although large pred-ators like sharks are not common in this area) and protection from currents to minimise energy expenditure (Campbell et al., 2012).

The seasonal residency pattern displayed by most stingrays and supported by the abacus plot and the GAM results can be defined as site fidelity or autumn and winter residency (Chapman et al., 2015) strongly influenced by the day of the year. This pattern did not appear to be influenced by the sex of individuals; however, this could also be a result of low sample size in the GAM. We obtained sex-related differences in residency, as males had an average residency over twice as high. The latter result could be influenced by the low number of individuals considered in the analysis. However, the GAM results did show inter-individual differences in presence, suggesting this is an important factor in shaping their residency patterns. This study adds to the similar seasonal patterns that have been reported for this species in other areas (Chaikin et al., 2020; Yaglioglu et al., 2015).

Seasonal movements occur for various reasons, like in response to fluctuations in prey availability, predator avoidance, shifts in environ-mental conditions, and onset of the reproductive season (Jaine et al., 2014; Papastamatiou and Lowe, 2012). Regarding the latter, reproduc-tive migrations are undertaken by mature individuals, and estimates of maturity size of common stingrays have been obtained elsewhere and show high regional variability (Saadaoui et al., 2015). Disc width at 50% maturity ranged from 22 to 33 cm for males and 24–42 cm for females (including transformed total length values in studies providing the equations to convert them). Using the lowest maturity sizes results in all individuals being classified as mature, while using the greatest size

estimates yields most males and none of the females as mature. Considering this, we assume an intermediate scenario where most in-dividuals of both sexes are mature. Joint migration of mature and large sub-adults have been reported, for example, in smalltooth sawfish *Pristis pectinata* (Graham et al., 2021). The same area could be used for different purposes, like reproduction for mature individuals and sea-sonal feeding grounds/refuges for immature individuals and subadults.

Further evidence to link the seasonal migration and Sado to the reproduction of common stingrays can be drawn from the timing of their movement and duration of their absence from the LSMP, as these coincide with the time and span of their reproductive season as shown by studies in other areas. For example, high abundances of stingrays have been reported in the Balearic Islands in June, likely for reproduc-tion (Morey et al., 2006); pregnant females with fully developed offspring appear in the Gulf of Gabès in June, neonates during the end of June and beginning of July (Saadaoui et al., 2015); and pregnant fe-males have been seen in Iskenderun Bay between May and early September and parturition events beginning in July (Ismen, 2003). Similarly, in the coast of the Levant stingrays aggregate and display courtship behaviour in March and more females in advanced gravid states are seen in June (Chaikin et al., 2020). The gestation period is estimated to last from four to six months (Ismen, 2003; Saadaoui et al., 2015), which suggest a yearly reproductive cycle (Saadaoui et al., 2015). Finally, estuaries provide shelter and thermoregulatory advan-tages for the reproduction of several elasmobranchs (Schlaff et al., 2014). The higher temperatures of estuaries are thought to provide stingrays with energetic advantages that shorten the gestation period (Jirik and Lowe, 2012). Considering this evidence, this seasonal pattern could be interpreted as parturition site fidelity to Sado (Chapman et al., 2015).

4.2. Conservation and management

Interpreting protection as being proportional to time spent inside a protected area, the overall (year-round) protection received by common stingrays in the LSMP is low, as indicated by the low global residency index (I_R = 0.21). However, residency also fluctuated greatly throughout the year as an outcome of their migration, which highlights the relevance of understanding long-term movements to optimize the efficiency of conservation and management efforts.

Stingrays mainly occupied the FPA and adjacent PPAs, where elas-mobranchs in general are under full protection because the activities permitted in them do not target these species. In the part of the LSMP where they were mostly detected the protected areas cover a continuous

11.09 km^2, over three times the largest 95% space use area we obtained However, the marine park is much narrower in an offshore direction, between approximately 450 m to a maximum of under 2 km along this segment. Therefore, compared to movements along the coast, offshore movements could expose them more to edge effect (Ohayon et al., 2021) and fishing the line (Kellner et al., 2007).

When migrating to Sado, stingrays seasonally expanded their space use beyond the LSMP's boundaries, which increases risk exposure (Villegas-Ríos et al., 2021). A natural reserve is already in place in Sado, however, the receivers at its entrance also were the ones to detect nine out of 23 individuals (39%) for the last time. This could be attributed to tag malfunction or loss, dispersal (e.g., Dp 18), but also to fishing mortality. However, with our study design we were not able to discern whether or not they were captured and if this occurred inside or outside of the reserve. The area between the LSMP and Sado is not protected and, at least until the LSMP's establishment, near-shore illegal drift nets were placed east of the marine park during spring and summer (Horta e Costa et al., 2013). Considering this, a permanent or seasonal protected corridor linking the LSMP and the Reserva Natural do Estuário do Sado could improve the contribution of both areas to the protection of common stingrays. For example, restrictions on fishing gear (e.g. banning the use of trammel nets and longlines) would reduce exposure to fishing pressure and improve the overall performance of the marine park (Chapman et al., 2005) and could even enhance the LSMP's early signs of reserve effect. Preliminary evidence of increases in size, biomass and abundance was found for four batoid species (Martínez-Ramírez et al., 2021; Sousa et al., 2018). Although some of the cited results must be interpreted with caution (Martínez-Ramírez et al., 2021), such findings are usually regarded as one of the initial signs of reserve effect (Di Franco et al., 2009). An increase in total length was also detected for common stingrays, although this data was not analyzed in depth due to sampling issues (Martínez-Ramírez et al., unpublished data). Future evaluations of the LSMP will be well complemented by the results on the movement patterns of common stingrays presented here, to adapt the currently implemented protection measures.

4.3. Shortcomings, future directions

Possible sources of bias in our study design could be found in the array's greater coverage of the FPA compared to the PPAs and BAs, and the tendency of some animals to remain around their tagging location (Espinoza et al., 2015). However, these results can also be the outcome of true area preferences for a few reasons. First, while stingrays were in the LSMP, the detection frequency median was of 154 s, suggesting a steady presence in this area. Secondly, during the nearly three years of the study only three individuals were briefly detected by the receivers in the buffer area, suggesting little use of this area. These could be transit areas in some cases, like for Dp 18. Lastly, during a previous experimental fishing study that took place twice a year over a total of 11 non-consecutive years no stingrays were captured in the buffer area (Martínez-Ramírez et al., 2021), hence the difficulty of tagging individuals in those areas.

Calculating step lengths using COAs served as proxy of the activity, however, this approach is too coarse to robustly infer their activity patterns. Future efforts using better suited devices like acceleration tags could produce data of greater detail and provide better estimations (Pereñíguez et al., 2022).

Other future efforts should focus on exploring sex-related differences. Although these results were not all robust, the much higher residency of males suggests there might be differences in protection, which should be clarified in the future to detect possible inequalities in protection.

Finally, the dispersal of individuals and the detection of one in Tejo using data obtained from the ETN, exemplify the need and benefits of collaborations to expand detection capabilities in future studies. Establishing collaborations with researchers that operate acoustic arrays in other areas can fulfil this requirement, improving subsequent studies and eventually the data that is used in conservation and management decisions (Abecasis et al., 2018; Ellis et al., 2019).

5. Contributions

DA and SK conceptualized and designed the study. SK, ACW and DA captured and tagged the individuals of this study and acquired the data. SK, ACW and DA maintained the receiver network. SK conducted the analyses, prepared the figures for publication, and drafted the article. SK, ACW and DA interpreted the data. SK, ACW and DA revised it critically for important intellectual content. All authors have approved the final article.

6. Permits

Capture, handling, and tagging were done under the 'catch and release' practices of the Portuguese Institute for Nature Conservation and Forests (permit n°11478683) and the Veterinary General Directorate (permit n° 2018-08-29 015730).

Declaration of Generative AI and AI-assisted technologies in the writing process

No Generative AI and AI-assisted technologies were used to write this manuscript.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ocecoaman.2023.106891.

References

Abecasis, D., Afonso, P., Erzini, K., 2014a. Combining multispecies home range and distribution models aids assessment of MPA effectiveness. Mar. Ecol. Prog. Ser. 513, 155–169. https://doi.org/10.3354/meps10987.

Abecasis, D., Afonso, P., Erzini, K., 2014b. Can small MPAs protect local populations of a coastal flatfish, *Solea senegalensis*? Fish. Manag. Ecol. 21, 175–185. https://doi.org/10.1111/fme.12061.

Abecasis, D., Afonso, P., O'Dor, R.K., Erzini, K., 2013. Small MPAs do not protect cuttlefish (*Sepia officinalis*). Fish. Res. 147, 196–201. https://doi.org/10.1016/j.fishres.2013.05.004.

Abecasis, D., Horta e Costa, B., Afonso, P., Gonçalves, E.J., Erzini, K., 2015. Early reserve effects linked to small home ranges of a commercial fish, *Diplodus sargus*, Sparidae. Mar. Ecol. Prog. Ser. 518, 255–266. https://doi.org/10.3354/meps11054.

Abecasis, D., Steckenreuter, A., Reubens, J., Aarestrup, K., Alós, J., Badalamenti, F., Bajona, L., Boylan, P., Deneudt, K., Greenberg, L., Brevé, N., Hernández, F., Humphries, N., Meyer, C., Sims, D., Thorstad, E.B., Walker, A.M., Whoriskey, F., Afonso, P., 2018. A review of acoustic telemetry in Europe and the need for a regional aquatic telemetry network. Animal Biotelemetry 6, 12. https://doi.org/10.1186/s40317-018-0156-0.

Alves, L.M.F., Correia, J.P.S., Lemos, M.F.L., Novais, S.C., Cabral, H., 2020. Assessment of trends in the Portuguese elasmobranch commercial landings over three decades (1986–2017). Fish. Res. 230, 105648. https://doi.org/10.1016/j.fishres.2020.105648.

Ban, N.C., Gurney, G.G., Marshall, N.A., Whitney, C.K., Mills, M., Gelcich, S., Bennett, N.J., Meehan, M.C., Butler, C., Ban, S., Tran, T.C., Cox, M.E., Breslow, S.J., 2019. Well-being outcomes of marine protected areas. Nat. Sustain. 2, 524–532. https://doi.org/10.1038/s41893-019-0306-2.

Cabral, S., 2014. Raja Spp. Spatial Dynamics in the Arrábida Marine Park. M. Sc. Thesis), Universidade do Algarve.

Calenge, C., 2006. The package "adehabitat" for the R software: a tool for the analysis of space and habitat use by animals. Ecol. Model. 197, 516–519. https://doi.org/10.1016/j.ecolmodel.2006.03.017.

Campbell, H.A., Hewitt, M., Watts, M.E., Peverell, S., Franklin, C.E., 2012. Short- and long-term movement patterns in the freshwater whipray (*Himantura dalyensis*) determined by the signal processing of passive acoustic telemetry data. Mar. Freshw. Res. 63, 341–350. https://doi.org/10.1071/MF11229.

Cartamil, D.P., Vaudo, J.J., Lowe, C.G., Wetherbee, B.M., Holland, K.N., 2003. Diel movement patterns of the Hawaiian stingray, Dasyatis lata: implications for ecological interactions between sympatric elasmobranch species. Mar. Biol. 142, 841–847. https://doi.org/10.1007/s00227-003-1014-y.

Chaikin, S., Belmaker, J., Barash, A., 2020. Coastal breeding aggregations of threatened stingrays and guitarfish in the Levant. Aquat. Conserv. Mar. Freshw. Ecosyst. 30, 1160–1171. https://doi.org/10.1002/aqc.3305.

Chapman, D.D., Feldheim, K.A., Papastamatiou, Y.P., Hueter, R.E., 2015. There and back again: a review of residency and return migrations in sharks, with implications for population structure and management. Ann. Rev. Mar. Sci 7, 547–570. https://doi.org/10.1146/annurev-marine-010814-015730.

Chapman, D.D., Pikitch, E.K., Babcock, E., Shivji, M.S., 2005. Marine reserve design and evaluation using automated acoustic telemetry: a case-study involving coral reef-associated sharks in the mesoamerican caribbean. Mar. Technol. Soc. J. 39, 42–55. https://doi.org/10.4031/002533205787521640.

Chevis, M.G., Godley, B.J., Lewis, J.P., Jackson Lewis, J., Scales, K.L., Graham, R.T., 2017. Movement patterns of juvenile hawksbill turtles *Eretmochelys imbricata* at a Caribbean coral atoll: long-term tracking using passive acoustic telemetry. Endanger. Species Res. 32, 309–319. https://doi.org/10.3354/esr00812.

Colloca, F., Carrozzi, V., Simonetti, A., Di Lorenzo, M., 2020. Using local ecological knowledge of Fishers to reconstruct abundance trends of elasmobranch populations in the strait of sicily. Front. Mar. Sci. 7, 508. https://doi.org/10.3389/fmars.2020.00508.

Cunha, A.H., Erzini, K., Serrão, E.A., Gonçalves, E., Borges, R., Henriques, M., Henriques, V., Guerra, M., Duarte, C., Marbá, N., Fonseca, M., 2014. Biomares, a LIFE project to restore and manage the biodiversity of prof. Luiz Saldanha marine park. J. Coast Conserv. 18, 643–655. https://doi.org/10.1007/s11852-014-0336-x.

Davidson, L.N.K., Dulvy, N.K., 2017. Global marine protected areas to prevent extinctions. Nature Ecology & Evolution 1, 0040. https://doi.org/10.1038/s41559-016-0040.

Di Franco, A., Bussotti, S., Navone, A., Panzalis, P., Guidetti, P., 2009. Evaluating effects of total and partial restrictions to fishing on Mediterranean rocky-reef fish assemblages. Mar. Ecol. Prog. Ser. 387, 275–285. https://doi.org/10.3354/meps08051.

Dudley, N. (Ed.), 2008. Guidelines for Applying Protected Area Management Categories. IUCN, Gland, Switzerland.

Dulvy, N.K., Pacoureau, N., Rigby, C.L., Pollom, R.A., Jabado, R.W., Ebert, D.A., Finucci, B., Pollock, C.M., Cheok, J., Derrick, D.H., Herman, K.B., Sherman, C.S., VanderWright, W.J., Lawson, J.M., Walls, R.H.L., Carlson, J.K., Charvet, P., Bineesh, K.K., Fernando, D., Ralph, G.M., Matsushiba, J.H., Hilton-Taylor, C., Fordham, S.V., Simpfendorfer, C.A., 2021. Overfishing drives over one-third of all sharks and rays toward a global extinction crisis. Curr. Biol. 31, 4773–4787.e8. https://doi.org/10.1016/j.cub.2021.08.062.

Dwyer, R.G., Krueck, N.C., Udyawer, V., Heupel, M.R., Chapman, D., Pratt, H.L., Garla, R., Simpfendorfer, C.A., 2020. Individual and population benefits of marine reserves for reef sharks. Curr. Biol. 30, 480–489.e5. https://doi.org/10.1016/j.cub.2019.12.005.

Ellis, J.R., Cruz-Martínez, A., Rackham, B.D., Rogers, S.I., 2004. The distribution of chondrichthyan fishes around the British Isles and implications for conservation. J. Northwest Atl. Fish. Sci. 35, 195–213. https://doi.org/10.2960/J.v35.m485.

Ellis, R.D., Flaherty-Walia, K.E., Collins, A.B., Bickford, J.W., Boucek, R., Walters Burnsed, S.L., Lowerre-Barbieri, S.K., 2019. Acoustic telemetry array evolution: from species- and project-specific designs to large-scale, multispecies, cooperative networks. Fish. Res. 209, 186–195. https://doi.org/10.1016/j.fishres.2018.09.015.

Espinoza, M., Heupel, MichelleR., Tobin, A.J., Simpfendorfer, C.A., 2015. Movement patterns of silvertip sharks (*Carcharhinus albimarginatus*) on coral reefs. Coral Reefs 34, 807–821. https://doi.org/10.1007/s00338-015-1312-0.

Fiúza, A.F.G., 1983. Upwelling patterns off Portugal. In: Suess, E., Thiede, J. (Eds.), Coastal Upwelling its Sediment Record. Springer US, Boston, MA, pp. 85–98. https://doi.org/10.1007/978-1-4615-6651-9_5.

Fleming, C.H., Fagan, W.F., Mueller, T., Olson, K.A., Leimgruber, P., Calabrese, J.M., 2015. Rigorous home range estimation with movement data: a new autocorrelated kernel density estimator. Ecology 96, 1182–1188. https://doi.org/10.1890/14-2010.1.

Gell, F.R., Roberts, C.M., 2003. Benefits beyond boundaries: the fishery effects of marine reserves. Trends Ecol. Evol. 18, 448–455. https://doi.org/10.1016/S0169-5347(03)00189-7.

Gonçalves, E.J., Henriques, R., Almada, V.C., 2002. Use of a temperate reef-fish community to identify priorities in the establishment of a Marine Protected Area. In: Beumer, J.P., Grant, A., Smith, D.C. (Eds.), Aquatic Protected Areas: what Works Best and How Do We Know? Proceedings of the World Congress on Aquatic Protected Areas, pp. 261–272. Cairns, Australia.

Graham, J., Kroetz, A., Poulakis, G., Scharer, R., Carlson, J., Lowerre-Barbieri, S., Morley, D., Reyier, E., Grubbs, R., 2021. Large-scale space use of large juvenile and adult smalltooth sawfish *Pristis pectinata*: implications for management. Endanger. Species Res. 44, 45–59. https://doi.org/10.3354/esr01088.

Hammerschlag, N., Skubel, R.A., Calich, H., Nelson, E.R., Shiffman, D.S., Wester, J., Macdonald, C.C., Cain, S., Jennings, L., Enchelmaier, A., Gallagher, A.J., 2016. Nocturnal and crepuscular behavior in elasmobranchs: a review of movement, habitat use, foraging, and reproduction in the dark. BMS 93, 355–374. https://doi.org/10.5343/bms.2016.1046.

Henriques, M., Gonçalves, E., Almada, V., 2007. Rapid shifts in a marine fish assemblage follow fluctuations in winter sea conditions. Mar. Ecol. Prog. Ser. 340, 259–270. https://doi.org/10.3354/meps340259.

Henriques, V., Guerra, M., Gaudêncio, M.J., Fonseca, P., Campos, A., Mendes, B., 2009. Biomares. Restoration and Management of Biodiversity in the Marine Park Site Arrábida-Espichel (PTCON0010). Action A4. Marine Habitat Characterization. Progress Report 2009. Relatório IPIMAR.

Henriques, V., Guerra, M.T., Mendes, B., Gaudêncio, M.J., Fonseca, P., 2015. Benthic habitat mapping in a Portuguese Marine Protected Area using EUNIS: an integrated approach. J. Sea Res. 100, 77–90. https://doi.org/10.1016/j.seares.2014.10.007.

Heupel, M.R., Simpfendorfer, C.A., 2005. Using acoustic monitoring to evaluate MPAs for shark nursery areas: the importance of long-term data. Mar. Technol. Soc. J. 39, 10–18. https://doi.org/10.4031/002533205787521749.

Horta e Costa, B., Gonçalves, L., Gonçalves, E.J., 2013. Vessels' site fidelity and spatio-temporal distribution of artisanal fisheries before the implementation of a temperate multiple-use marine protected area. Fish. Res. 148, 27–37. https://doi.org/10.1016/j.fishres.2013.08.001.

Humphries, N., Simpson, S., Sims, D., 2017. Diel vertical migration and central place foraging in benthic predators. Mar. Ecol. Prog. Ser. 582, 163–180. https://doi.org/10.3354/meps12324.

Hussey, N.E., Kessel, S.T., Aarestrup, K., Cooke, S.J., Cowley, P.D., Fisk, A.T., Harcourt, R.G., Holland, K.N., Iverson, S.J., Kocik, J.F., Mills Flemming, J.E., Whoriskey, F.G., 2015. Aquatic animal telemetry: a panoramic window into the underwater world. Science 348, 1255642. https://doi.org/10.1126/science.1255642, 1255642.

Ismen, A., 2003. Age, growth, reproduction and food of common stingray (*Dasyatis pastinaca* L., 1758) in İskenderun Bay, the eastern Mediterranean. Fish. Res. 60, 169–176. https://doi.org/10.1016/S0165-7836(02)00058-9.

IUCN, Jabado, R.W., Chartrain, E., De Bruyne, G., Derrick, D., Dia, M., Diop, M., Doherty, P., Leurs, G.H.L., Metcalfe, K., Pacoureau, N., Pires, J.D., Ratão, S., Seidu, I., Serena, F., Soares, A.-L., Tamo, A., VanderWright, W.J., Williams, A.B., 2021. The IUCN red list of threatened species. https://doi.org/10.2305/IUCN.UK.2021-2RLTS.T161453A124488102 en.

Jaine, F., Rohner, C., Weeks, S., Couturier, L., Bennett, M., Townsend, K., Richardson, A., 2014. Movements and habitat use of reef manta rays off eastern Australia: offshore excursions, deep diving and eddy affinity revealed by satellite telemetry. Mar. Ecol. Prog. Ser. 510, 73–86. https://doi.org/10.3354/meps10910.

Jirik, K.E., Lowe, C.G., 2012. An elasmobranch maternity ward: female round stingrays *Urobatis halleri* use warm, restored estuarine habitat during gestation. J. Fish. Biol. 80, 1227–1245. https://doi.org/10.1111/j.1095-8649.2011.03208.x.

Johnston, J.D., Dunn, C.J., Vernon, M.J., 2019. Tree traits influence response to fire severity in the western Oregon Cascades, USA. For. Ecol. Manag. 433, 690–698. https://doi.org/10.1016/j.foreco.2018.11.047.

Kellner, J.B., Tetreault, I., Gaines, S.D., Nisbet, R.M., 2007. Fishing the line near marine reserves in single and multispecies fisheries. Ecol. Appl. 17, 1039–1054. https://doi.org/10.1890/05-1845.

Kraft, S., Gandra, M., Lennox, R.J., Mourier, J., Winkler, A.C., Abecasis, D., 2023. Residency and space use estimation methods based on passive acoustic telemetry data. Mov Ecol 11, 12. https://doi.org/10.1186/s40462-022-00364-z.

Kranstauber, B., Kays, R., LaPoint, S.D., Wikelski, M., Safi, K., 2012. A dynamic Brownian bridge movement model to estimate utilization distributions for heterogeneous

animal movement: the dynamic Brownian bridge movement model. J. Anim. Ecol. 81, 738–746. https://doi.org/10.1111/j.1365-2656.2012.01955.x.

Last, P.R., Manjaji-Matsumoto, B.M., Naylor, G.J.P., White, W.T., 2016. Stingrays, family Dasyatidae. In: Last, P.R., White, William Toby, de Carvalho, M.R., Séret, B., Stehmann, M., Naylor, Gavin J.P., Marshall, L. (Eds.), Rays of the World. CSIRO Publishing, Clayton South VIC, pp. 522–618.

Le Port, A., Lavery, S., Montgomery, J.C., 2012. Conservation of coastal stingrays: seasonal abundance and population structure of the short-tailed stingray Dasyatis brevicaudata at a Marine Protected Area. ICES (Int. Counc. Explor. Sea) J. Mar. Sci. 69, 1427–1435. https://doi.org/10.1093/icesjms/fss120.

Lédée, E.J.I., Heupel, M.R., Tobin, A.J., Knip, D.M., Simpfendorfer, C.A., 2015. A comparison between traditional kernel-based methods and network analysis: an example from two nearshore shark species. Anim. Behav. 103, 17–28. https://doi.org/10.1016/j.anbehav.2015.01.039.

Lester, S., Halpern, B., Grorud-Colvert, K., Lubchenco, J., Ruttenberg, B., Gaines, S., Airamé, S., Warner, R., 2009. Biological effects within no-take marine reserves: a global synthesis. Mar. Ecol. Prog. Ser. 384, 33–46. https://doi.org/10.3354/meps08029.

MacKeracher, T., Diedrich, A., Simpfendorfer, C.A., 2019. Sharks, rays and marine protected areas: a critical evaluation of current perspectives. Fish Fish. 20, 255–267. https://doi.org/10.1111/faf.12337.

Martínez Ramírez, L., Priester, C.R., Sousa, I., Erzini, K., Abecasis, D., 2021. Reserve effect of a small North-East Atlantic marine protected area (Arrábida, Portugal) on soft-sediment fish species. Aquat. Conserv. Mar. Freshw. Ecosyst. 31, 1584–1595. https://doi.org/10.1002/aqc.3577.

Martins, M., 2021. PTtidaltools - R Package to Retrieve and Manipulate Data from the Portuguese National Hydrographic Institute. https://doi.org/10.5281/ZENODO.4562490.

Matley, J.K., Klinard, N.V., Barbosa Martins, A.P., Aarestrup, K., Aspillaga, E., Cooke, S.J., Cowley, P.D., Heupel, M.R., Lowe, C.G., Lowerre-Barbieri, S.K., Mitamura, H., Moore, J.-S., Simpfendorfer, C.A., Stokesbury, M.J.W., Taylor, M.D., Thorstad, E.B., Vandergoot, C.S., Fisk, A.T., 2021. Global trends in aquatic animal tracking with acoustic telemetry. Trends in Ecology & Evolution S0169534721002470. https://doi.org/10.1016/j.tree.2021.09.001.

Mazerolle, M.J., 2023. AICcmodavg: Model Selection and Multimodel Inference Based on (Q)AIC(c). R package version 2.3.2.

Morey, G., Moranta, J., Riera, F., Grau, A.M., Morales-Nin, B., 2006. Elasmobranchs in trammel net fishery associated to marine reserves in the Balearic Islands (NW Mediterranean). Cybium 30, 125–132.

Musick, J.A., 1999. Life in the slow lane: ecology and conservation of long-lived marine animals. In: Musick, J.A. (Ed.), Ecology and Conservation of Long-Lived Marine Animals. American Fisheries Society. https://doi.org/10.47886/9781888569155.

Ohayon, S., Granot, I., Belmaker, J., 2021. A meta-analysis reveals edge effects within marine protected areas. Nat Ecol Evol 5, 1301–1308. https://doi.org/10.1038/s41559-021-01502-3.

Papastamatiou, Y.P., Lowe, C.G., 2012. An analytical and hypothesis-driven approach to elasmobranch movement studies. J. Fish. Biol. 80, 1342–1360. https://doi.org/10.1111/j.1095-8649.2012.03232.x.

Pereñíguez, J.M., Venerus, L.A., Gutiérrez-Cánovas, C., Abecasis, D., Ciancio, J.E., Jiménez-Montalbán, P., García-Charton, J.A., 2022. Acoustic telemetry and accelerometers: a field comparison of different proxies for activity in the marine environment. ICES (Int. Counc. Explor. Sea) J. Mar. Sci. 79, 2600–2613. https://doi.org/10.1093/icesjms/fsac190.

Quero, J.-C., 1998. Changes in the Euro-Atlantic fish species composition resulting from fishing and ocean warming. Ital. J. Zool. 65, 493–499. https://doi.org/10.1080/11250009809386873.

R Core Team, 2020. R: A Language and Environment for Statistical Computing.

Saadaoui, A., Saïdi, B., Enajjar, S., Bradai, M.N., 2015. Reproductive biology of the common stingray Dasyatis pastinaca (linnaeus, 1758) off the Gulf of Gabès (central mediterranean sea). Cah. Biol. Mar. 56, 389–396.

Santos, R., Medeiros-Leal, W., Novoa-Pabon, A., Crespo, O., Pinho, M., 2021. Biological knowledge of thornback ray (Raja clavata) from the azores: improving scientific information for the effectiveness of species-specific management measures. Biology 10, 676. https://doi.org/10.3390/biology10070676.

Schlaff, A.M., Heupel, M.R., Simpfendorfer, C.A., 2014. Influence of environmental factors on shark and ray movement, behaviour and habitat use: a review. Rev. Fish Biol. Fish. 24, 1089–1103. https://doi.org/10.1007/s11160-014-9364-8.

Serra-Pereira, B., Erzini, K., Maia, C., Figueiredo, I., 2014. Identification of potential essential fish habitats for skates based on Fishers' knowledge. Environ. Manag. 53, 985–998. https://doi.org/10.1007/s00267-014-0257-3.

Simpfendorfer, C.A., Heupel, M.R., Hueter, R.E., 2002. Estimation of short-term centers of activity from an array of omnidirectional hydrophones and its use in studying animal movements. Can. J. Fish. Aquat. Sci. 59, 23–32. https://doi.org/10.1139/f01-191.

Smit, A.J., Roberts, M., Anderson, R.J., Dufois, F., Dudley, S.F.J., Bornman, T.G., Olbers, J., Bolton, J.J., 2013. A coastal seawater temperature dataset for biogeographical studies: large biases between in situ and remotely-sensed data sets around the coast of South Africa. PLoS One 8, e81944. https://doi.org/10.1371/journal.pone.0081944.

Sousa, I., Baeyaert, J., Gonçalves, J.M.S., Erzini, K., 2019. Preliminary insights into the spatial ecology and movement patterns of a regionally critically endangered skate (Rostroraja alba) associated with a marine protected area. Mar. Freshw. Behav. Physiol. 52, 283–299. https://doi.org/10.1080/10236244.2019.1705805.

Sousa, I., Gonçalves, J.M.S., Claudet, J., Coelho, R., Gonçalves, E.J., Erzini, K., 2018. Soft-bottom fishes and spatial protection: findings from a temperate marine protected area. PeerJ 6, e4653. https://doi.org/10.7717/peerj.4653.

Speed, C.W., Cappo, M., Meekan, M.G., 2018. Evidence for rapid recovery of shark populations within a coral reef marine protected area. Biol. Conserv. 220, 308–319. https://doi.org/10.1016/j.biocon.2018.01.010.

Thieurmel, B., Elmarhraoui, A., 2019. R Package "Suncalc": Compute Sun Position, Sunlight Phases, Moon Position and Lunar Phase, vol. 10.

Tiralongo, F., Messina, G., Lombardo, B.M., 2018. Discards of elasmobranchs in a trammel net fishery targeting cuttlefish, Sepia officinalis Linnaeus, 1758, along the coast of Sicily (central Mediterranean Sea). Regional Studies in Marine Science 20, 60–63. https://doi.org/10.1016/j.rsma.2018.04.002.

Villegas-Ríos, D., Claudet, J., Freitas, C., Moland, E., Thorbjørnsen, S.H., Alonso-Fernández, A., Olsen, E.M., 2021. Time at risk: individual spatial behaviour drives effectiveness of marine protected areas and fitness. Biol. Conserv. 263, 109333. https://doi.org/10.1016/j.biocon.2021.109333.

Villegas Ríos, D., Freitas, C., Moland, E., Thorbjørnsen, S.H., Olsen, E.M., 2020. Inferring individual fate from aquatic acoustic telemetry data. Methods Ecol. Evol. 11, 1186–1198. https://doi.org/10.1111/2041-210X.13446.

Ward, C.R.E., Bouyoucos, I.A., Brooks, E.J., O'Shea, O.R., 2019. Novel attachment methods for assessing activity patterns using triaxial accelerometers on stingrays in the Bahamas. Mar. Biol. 166, 53. https://doi.org/10.1007/s00227-019-3499-z.

Wood, S.N., 2011. Fast stable restricted maximum likelihood and marginal likelihood estimation of semiparametric generalized linear models. J. Roy. Stat. Soc. 73, 3–36.

Yaglioglu, D., Deniz, T., Gürleck, M., Ergüden, D., Turan, C., 2015. Elasmobranches bycatch in a bottom trawl fishery in the Iskenderun Bay, northeastern mediterranean. Cah. Biol. Mar. 56, 237–243.