

# Neglected fishery data sources as indicators of pre-industrial ecological properties of Mediterranean swordfish (*Xiphias gladius*, Xiphiidae)

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## Abstract

Management of fish populations and ecosystems suffers from data and knowledge gaps, particularly with respect to how humans and nature affect dynamics at multi-decadal and longer time scales. However, collection of new data which indicates population or ecosystem status is slow and expensive. Here we analyse c. 110 years of neglected fishery data for an overexploited top predator, swordfish, in the Mediterranean Sea. These data are available at scales of high time–space–biological resolution (i.e., sub-weekly, sub-regional sea; individual weights) and allow different ecological questions to be addressed than is possible with coarsely scaled data (e.g., annually resolved total catches aggregated over large sea areas). We constructed regional indicators of population status (relative abundance, mean individual size and its variability, migration phenology) covering most of the 20th century, and pre-dating other population datasets by 50–70 years. The length and duration of these new time series allowed detection of significant multi-annual/decadal variations in abundance and mean weight not detectable in shorter, more recent time series. These new data and evidence of multi-annual variability in population variables improve knowledge of Mediterranean swordfish ecology. The findings provide a new basis on which further historical data recovery and analysis of contemporary data can provide new perspectives and opportunities for quantifying vulnerability of populations to exploitation and climate change.

## KEYWORDS

fishery, historical ecology, Mediterranean, multi-decadal variability, population, swordfish

## 1 | INTRODUCTION

Fish population dynamics are driven by variations and trends in multiple forcings at multiple time scales. The forcings include natural and anthropogenic processes, including ocean–climate phenomena,

exploitation, eutrophication, and species interactions (e.g., predation, competition) (FAO, 2018a, 2018b; IPCC, 2019). Cumulatively, these processes drive variations in physiological and demographic rates, productivity, abundance, distribution, and ultimately the potential yields from commercial fisheries.

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However, for many exploited fish populations, indicators of biomass fluctuations cover a relatively recent and short period of exploitation history (i.e., last 2–4 decades; Caswell et al., 2020; Jackson et al., 2001; Pauly, 1995; Thurstan, 2022). Furthermore, datasets describing other indicators of variations in biological properties and life history (e.g., body sizes, growth rates, phenologies of spawning migrations and reproduction) of populations, are generally fewer and shorter (Asch, 2019; Jackson et al., 2001; Lotze et al., 2014; McClenachan et al., 2012). Consequently, the impacts of human activities and other ecological processes that occur in the sea at longer time scales, manifested as either trends, cycles, or threshold-like regime shifts, are not evident from most available short time series. Estimates of biomass and distribution of marine life, and how it has fluctuated over time based on recent, short time series, can therefore provide a limited perception of the status of populations and ecosystems (Caswell et al., 2020; MacKenzie & Ojaveer, 2018; Pauly, 1995; Thurstan, 2022).

Moreover, it is becoming clear that the productivity of fish stocks varies over multi-annual and multi-decadal time scales (Chang et al., 2020; Lindegren et al., 2018; Mildenerger et al., 2020; Tableau et al., 2019; Vert-pre et al., 2013), and co-varies with ecosystem forcing (Mantzouni et al., 2010; Minto et al., 2014). Detecting and quantifying these variations so they can be incorporated into models of population dynamics and stock assessment requires long time series covering periods of varying productivity (Peterman et al., 2003; Tableau et al., 2019). This requirement is increasingly being recognised for many fish stocks and species in the global ocean (ICES, 2022).

Here we consider multi-annual and multi-decadal scale variability in the population ecology of a charismatic, highly migratory top predator in the Mediterranean Sea, swordfish (*Xiphias gladius*, Xiphiidae). This species has high commercial value and has been exploited since Antiquity (Romeo et al., 2014). Its estimated biomass is at its lowest level since available time series started in 1950, with largest decline occurring during the 1970–1990s (Figure 1) (ICCAT, 2020c). A fisheries management plan to recover stock biomass has been developed and implemented (ICCAT, 2017a).

The assessment of stock status, as is typical for most stock assessments, is however based on limited and incomplete information (ICCAT, 2020c). For example, the underlying data used in the Mediterranean swordfish stock assessment are fishery-dependent catch and effort information, whose coverage by different countries has increased over time since the 1950s. Consequently perceptions of stock status and dynamics are more reliable later in the time series (i.e. especially after c. 1985) when both catch and fishing effort data are more widely available than earlier (ICCAT, 2020c). Furthermore, other biological data, such as sizes of captured individuals, growth rates and maturity information, which can increase the reliability of estimates of population dynamics and productivity, are limited; for example, the sizes of individuals caught by different gears or in different regions are available since 1972 or later (ICCAT, 2020c). When biological and fishery-related data are limited, simpler models requiring more assumptions of fishing activity and biological

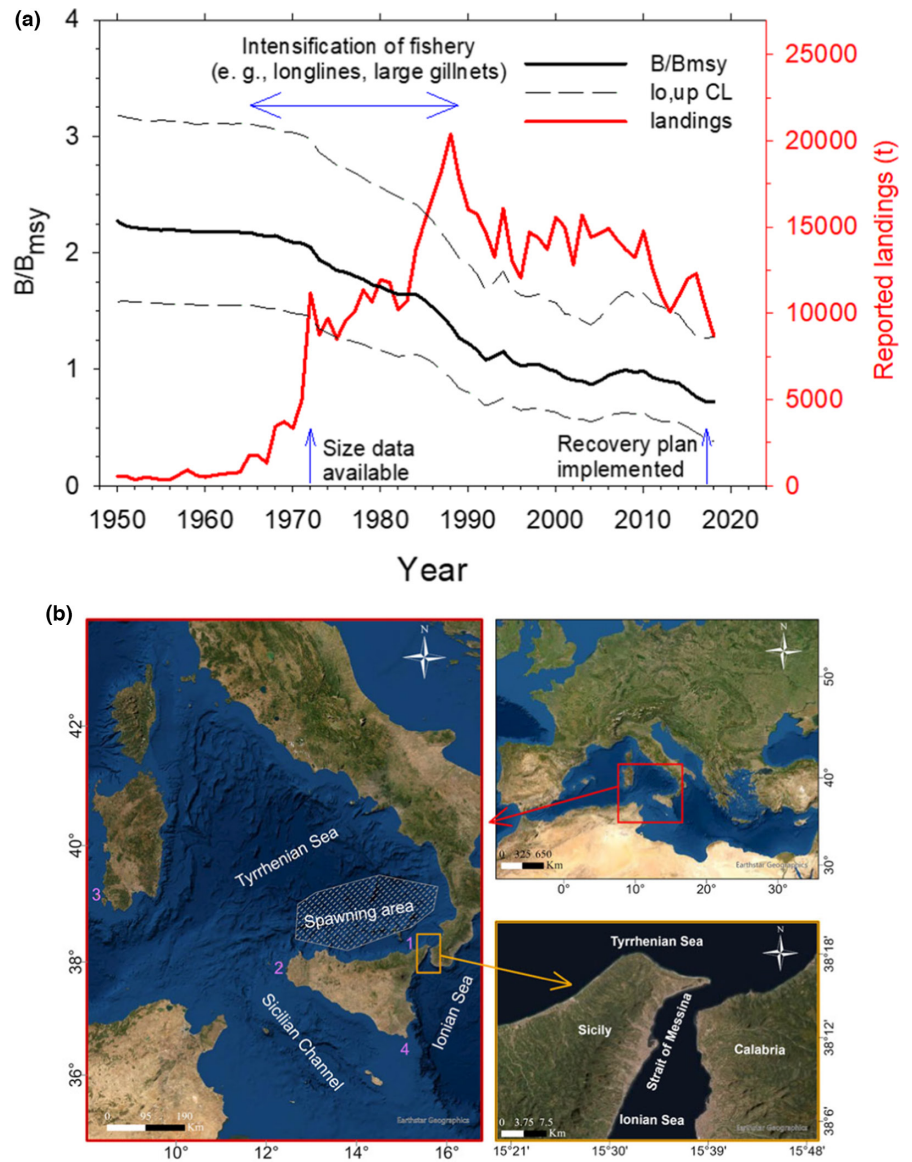
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properties (e.g., stable with no or random variability) are necessary (Chang et al., 2020; Free et al., 2020; Rosenberg et al., 2014). Outputs from such models can therefore become uncertain and biased relative to true population dynamics (Bouch et al., 2021; Free et al., 2020; Rosenberg et al., 2014), especially if fluctuations of past or future forcings vary non-randomly (e.g., have trends, cycles or regime shifts). This in turn affects perception of the magnitudes and frequencies of population variations and its vulnerability to human impacts.

Lastly, nearly all input data for the assessment of Mediterranean swordfish originate from commercial fisheries. In such cases, there is a risk that the outputs will be uncertain due to various fishery-related issues (e.g., incomplete seasonal and spatial coverage of population properties and distribution; technological advances in operational fishing processes which affect effort estimates) (Free et al., 2020).

In summary, various data limitations lead to uncertainty in estimates of the long-term dynamics of the Mediterranean swordfish population. This uncertainty is most evident for the period before 1950 when no indicators of either stock status or biology are available, but also during the assessment period from 1950 to the

**FIGURE 1** Temporal dynamics of fishery and population biomass indicators (ICCAT, 2020c), and map of locations (marked with numbers 1–4) of bluefin tuna traps used for data recovery for bycatches of swordfish, *Xiphias gladius*. (a) time series of estimated median stock biomass scaled to the biomass estimated to support sustainable yields in the long-term ( $B/B_{msy}$ ; solid black line) with upper and lower 95% confidence limits (dashed black lines) and reported landings (red line) for 1950–2018 (ICCAT, 2020c). The panel also shows when fishing intensified with implementation of higher effort and new technologies, size data became available for population assessments and a recovery plan was implemented (ICCAT, 2017a, 2020c). (b) Numbers 1–4 on map at left refer to the following sites where swordfish bycatch data were recovered from trap records: Milazzo, Favignana + Formica, Portosucuso and Capo Passero, respectively. The cross-hatched area approximates the location of the spawning area in the southeastern Tyrrhenian Sea. Inset maps show the wider geographic area and the narrow Strait of Messina (between Sicily and the Italian mainland) through which many swordfish migrate towards the spawning area and where historical swordfish fisheries operate (Romeo et al., 2015b)



mid-1980s when data coverage is still insufficient for construction of more data-rich models (Mantopoulou-Palouka & Tserpes, 2020; Winker et al., 2020). Notably this period (1950s- mid-1980s) is when the fishery was intensifying due to increased effort and deployment of new gear types (e.g., longlines, large gillnets; ICCAT, 2020c). There is a need for new datasets and knowledge of the historical development of Mediterranean swordfish fisheries, and particularly from countries such as Italy, which have large catch shares of available quotas (ICCAT, 2017b, 2020c).

We have therefore investigated sources of swordfish data other than those from targeted fisheries, which could provide a longer perspective to population dynamics. We have found that one of the longest and consistently applied fishing gear types in the Mediterranean (i.e., the traditional coastal trap for exploitation of migrating Atlantic bluefin tuna (*Thunnus thynnus*, Scombridae; Addis et al., 2012; Ambrosio & Xandri, 2015; Fromentin, 2009; ICCAT, 2012) can be a source of detailed and temporally resolved historical swordfish data (MacKenzie et al., 2021). Some of these traps, including four whose catch records we have investigated, are located

close to, or along migratory routes towards, a major spawning area for swordfish in the Mediterranean Sea (southeastern Tyrrhenian Sea; Cavallaro et al., 1991; Romeo et al., 2009, 2011). Their relative proximity to a key habitat for the species life history increases the probability that some individuals could be captured during migration to or from this spawning area. Our recovery and evaluation of the trap records and accounts for these traps demonstrated that the reports and accounts contained bycatch data in numbers and or weights during many years from 1896 to 2010. Although some other studies have reported bycatch of swordfish in tuna traps (Ambrosio & Xandri, 2015; Britten et al., 2014; Cetti, 1777; ICCAT, 2017b), such bycatches have so far received little scientific attention or quantitative analysis.

Here we investigate the temporal-spatial variability in the bycatch data, which we have recovered. We hypothesise that, as with some other studies employing bycatch data to investigate marine population ecology and biodiversity (Ferretti et al., 2008; McClenachan et al., 2012; Wang et al., 2020), these data can potentially provide new knowledge of the long-term dynamics of the

ecology of this species in the Mediterranean Sea, including whether variations in time and space are random or contain significant patterns. New long data sets such as these could provide new baselines for analyses of past and future effects of human impacts such as exploitation and climate variability/change on swordfish ecology in this region. They could also provide a basis for developing new long-term indicators of population status and its role in the food web (Coll et al., 2016; Romeo et al., 2015b).

## 2 | MATERIALS AND METHODS

We have analysed recently recovered and compiled bycatch data from four bluefin tuna trap locations in Italy. Full details of the data sources, data variables available for recovery, choice of locations and the trap design (sizes, mesh sizes, depth of deployment) are provided in a related data paper (MacKenzie et al., 2021). The raw data have been deposited at Pangaea.de where they are available to the scientific community. Here we provide a brief description of the data and how the fishery which provided the data operated.

The traps used in Italy are similar to those deployed in other Mediterranean and east Atlantic tuna trap fisheries (Addis et al., 2008; Ambrosio & Xandri, 2015; Ravier & Fromentin, 2001). The design of the traps (sizes and numbers of nets, mesh sizes, etc.) and method of deployment have undergone few or no change throughout the time period (Addis et al., 2008; Ambrosio & Xandri, 2015; Ravier & Fromentin, 2001). They are typically deployed in shallow coastal waters where they intercept migrating schools of bluefin tuna. The traps that we have considered are c. 1.5–2 km long and were anchored at 30–40 m depth within a few km of the shore (MacKenzie et al., 2021). They are put in place in spring or mid-summer for 2–4 months and typically emptied 2–3 times/week (detailed deployment times are presented in Results). Because of the stable technology, the traps can be considered to be a standardised sampler within sites (i.e., over time) and among sites (Addis et al., 2012; Ambrosio & Xandri, 2015). Details of the sizes, mesh sizes and deployment locations considered in this investigation are summarised elsewhere (MacKenzie et al., 2021).

The locations of the traps whose data we recovered and analysed were three sites on Sicily (Milazzo, Favignana and Formica [hereafter referred to as F + F], Capo Passero) and one site on Sardinia (Portoscuso); the locations are situated along swordfish migration routes, as demonstrated by the presence of a well-developed modern pelagic fishery targeting swordfish in these areas (Di Natale et al., 2005; Mariani et al., 2015; Romeo et al., 2015a). Traps were deployed seasonally at times to capture migrating bluefin tuna; however, both bluefin tuna and swordfish spawn at similar times in this region (Mather et al., 1995; Romeo et al., 2009), thereby increasing the possibility that the traps also could catch migrating swordfish. Among these tuna traps, Portoscuso is the only one regularly active today (Addis et al., 2012).

The bycatch data were recovered from various historical documents associated with the traps (e.g., reports, archives and

company accounts for the traps), and consisted of the dates of capture, numbers and whole (i.e., non-gutted) weights of captured swordfish (usually group weights on a given fishing day, but some locations also had individual weights), and the start and end dates of the seasonal operation of the traps. Traps were typically deployed for a few months during spring–summer, and emptied 1–3 times per week.

The F + F trap locations are separated by c. 10 km. Trap records for these two sites contained separate catch data in numbers for many but not all years. During the 1970s, the records (in numbers and weight) are combined for the two locations and it is not possible to assign the catches to the specific locations. During the entire time period (1931–2002), the two traps were nearly always checked or emptied on separate days (i.e., swordfish were reported to have been caught on the same day at both traps on fewer than 10 days). Because of (i) the close proximity of these traps, (ii) the absence of trap-specific catch information for several years and (iii) the alternation of days when the traps were emptied, we consider the two traps to represent samplers at the same location and effectively only 1 trap. For our analyses, we therefore used the combined numbers (or weights) caught at both sites and considered the records as a single time series representative for catches at this location.

### 2.1 | Catch and effort data

We summed the numbers and weights of swordfish caught during each season to calculate the total annual catch in numbers and kilogram per location. In most locations and for most years, the date of first and last gear deployments were available; we considered that the interval between these dates was an approximate indicator of annual fishing effort at each trap site. We evaluated whether length of fishing season influenced the numbers of swordfish caught within each location and across locations throughout the time period of our study (1896–2010). These comparisons used scatterplots, linear regression and analysis of covariance (ANCOVA) analyses, and showed that the numbers of swordfish caught per year tended to be significantly associated with fishing effort (details in [Supplementary Material](#)). As a consequence, we calculated an annual Catch Per Unit Effort (CPUE) index as the number of individuals caught per day at each trap location during each fishing season and used this time series when comparing catch levels over time within sites and across sites in subsequent analyses. The CPUE index for each year and site was calculated as the sum of swordfish caught per year divided by the number of days when the trap was deployed (“fishing days”) per year.

Preliminary visual inspection of both the raw numbers caught per season and the CPUE data showed that these data tended to be skewed as is common for fishery catch data. We transformed these data using natural logarithms to stabilise variances and improve normality of distributions before further statistical analyses.

We used one-way ANOVA with minimum significance level  $p = .05$  to evaluate and compare the response variables among



the four sites. The variables included the overall levels of fishing effort (i.e., number of fishing days per year), numbers of swordfish caught, ln CPUE and mean weight and phenology metrics (descriptions below). ANOVA assumes homogeneity of variances among groups. We checked for homogeneous variances using Levene's test (Levene, 1960; SAS, 2016), and if variances were heterogeneous, applied an ANOVA method which accommodates variance heterogeneity (SAS, 2016; Welch, 1951). For all response variables analysed, highly significant ( $p < .001$ ) group differences were detected assuming homogeneous variances (details below); for variables having heterogeneous variance among groups, differences were still highly significant ( $p < .001$ ) when compared using heterogeneous variance tests. Because ANOVA is robust to modest deviations from homogeneous variance (Box, 1953), and our results were similar when analysed with both methods, we only present detailed results for tests assuming homogeneous variances.

As we were particularly interested in the long-term temporal development of CPUE, we analysed the time series available within each site for evidence of both overall long-term trends and multi-annual variations. We used General Additive Modelling (GAM; Wood, 2006) as implemented in the Proc GAMPL procedure (SAS, 2016) of SAS version 9.4 to evaluate temporal variability in the lnCPUE data within each site. The number of degrees of freedom for the overall fit of the GAM was restricted to a modest level (i.e., total of 5, including 3 for the spline fit, 1 for the intercept and 1 for a variance dispersion parameter; SAS, 2016). A modest number of degrees of freedom facilitates fitting of nonlinear patterns in the data and interpreting the results while reducing the possibility of overfitting the model to the data (Lindgren et al., 2018; Zucchetto et al., 2010). Model fitting used a normal distribution with an identify link function (SAS, 2016; Wood, 2006) (Wood, 2006; SAS, 2016). Goodness-of-fit was assessed using information criteria (AIC) (Burnham et al., 2011). For analyses involving ln CPUE where sample sizes were relatively small, we used the AIC<sub>c</sub> criterion which accounts for small samples sizes (Burnham et al., 2011). We compared AIC and AIC<sub>c</sub> values for models including the fitted spline effects with null models (i.e., the overall respective time series mean), calculated the difference in AIC or AIC<sub>c</sub> for the spline fit and null models, and estimated the information criterion-based evidence ratio in favour of the best-fitting model (Burnham et al., 2011). We derived 95% confidence limits for fitted trends and displayed the results together with the raw data in time series plots.

As most of the swordfish caught in this study are believed to be individuals migrating to or from a major spawning area (southern Tyrrhenian Sea), we conducted a regional, multi-site random walk analysis of the temporal variability in ln CPUE. The objective of this analysis was to identify whether regional trends in abundance were evident across sites, and to describe the relative size of the population by combining the observations from the four sites where CPUE data are available. This analysis assumes that in combination the

observations from the four sites are proportional to the development of the size of the population. However, the relative CPUEs between the four sites are allowed to vary. The observations of CPUE are assumed to follow a log-normal distribution, whose mean is described by the population size and the proportion of the population seen at each site. As the variability of observations was different across sites, the model allows different levels of standard deviation between sites. The observation model is summarised by:

$$\ln(\text{CPUE}_{y,s}) \sim \mathcal{N}(\ln(N_y p_{y,s}), \sigma_s^2), \quad y = 1896, \dots, 2010, \quad s = 1, \dots, 4 \quad (1)$$

Here  $N_y$  is the relative population size in year  $y$ ,  $p_{y,s}$  is the fraction of the population at site  $s$  relative to that at all four sites combined and  $\sigma_s^2$  is the observation variance at site  $s$ . The development of the population size  $N_y$  is described by a random walk process at the logarithm scale:

$$\ln N_{y+1} = \ln N_y + \epsilon_y, \quad \text{where } \epsilon_y \sim \mathcal{N}(0, \sigma_N^2) \quad (2)$$

$\sigma_N$  is a model parameter describing the distribution of the yearly changes in the population size. The fraction at each site  $p_{y,s}$  is also allowed to develop over time. The four-dimensional process  $(p_{y,i})_{y=1896, \dots, 2010}$  must have each  $p_{y,s}$  between 0 and 1 and  $p_{y,1} + \dots + p_{y,4} = 1$ . This was achieved by using the additive logistic transformation. First a three-dimensional random walk process was set up for  $\alpha_y$ , such that:

$$\alpha_{y+1,i} = \alpha_{y,i} + \eta_{y,i}, \quad \text{where } \eta_{y,i} \sim \mathcal{N}(0, \sigma_\alpha^2) \quad \text{for } i = 1, 2, 3 \quad (3)$$

and  $\sigma_\alpha$  is the model parameter describing the variation over time in the fraction. The logistic transformation then defines:

$$p_{y,i} = \begin{cases} \frac{e^{\alpha_i}}{1 + e^{\alpha_1} + e^{\alpha_2} + e^{\alpha_3}} & \text{for } i = 1, 2, 3 \\ 1 - p_{y,1} - p_{y,2} - p_{y,3} & \text{for } i = 4 \end{cases} \quad (4)$$

This defines the model. Notice that the random processes ( $\alpha$  and  $\ln N$ ) are defined even if the corresponding observations are missing for some sites in some years. All model parameters are estimated by optimising the marginal likelihood function, where the random processes ( $\alpha$  and  $\ln N$ ) are integrated over the Laplace approximation.

We evaluated the null hypothesis that the fitted ln CPUE from the model was constant over time by checking whether  $\sigma_N^2$  differed significantly from 0. This test identifies whether significant temporal variations in ln CPUE were present. Evidence of significant temporal variations was also assessed using AIC by comparing AIC from the fitted model with null models in which there were no differences in ln CPUE among sites (i.e., mean ln CPUE was the overall mean among sites) and in which ln CPUE differed among sites (i.e., a model in which ln CPUE differed between sites, but not over time).

## 2.2 | Weight data—derivation and analysis of time series

Weight data were available for most traps but at various levels of resolution and for differing numbers of years. In one trap (Portosucoso), each individually captured swordfish was weighed (total weight in kg). In other locations (F + F, Capo Passero), only group weights with numbers caught for each day were available; these data permitted estimation of a daily mean weight. In another site (Milazzo), group weights and group numbers were available by month, and only for a small number of years (1896–1901); no weight data at any level of aggregation was available after 1901 for this site. Unless otherwise stated, weights in this report refer to whole (ungutted) weights, also known as round weights.

For sites (F + F, Capo Passero) reporting catches as group weights and numbers on each trap-emptying day, we calculated the mean weight of swordfish caught on that day (hereafter referred to as daily means). We used GAM to evaluate temporal variability across years as was done with the In CPUE data. We then derived annual mean weights from these daily means and plotted the time series of annual mean weights for visual inspection and statistical analysis (GAM) of long-term trends and multi-annual variations.

At one site (Portosucoso), swordfish were weighed individually. We aggregated these data to produce daily and annual means to match the resolution of the weight data from the other two traps and conducted GAM analyses on these data. Moreover, as shown in Results, on many days during each year, single individuals of swordfish were captured at the other two sites (F + F, Capo Passero). We therefore used the subset of single-swordfish capture days at F + F and Capo Passero, and the individually measured swordfish at Portosucoso to produce separate time series of weights based on individual swordfish. We then conducted GAM analyses on these individually resolved time series to quantify temporal variability across years. As the distribution of weights within a site and year was often skewed, all weights were common log (base 10) transformed to stabilise variances and improve normality of distributions prior to calculation of mean weights and further statistical analysis.

To estimate the age composition of the catches, we used conversion factors and weight-length-age relationships employed by the ICCAT Mediterranean swordfish assessment working group (ICCAT, 2020b, 2020c). Whole weights (also known as round weights, RW) were first converted to gutted-gilled weights (GGW;  $RW = 1.14 * GGW$ ; (ICCAT, 2020b; Tserpes et al., 2017), which were then used to estimate lower jaw fork length (LJFL), using the relationship derived by Tserpes et al. (2017) (ICCAT, 2020b) for Mediterranean swordfish. We then applied these lengths in a Mediterranean combined-sex length-at-age relationship (Tserpes & Tsimenides, 1995) to estimate ages.

## 2.3 | Weight data—comparisons among sites and with other non-trap datasets

We analysed our weight data for long-term spatial differences in mean weight among the sites using one-way ANOVA as described

above. The weights used in this comparison were site-specific long-term means. These site-specific means and variances were derived by calculating the overall mean weight and variance based on the annual means within each site. Where differences between sites were significant ( $p < .05$ ), we conducted a posthoc Student-Newman-Keuls (SNK) multiple comparison test to identify which locations differed from others.

We also compared via an additional ANOVA the site-specific long-term mean weights from the traps with other sources of commercially derived swordfish weight data to evaluate whether there were differences. These sources include fishery data for three individual harpoon vessels from the Strait of Messina whose captains kindly provided us their private logbooks from which we could derive the numbers and mean weights of swordfish captured for specific multi-annual time periods of their operations. In addition, we calculated the annual mean weight of swordfish captured by the entire harpoon fleet operating in the Strait of Messina for the years 2002–2015 (Battaglia et al., 2018). All of these harpoon vessels target swordfish on their way to and from the spawning aggregation area in the Tyrrhenian Sea and near the Milazzo tuna trap. These vessels therefore probably capture swordfish from the same spawning aggregation as the traps used in our study (Perzia et al., 2016). Lastly, we included the mean weight of all swordfish reported caught in the Mediterranean from all commercial fishing operations and as compiled by ICCAT; these data are available for the years 1972–2018 (ICCAT, 2020c). These analyses involving other gears and time periods provide a wider comparative context for the trap-derived data.

## 2.4 | Estimation of annual and long-term average catch phenologies

The data extraction and compilation process showed that the catch data were highly temporally resolved within years. This resolution permits estimation of the annual catch phenology at each site which can provide insight into dynamics of migration, habitat use and fishing activity. For each year at each site, we calculated the relative cumulative catch throughout the year, expressed as a percentage of a given year's total catch. For this analysis, we used years having at least 15 swordfish captures in calculation of annual phenologies; years having fewer catches were assumed to contain too few data for resolution of phenologies. The seasonal cumulative catch curves represent year-specific catch phenologies for each site. Given this phenology, we extracted the Julian days when the 10th, 50th and 90th percentiles of the annual phenology occurred. We assumed that these percentiles approximate the initiation, midpoint and end of the migration of swordfish past the regions where the traps were located. We then calculated the long-term mean and standard errors of the annual initiation, midpoint and end of the migration for each site. We compared these migration thresholds among sites using one-way ANOVA, and where differences occurred ( $p < .05$ ), conducted post-hoc SNK multiple comparison test to identify which sites differed.

We then calculated the long-term average phenologies of the catches at each site by aggregating all catches on each trap-emptying day across years chronologically by Julian day during the year. Then we calculated the cumulative catch throughout the year for the entire time period to derive a long-term average seasonal phenology at each site.

Fish migration phenologies are often size-dependent (Jansen & Gislason, 2011; Tamario et al., 2019). This possibility can be investigated using the individually weighed swordfish data available from F + F and PS where 100s of individual swordfish have been weighed (see Results). We evaluated whether there were significant seasonal trends in log weight of captured individual swordfish during the year. These analyses were conducted using the aggregated dataset of individual weights across years because samples sizes in individual years were generally too low for meaningful analysis. We also plotted the log weight vs. Julian date to inspect visually the seasonal development of weight for appearance of unusually large or small individuals.

### 3 | RESULTS

#### 3.1 | General overview of recovered data

Time series of catches, effort and size (weight) were recovered from four sites. Time series for a given variable from the four sites differed in duration, timing and frequency, and all had multiple gaps lasting from 1 to >20 years (Table 1; details below). The earliest time series began in 1896 (Milazzo) and the most recent data were from 1992 to 2010 except for a gap from 1998 to 2002 (Portoscuso). The longest time series was from F + F containing 55 years from 1931 to 202 (gap from 1985 to 1999). No single year is represented by all four sites and there were only 10 common years when three sites (Milazzo, F + F, Capo Passero) were represented. The greatest temporal overlap between pairs of time series was for Milazzo and F + F (23 years).

The number of swordfish caught per season was generally highest at Milazzo, followed by F + F, Capo Passero and Portoscuso, with the Milazzo catches being significantly higher than those at other sites (Table 1 and Figure 2; one-way ANOVA, with SNK multiple comparison test:  $p < .05$ ). Significant amounts of the variability in

catches among years within and among sites were due to duration of the fishing season as an approximate indicator of fishing effort (Figure S1; Supplementary Material).

#### 3.2 | Time-space variations in ln CPUE

Effort-standardised catches (i.e., ln CPUE) differed significantly among most sites (Figure 2; one-way ANOVA,  $p < .0001$ ;  $R^2 = .37$ ). Milazzo had the highest mean ln CPUE, followed by Portoscuso whose ln CPUE was significantly lower; these sites had significantly higher mean ln CPUE than both F + F and Capo Passero, whose means were not significantly different (Figure 2). On average, the median daily catch rate at Milazzo was c. 1 swordfish per day throughout the fishing season, whereas the median catch rate at Capo Passero and the other two sites was c. 0.5 and 0.4 swordfish per day, respectively.

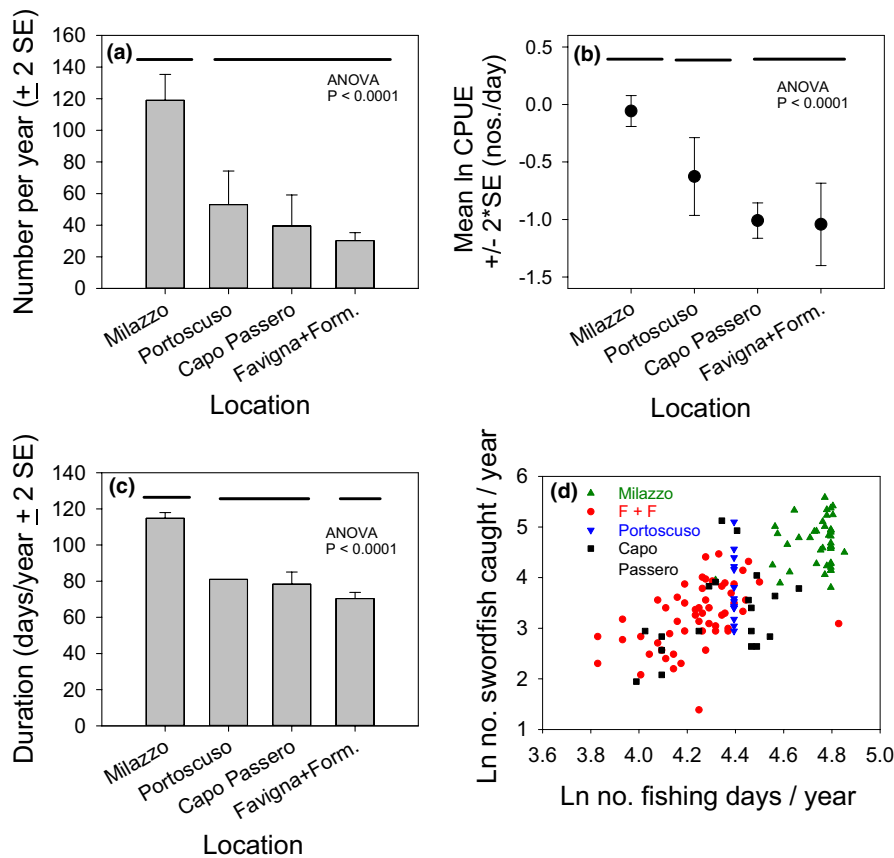
There were relatively few years in common for ln CPUE data among the sites (Figure 3). For example, the Portoscuso data were from a recent time period (1992–2010), and was almost completely isolated in time from the other sites. However, among the other three sites there were sufficient number of years in common to allow evaluation of potential spatial co-variations.

However, there was some spatial co-variability in the ln CPUE among sites. The highest correlation ( $R^2 = .13$ ;  $p = .05$ ) was between Milazzo and F + F where 23 years were in common. Correlations among other pairs of sites were less significant, possibly due to the smaller number of years in common (number of common years for comparisons were 12 and 15).

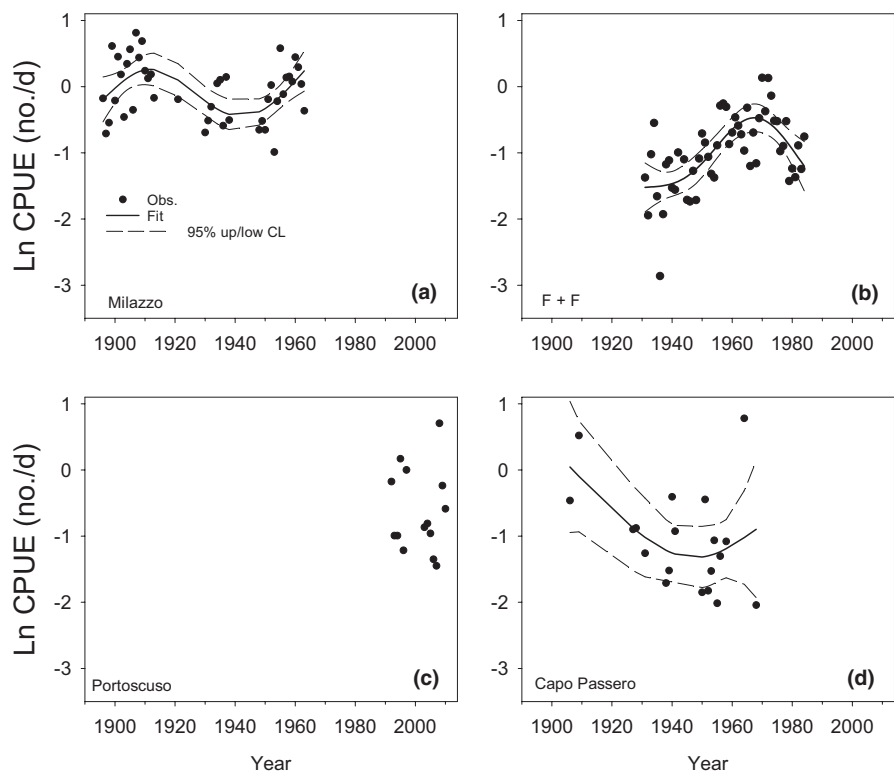
There were significant inter- and multi-annual variations in ln CPUE within some of the sites as estimated by GAM and as revealed both by statistical significance tests and information-based criteria (Figure 3, Table S1). At Milazzo, whose ln CPUE was the largest among the four sites (Figure 2), the fitted spline for the year effect was significant ( $p < .0001$ ; 3 df), and the spline model had an  $AIC_c$  value c. 6 smaller than a model based on the overall mean (null model). This means that the spline model is c. 20 times more likely to represent the temporal variations than the null model; explained deviance of the GAM was 27%. The main temporal variations were a long decline between c. 1910 and 1950, after which ln CPUE

TABLE 1 Meta-data summary of swordfish bycatch and size data recovered from four bluefin traps in Italy

Characteristic	Location				Total
	Milazzo, Sicily	Favignana & Formica Sicily	Portoscuso, Sardinia	Capo Passero, Sicily	
Total number of swordfish caught	5119	1670	744	652	8185
Number of individually weighed swordfish	0	706	815	71	1592
Time period of data available (including missing years)	1896–1963	1931–2002	1992–2010	1906–1968	1896–2010
Number of years with data available	43	55	14	19	114
Number of daily catch records (nos. and group or individual weights)	4706	1016	743	140	6605



**FIGURE 2** Fishing effort and catch data for swordfish caught as bycatch in four bluefin tuna traps in Italy 1896–2010. Panels a, b, c: mean and standard errors of numbers of swordfish caught, ln catch per unit effort (CPUE), and duration of fishing season in days across years at four trap locations in Italy



**FIGURE 3** Interannual variability in ln CPUE (numbers caught per day) for bycaught swordfish in four bluefin tuna traps in Italy. Trap locations shown on panels (see also Figure 1). The fitted trend lines and 95% confidence limits are from GAM analyses. See text and Table S1 for details of methods and statistical results

increased until the end of this series in 1963 (Figure 3). The increase during this period corresponded to an approximate doubling in median CPUE. The Portoscuso time series was too short and variable to produce evidence of significant temporal variations ( $p > .05$  for spline year effect).

The longest time series (1931–2002, from F + F) showed significant multi-annual variations (Figure 3, Table S1). The spline fit for the year effect was significant ( $p < .0001$ ; 3 df; explained deviance = 0.42), and its  $AIC_c$  value was 23 units smaller than the null model (Table S2). The difference in  $AIC_c$  values indicates that there



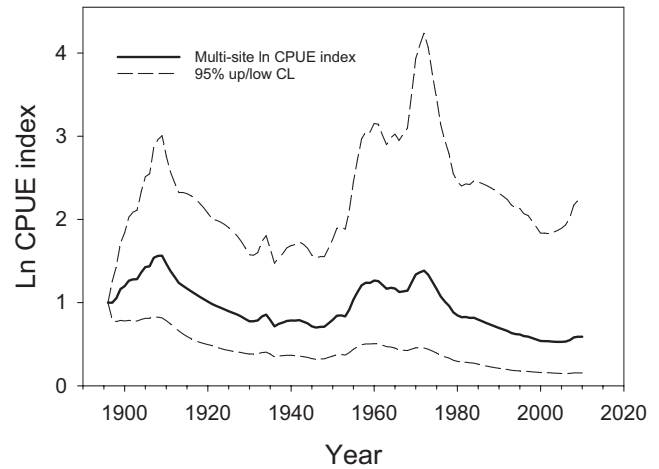
is c. 107,000 times more evidence to support the spline fit model than the null model. The time series has a long gap between the final 2 years in the time series (2000, 2002), and the next earliest year (1984). We re-ran the GAM analysis using data only until 1984, and this analysis also yielded a significant year effect ( $p < .0001$ ; explained deviance = 0.45); the difference in  $AIC_c$  values was 24 corresponding to an evidence ratio of c. 192,000 in favour of the spline fit model (Table S1). The main temporal variations from this analysis were a relatively low, stable level of ln CPUE during the 1930s, after which ln CPUE increased until the late 1960s. ln CPUE then declined from c. 1970 until the end of the series in 1984. The two late years (2000, 2002) omitted from this analysis had ln CPUE values similar to or lower than those in the early 1980s.

Time trends at Capo Passero were also significant ( $p = .0068$ ) and explained 24% of the time series deviance (Figure 3, Table S1). However, the model had relatively weak evidence compared with the null model, as seen by the  $p$  value and also based on delta  $AIC_c$  criterion. Similarly, at Portosucoso, the evidence for significant temporal variations was weak ( $p = .1$ ; evidence ratio = 0 in favour of a spline fit). The lack of stronger evidence for significant temporal variations at these two sites may be partly due to the relatively small number of years of data available.

When data from all sites and years were combined into a single analysis, which accounted for differences in data coverage and possible temporal variations, significant temporal fluctuations in ln CPUE were evident ( $p = .004$ ; Figure 4). This model is much more likely to represent the temporal variability than null models based either on the overall mean ln CPUE of the dataset or on site-specific mean ln CPUEs (delta  $AIC = 22.5$  and  $81.6$  respectively, corresponding to evidence ratios exceeding  $10^4$  and  $10^{17}$ ). The main temporal variations identified by the analysis were a decline from the late 1800s to early 1900s until the 1940s–1950s, and an increase during the late 1950s–early 1970s to CPUE levels seen in the early 1900s. During and after the 1970s, a long decline occurred until the end of the time series in early 2010s. The amplitude of the decline in ln CPUE since the 1960s–mid-1970s until the late 1990s–2000s was c. 50%. Collectively, these multi-annual variations were significant, although there was no significant overall decline between the late 1800s and 2000s. AIC comparison of these models also demonstrated that different sites had different mean ln CPUEs, even after accounting for temporal variations (delta  $AIC = 59.1$  and evidence ratio  $>10^{12}$ ). In general, and as seen by single-site analyses using one-way ANOVA (see above), ln CPUE tended to be higher at Milazzo and Portosucoso than at F + F and Capo Passero. This is also evident in time series plots of site-specific standardised ln CPUE indices (Figure S2; Supplementary Material).

### 3.3 | Weight analyses—general data availability

Weight data were available in a fragmented and inconsistent manner both within and among sites. At Milazzo, group weights and numbers caught were available by month for 1896–1901. These were used to estimate mean weight by month for those years. At F + F, PP and



**FIGURE 4** Long-term inter-annual variability in a multi-site index (with 95% confidence limits) of swordfish abundance the Tyrrhenian Sea-Strait of Messina area during 1896–2010. The index is based on a random walk time series analysis of swordfish bycatches in four bluefin tuna traps in Italy. See text for methodological details

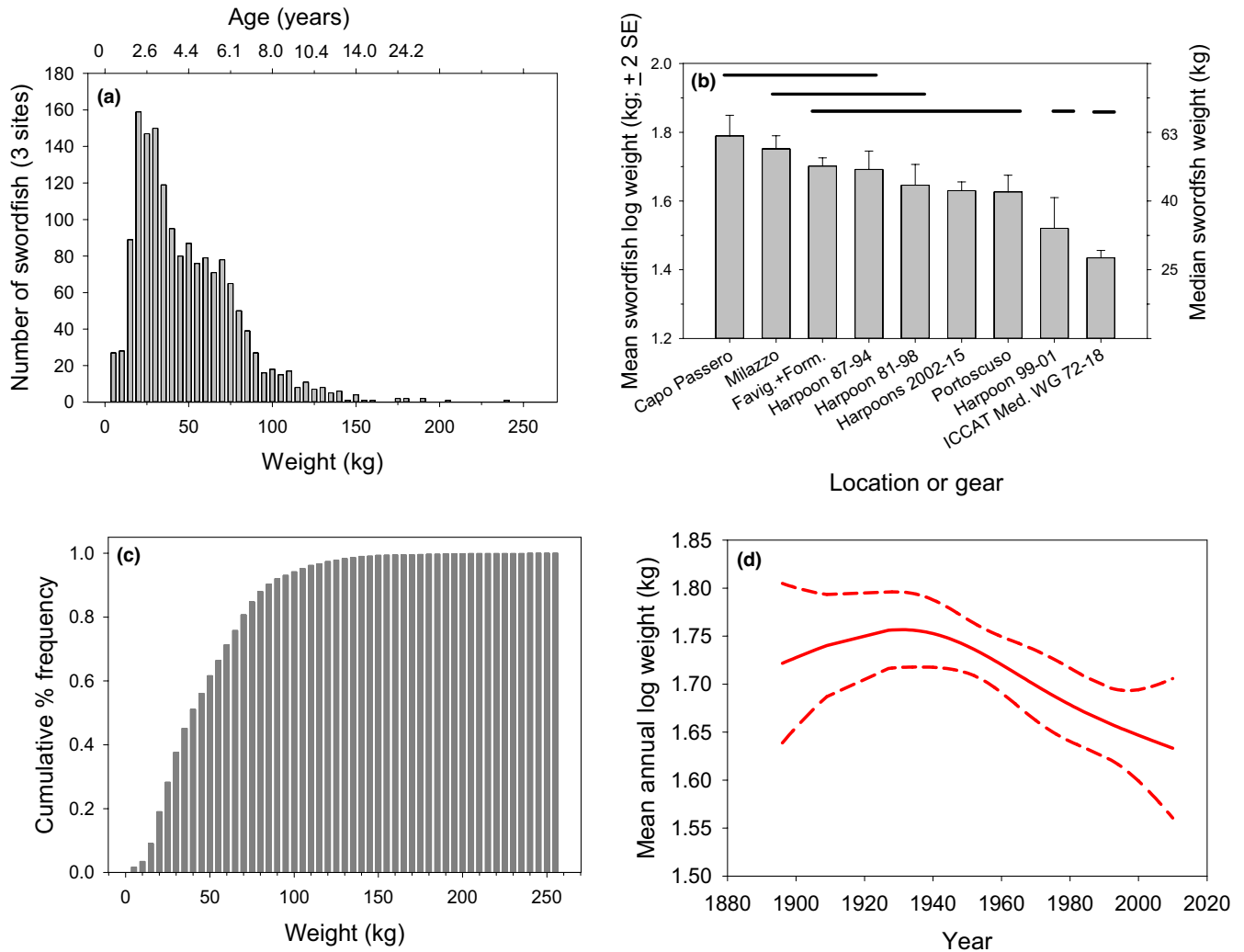
CP, group weights were available together with the numbers caught on each day the trap was emptied. By combining the group weights and catch numbers for each day, it was possible to derive a mean individual weight on different days throughout the fishing season at these three sites.

However, on most days, only single swordfish were caught. As a result, on these days, it was possible to analyse individual-level weights. Such data were available for all three sites where daily weight data were recorded. The number of individually weighed swordfish at each site was 815, 706 and 71 (total = 1592) at Portosucoso, F + F and Capo Passero, respectively.

Frequency distributions of the raw individual weight data were skewed but generally peaked in the size range 20–40 kg (Figure 5, Figure S3). The overall mean log weight was 1.6 (SD = 0.3;  $N = 1592$ ), corresponding to a geometric mean of 40 kg (back-transformed SD range = 20–79 kg); the smallest and largest individuals caught were 3 and 237 kg, respectively). Given weight-length and length-at-age relationships from literature (see Methods), most of the individually caught swordfish captured were aged 2–4 years (Figure 5).

One-way ANOVA showed that there were significant differences ( $p < .0001$ ) among the mean sizes of swordfish caught in different traps and with other fishing gears (Figure 5). Overall, the different trap locations and gear types explained 66% of the variation in the long-term mean weights among groups.

Among the traps, there was generally much overlap in mean sizes with only weak evidence of significant differences. The largest were caught at Capo Passero and Milazzo (geometric means 56–62 kg). The trap having smallest geometric mean weight was Portosucoso (42 kg); swordfish caught here were significantly ( $p < .05$ ) smaller than those caught at Capo Passero and Milazzo, but not from those caught at F + F (Figure 5). When compared with other fishing methods, the smallest mean sizes were those caught by the aggregate commercial fishery in the entire Mediterranean as compiled by



**FIGURE 5** Weight data for swordfish caught as bycatch in Italian bluefin tuna traps during 1896–2010. (a) frequency distribution of 1592 individually weighed and age-estimated swordfish caught at three sites (F + F, Portoscuso, Capo Passero) (histograms for each site are available in Figure S3). The ages were estimated from the individual weights and length–weight and length–age relationships used in the Mediterranean swordfish stock assessment and literature. For reference, swordfish in the Mediterranean are assumed to reach maturity at age 3 (i.e. 50% probability of having reached maturity at age 3; ICCAT, 2020c). (b) bar chart of mean weights (with 2 standard errors) of swordfish caught in four bluefin tuna traps and by different gear types and time periods in the Mediterranean Sea (mean weights displayed here are means of annual means). Trap locations are given in Figure 1. Data sources are given in main text. Horizontal lines at top of panel show group means which do not differ according to a SNK multiple comparison test, following one-way ANOVA. (c) Cumulative relative weight composition of swordfish caught as bycatch in three bluefin tuna traps in Italy during 1927–2010. (d) Long-term trend in log mean annual weight of swordfish caught as bycatch in four Italian bluefin tuna traps from 1896–2010. The trend and 95% confidence limits (solid and dashed lines, respectively) are fitted with GAM. Statistical details available in Table S2. See Figure S4 for a version of this figure panel which includes observations

ICCAT (geometric mean = 25 kg), followed by one of the harpoon vessels. These groups differed significantly from each other and from all other gear types and locations.

### 3.3.1 | Site-specific temporal variability in mean annual weights across years

Inter-annual datasets from three sites (F + F, Capo Passero and Portoscuso) were sufficiently long for GAM analysis of temporal variations in mean annual log weights. Milazzo had annual mean

weights for 6 years, which was considered too few for GAM analysis. At F + F, whose time series was the longest, GAM identified multi-annual and long-term variations (Figure S4, Table S2). Geometric mean annual weights initially increased from c. 40 kg in the 1930s to about 55 kg during the early late 1950s–early 1960s; weights then declined over the following decades to the present to c. 40 kg. Some individual years during the 1970s reached historic low values of c. 25 kg, which were never reached before throughout the long history of traps recovered here (Figure S4).

Mean annual weights at Portoscuso also showed significant temporal variations. Here, weights were relatively low in the mid-late

1990s and increased during the 2000s (Table S2; Figure S4). The initial weights (i.e., c. 40 kg during the 1990s) were similar to the mean annual weights at the end of the F + F time series. Mean annual weights at Capo Passero were some of the largest at all sites and years, but varied relatively little throughout the time period (Figure S3; Table S2). Mean weights during the 6 years for which data were available at Milazzo were c. 56 kg and therefore similar to mean weights a few years later (at Capo Passero in 1906 and 1909) and during most of the period for which data were available at F + F until the decline in the 1970s–early 1980s (Figure S4).

Given that there was only weak evidence of differences in mean log weights between sites (Figure 5), we conducted a GAM using all data from the four sites to investigate the general temporal variability in the entire region. This analysis also showed that geometric mean weights started at c. 56 kg in the late 1890s–early 1900s and had minor fluctuations until c. 1950, after which mean weights gradually decreased to a minimum (c. 45 kg) in the 2000s–2010s (Figure 5; Figure S4). The decline from initial weight to the minimum represents a decline of 20%. The fitted time trend explained more variance than a simple mean (evidence ratio = 170 and explained deviance = 11%,  $p < .0001$ ; Table S2).

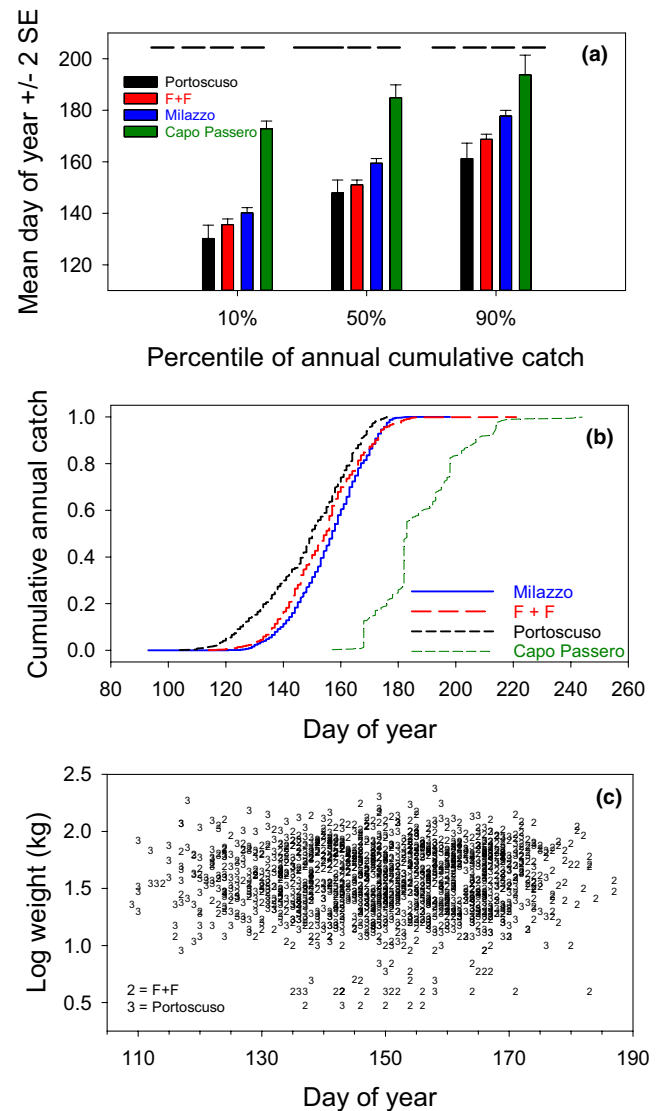
We repeated the temporal analyses using weight data at different levels of aggregation scales (i.e., using daily- or monthly group means in time series analyses; using individual weights). In general, analyses at these scales showed similar multi-annual and –decadal scale variations as when using weight data aggregated on a yearly basis (details in Supplementary Material).

### 3.4 | Catch phenologies

Annual catch phenologies differed significantly among the sites for each migration threshold indicator (arrival, midpoint, end; ANOVA:  $p < .001$  for each percentile; Figure 6 upper panel). Site explained 79%, 73% and 52% of the variance among sites for the 10th, 50th and 90th percentiles, respectively. In general, swordfish were caught earliest at Portoscuso, arriving here on average c. 7 and 10 days earlier than at F + F and Milazzo, respectively.

This sequence of catch phenology (i.e., Portoscuso, F + F, Milazzo) was also similar at the end of the season, as estimated by the timing of the 90th percentiles of annual cumulative catch. Three sites (Milazzo, F + F, Portoscuso) had significantly earlier phenologies than the Capo Passero trap, which was c. 20–30 days later than the other sites for all percentiles. These patterns are also evident in the long-term average catch phenologies at the four sites (Figure 6).

We observed no significant trend in individual weights of captured swordfish during the fishing season at F + F or PS ( $R^2 < 1\%$ ;  $p > .1$  in both cases). However visual inspection of the seasonal development of the individual log weight showed that very small swordfish (i.e., log weight  $< c. 1$ , corresponding to weights of  $< 10$  kg) tended to appear only after Julian day 130 at both F + F and PS



**FIGURE 6** Catch phenologies for swordfish caught as bycatch in bluefin tuna traps at four locations in Italy. (a) Mean ( $\pm 2$  SE) 10th, 50th and 90th percentiles of annual catches at each sites based on annual time series of catches at the different sites. The horizontal lines at top of panel indicate sites which differ significantly from others according to ANOVA and SNK multiple comparison test. See map in Figure 1 for locations. (b) The complete annual long-term average catch phenology at each site, obtained by aggregating data from each site across years and sequencing throughout the year. The time periods differ among sites, but stretches from 1896 to 2010. (c) Seasonal variation in individual log weight of captured swordfish at the two sites (F + F, PS) having largest sample size. Note that for clarity, two data points on days 198 and 221 are omitted for presentation but were included in comparison of variance before and after day 130; these swordfish had log weights of 1.57 and 1.99 (37 and 98 kg, respectively)

(Figure 6). The appearance of these small individuals led to a significant increase in variance of individual log weights (Bartlett's test for homogeneity of variances:  $p = .01$ ,  $.10$  and  $.002$  for F + F, PS and the combined F + F and PS data, respectively).

## 4 | DISCUSSION

One of the objectives of our investigation was to determine whether swordfish bycatch data from bluefin tuna trap records could potentially be used to generate new knowledge and indicators of swordfish ecology. Our study demonstrates that tuna traps at four locations in southern Italy regularly caught swordfish as bycatch (i.e., c. 30–120/year/trap). There were several significant patterns and variations in three different types of data (abundances, sizes and migration phenologies) from these sources. This finding supports the possibility that the trap data for non-target species such as swordfish can provide new ecological insights and potential indicators of status for some non-target species. We provide below some further ecological description and interpretation of the time series, and discuss their potential value and limitations as indicators of population status.

### 4.1 | Trap-based indicators of time–space variability of swordfish ecology in the western Mediterranean (Sicily-Sardinia-Tyrrhenian Sea–Strait of Messina)

Trap records contained data for many years before the start (1950) of the current stock assessment and for years when other data sources used in the stock assessment are limited to targeted commercial fisheries. Furthermore, the recovered bycatch data displayed several consistent patterns of variability over time and space.

First, there were long-term spatial differences in mean ln CPUE between sites. The highest CPUEs were at the trap (Milazzo) located closest to a major Mediterranean spawning site for swordfish (Cavallaro et al., 1991; Palko et al., 1981; Romeo et al., 2009). Catch rates at other sites were generally lower. Spatial differences among sites could be related to proximity to the spawning area, site-specific bathymetric and hydrographic circumstances, or other factors; resolving causes for spatial differences requires further study.

Second, there were significant multi-annual variations in ln CPUE both within individual sites and in across-site indices of ln CPUE. These variations showed peaks in the start of the time series and again in the late 1960s–1970s. Between these periods and since the 1970s, ln CPUE was declining or lower. Although these variations were significant, there was no overall long-term decline from the start of the period to the end. The causes of these variations are likely a complex interaction between climate–hydrographic and fishing effects on overall swordfish abundance and migration behaviour. Resolving these causes and interactions requires new dedicated studies, including further recovery and analysis of historical data, and new statistical, modelling and tagging studies.

Third, we quantified the size and age range of swordfish captured by the traps and which migrate through these coastal waters. The size and estimated age composition was dominated by individuals c. 20–75 kg and c. 2.5–6 years old. Given the weight-at-age and maturity-at-age relationships (i.e., 50% probability of maturity at age 3.5 years and 37 kg; ICCAT, 2020 stock assess wg report), the

captured swordfish were a mixture of larger juveniles and adults, with approximately half of the individuals being mature. The true probability of maturity-at-size or -age of swordfish in the Mediterranean is uncertain and probably differs throughout the region (ICCAT, 2020a). Italian trap-caught swordfish in the late 1800s–early 1900s in this study were similar in size to swordfish caught in coastal Italy in the 1870s which weighed on average c. 50 kg (Goode, 1883), and to those caught in a Spanish bluefin tuna trap at Barbate (Gulf of Cadiz) during 1929–1962; these swordfish weighed on average 62 kg (no year-specific weight data were reported; Rodríguez-Roda, 1964).

There was significant multi-annual variability in the geometric mean weights of swordfish over time at all sites in our study. These variations were evident especially with annually resolved and daily or monthly resolved time series (which were longest), and to a lesser extent with individually resolved time series (which were shortest). Geometric mean weight (annually resolved data) increased from the late 1930s until 1950 but then declined gradually until the present. Declines in mean weight have been reported previously for Mediterranean swordfish but only for more recent time periods: e.g., during 1978–1997 in longline fisheries in the northern Ionian Sea (de Metrio et al., 1999); during 1976–2003 by a harpoon vessel (Di Natale et al., 2005) in the Strait of Messina, and during 1998–2005 for longline fisheries in the eastern Mediterranean Sea (Ionian Sea, Aegean Sea and Levantine basin) (Damalas & Megalofonou, 2014). In contrast, the historical trap data indicate that these more recently observed declines may be a continuation of declines which started at least 1–2 decades earlier, and that mean weights were relatively stable and high before the 1960s.

The decline in geometric mean weight during the 1930s–1940s to 1990s–2010s was c. 21% (from c. 56 to 45 kg). This decline in body weight corresponds to decreases in the probability of maturity from 0.82 to 0.66 (de la Serna et al., 1996) and individual fecundity of 26% (based on the conversions of round and gutted-gilled weights to LJFL (ICCAT, 2020b; Tserpes et al., 2017) and the relationship between LJFL and fecundity (de la Serna et al., 1996). Declines in body size in exploited populations can be due to changes in individual growth rate, recruitment (e.g., via production of a series of strong year classes, which should also be evident as an increase in abundance indices such as our ln CPUE time series), or over-exploitation of the population (Jennings, 2007; Ward et al., 2000). Notably in our case, the decline in mean weight was not accompanied by an increase in ln CPUE (as could be expected if recruitment was increasing), but occurred at approximately the same time as exploitation levels and methods were changing and leading to overall increases in landings (Di Natale et al., 2005; ICCAT, 2017b).

Declines of this magnitude in reproductive properties, if present in the entire Mediterranean population, would have impacts on population egg production, especially if they were not compensated by an increase in ln CPUE, which was stable or started declining in our datasets (in the mid 1970s). Consequently, and assuming that trends detected in these traps are part of the wider Mediterranean-scale declines in mean size discussed above, egg production by the stock was probably falling during the 1960s–1980s. This would reduce

recruit production potential by the stock and may have contributed to the estimated overall decline in stock biomass estimated by ICCAT (Figure 1).

In addition to the multi-annual variability in geometric mean weight (and  $\ln$  CPUE), there was substantial within-year and inter-annual variability in both variables. This variability is likely due to localised spatial variations in swordfish demographics and shorter-term responses of swordfish distributions and migration behaviour to local oceanographic and prey distributions. For example, swordfish catch rates in the eastern Mediterranean Sea vary seasonally and with lunar phase during the summer (Damalas & Megalofonou, 2014), and catch rates of another large migratory fish (bluefin tuna) by one of our studied traps is influenced by local, short-term variations in weather and oceanographic conditions (Addis et al., 2008, 2013). The new datasets identified and developed here allow investigation of such possibilities for trap-captured swordfish.

Fourth, there were clear seasonal phenologies in the annual catch time series at each of the sites. Most of the sites had phenologies whose functional shapes were similar to each other (i.e., gradual increase in catches during the fishing season, rather than a few single large catch events during the year which would produce phenologies characterised by large step-by-step increases throughout the year). This catch pattern probably reflects the generally solitary, non-schooling nature of swordfish in the region (Romeo et al., 2009, 2015a).

Although catch phenologies for the different sites had relatively similar shapes, there were small but significant differences in timings among sites. Swordfish tended to first appear (and disappear) at Portoscuso, c. 7 and 10 days earlier than at F + F and Milazzo, respectively. The earlier arrival (and departure) timing at Portoscuso may be due to a confounding effect of data availability and climate change: this site only had data from 1992–2010 whereas other sites only had data from earlier decades. Warmer conditions since the 1990s have shifted catch phenology for another large migratory fish at this same site (bluefin tuna; Addis et al., 2008) approximately 1 month earlier. It is unknown whether Mediterranean swordfish migration is now earlier than in the past and this hypothesis needs verification in future studies (e.g., further recovery and analysis of trap and existing harpoon fishery data). The southernmost site, Capo Passero, had a much later phenology (c. 20–30 days) than the other three sites. This was due to the later timing of seasonal deployment of the trap, presumably to catch bluefin tuna migrating away from, rather than towards, its spawning sites (Ambrosio & Xandri, 2015).

We observed that there was no seasonal trend in individual body weight during the fishing season. However, the variability of swordfish sizes increased, as small individuals (c. 3–10 kg, or approximately 1–2 year olds) started to appear in catches after c. day 130 at F + F and PS, corresponding to c. 3–4 weeks after the traps had been deployed, operational and already capturing larger individuals. The later appearance of very small individuals suggests that these individuals have different migration behaviour than older individuals, but the details of this behaviour and the habitats used by these sizes of swordfish are presently unknown and require more investigation.

Trap catches had overlapping seasonality with the ancient harpoon fishery for swordfish in the Strait of Messina, whose landings are highest in June–July (Battaglia et al., 2018; Di Natale et al., 2005; Romeo et al., 2015a). This harpoon fishery targets surface-swimming and resting swordfish which migrate to and from the major spawning area in the southeastern Tyrrhenian Sea. Given that the traps are located in close proximity to the same spawning area, and the similar phenologies of trap bycatches and the harpoon fishery, it is highly likely that the trap-derived catch phenologies represent real changes in seasonal occurrence of swordfish in these coastal waters and therefore migration behaviour towards and away from the spawning area. Trap data, particularly if derived from many sites, could provide a broader spatial view of swordfish migration, distribution and phenology which could complement existing, spatially restricted harpoon data sets (mainly from the Strait of Messina). Consequently, a collection of traps deployed in different areas can help elucidate migration behaviour and distribution.

#### 4.2 | Gear effects on estimated sizes

The mean annual sizes observed in the traps, even at its minimum in the early 1980s and early 1990s, are generally larger than the mean sizes caught in the overall commercial fishery for swordfish in the Mediterranean as estimated by ICCAT for the period 1985–2015. The mean sizes of swordfish caught in the Italian traps, particularly in the early-middle decades of the 20th century, as well as those caught in a Spanish trap (Rodríguez-Roda, 1964), were therefore approximately twice as large as those caught by the current Mediterranean commercial fishery. Trap-caught swordfish were approximately similar in size to, or slightly larger than, harpoon-caught swordfish summarised here and in the literature (Di Natale et al., 2005).

The differences in mean annual sizes between data sources are at least partly due to differences in gear type and deployment location, as well as differences in exploitation over time. For example, the main legal fishing gear in use now are longlines which tend to catch relatively small individuals when deployed in offshore waters near the surface (ICCAT, 2019); deeper deployments of longlines tend to catch larger individuals than shallower deployments (Cambie et al., 2013; ICCAT, 2019). In addition, as seen in our data compilation, harpoon-caught swordfish during 2002–2015, primarily in the Strait of Messina, were also significantly larger than those caught by the entire Mediterranean fishery during 1985–2015 (c. 42 vs. 27 kg), but still smaller than trap-caught swordfish in the early 1900s. However, if data were available from harpoons (or longlines) from the early 1900s, it is likely that mean sizes from these gears would also have been larger than they are now; this suggestion is supported by the observation that mean size of harpoon-caught swordfish by a single vessel in the Strait of Messina during 1976–early 1980s was c. 60 kg, before declining to c. 40 kg in the early 2000s (Di Natale et al., 2005), when spawner biomass had been reduced by c. 67% (ICCAT, 2019). Actions to increase mean size in the population could include



an overall reduction in exploitation, deployment of fishing methods with lower selectivity for small individuals and reduction of exploitation in areas inhabited by small individuals.

Resolving causes of changes in mean size is best done by comparisons of trends *within* gear types deployed at consistent times and locations, and then comparing trends across gear types; however until now, most data sets for mean weight only cover the time period when fishing pressure was already high. Our new data sets include a long time period when harvest removals were much lower (ICCAT, 2019) and provide new knowledge of the timing of when declines in size occurred and its possible cause.

### 4.3 | Benefits and limitations of trap data as indicators of status of Mediterranean swordfish

Traditional bluefin tuna trap data have been shown earlier to generally have key properties suitable for use as an indicator of population status of certain fish species e.g., bluefin tuna (Addis et al., 2008; Ambrosio & Xandri, 2015; Ravier & Fromentin, 2001). These properties include similar sampling methodology and an extensive temporal-spatial coverage in the region. Furthermore, the long duration of these time series predates many key human impacts (e.g., overexploitation; climate change) in the Mediterranean Sea (Piroddi et al., 2017); recovered trap bycatch data therefore represent a potential for comparing swordfish ecology under different conditions of human impact, ecosystem states and oceanographic-climate forcing, and for establishing new baselines of demographic variability with which future changes can be compared.

This study has started to investigate whether such properties are also evident for other catch species than the target species and for properties which can be derived from the catch records associated with such traps. Our findings indicate that records from four such traps can be used to improve our understanding of three demographic characteristics (abundance, mean size, migration phenology) of swordfish ecology in this region and illustrate the levels of variability that have occurred over long time scales, including those before Mediterranean swordfish fisheries intensified. Consequently, our findings and results should be considered as initial steps towards increasing the potential value and utility of traditional trap data for bycatch species such as swordfish in the Mediterranean Sea. Given that most ecological and fishery datasets for this species in the Mediterranean are of more recent origin and shorter duration (Damalas et al., 2015; ICCAT, 2020c), these new datasets are now the oldest and some of the longest time series available for this species both in the Mediterranean and globally. Searches for and recovery of data from other trap and historical records could be productive and strengthen the patterns detected here.

It is possible that these datasets and findings can already contribute new understanding of ecological processes affecting swordfish population dynamics in the Mediterranean Sea. Furthermore as a nontarget source of fishery data, they could support knowledge derived from other fishery-dependent data sources. Some

comparisons of our results with those from previous investigations suggest that these perspectives may be promising. First, the declines in ln CPUE and mean weight in the trap data since the 1960s–1980s and relatively low levels since then are consistent with (1) declines seen in other data sets (references above); (2) the large increase in exploitation between the 1950s and 1970s (Figure 1) (Damalas & Megalofonou, 2014; Di Natale et al., 2005; ICCAT, 2019); and (3) the international assessment of swordfish ecological status in the Mediterranean (ICCAT, 2020c).

Second, several other spatially limited datasets from the commercial swordfish fishery and which represent swordfish abundance metrics for local regions are used in the assessment for this stock (ICCAT, 2020c); in this way, the trap data compiled here have similar spatial representativeness (i.e., they only cover part of the entire range of the assessed population) as these other datasets. They may therefore be an indicator of abundance in this region, in addition to the targeted fishery dataset.

Third, the data compiled and analysed in our study have been collected from a key portion of the population range where a large (but unknown) portion of the stock aggregates on an annual basis, rather than from a peripheral area where demographics might be particularly sensitive to ecosystem variations (e.g., local prey abundances in feeding area) or which might only be occupied occasionally by a small proportion of the stock biomass.

These demographic, life history and biogeographic perspectives suggests that changes detected in this investigation could reflect changes in stock biomass and ecology. Consequently, trap data such as these could be (come) effective fishery-independent, technologically stable indicators of population status, because they have both a long history and potential future role in population monitoring.

The potential applications of these data however must be cautioned by acknowledging and overcoming their limitations. One limitation in our recovered data is the heterogeneous temporal coverage within a given trap location and among traps from different locations. A second limitation is the spatial coverage of the traps (i.e., from a relatively small area) used in our investigation, relative to the Mediterranean-wide distribution of swordfish. For example, it is presently unknown whether or to what extent the variations and trends seen at these locations have co-occurred elsewhere in the Mediterranean and therefore how representative our findings are of dynamics of the entire Mediterranean population of swordfish. These temporal-spatial limitations are discussed further in the [Supplementary Material](#). Both of these limitations can be partly overcome by, for example, increasing the number of recovered trap datasets to fill time–space gaps in coverage, and complementing analyses of these data with analyses using other kinds of data (e.g., commercial catch, tagging, genetic and other biological data).

Fisheries management is increasingly being encouraged to become more ecosystem-based, integrative and adaptive to changing conditions (FAO, 2018a; FAO, 2018b; ICES, 2019). However a prerequisite for such an integration is knowledge of past dynamics that can be used to parameterise dynamic (non-stationary) models of

population processes (Chang et al., 2020; Mildenerger et al., 2020; Tableau et al., 2019). Here we have identified new data sources and recovered datasets which potentially can quantify the (in)stability of population dynamics over modest-long time scales and in particular over periods of time earlier and longer than those currently used in the Mediterranean stock assessment.

The investigated trap records contain species bycatch data in amounts that allow detection of significant and ecologically-relevant temporal and spatial variations in biological properties. This finding is consistent with earlier applications of bycatch data from fisheries for other target species for studying their ecologies and the biodiversity of communities (Ferretti et al., 2008; Wang et al., 2020). However, what makes these data unusual compared with many earlier bycatch studies is their origin from a relatively early time period compared with the timing of intensification of fishing, and their long duration and higher temporal and biological resolution (individual-level size information). Even greater insight into swordfish ecology and more generally the dynamics and biodiversity of large coastally migrating fish species (e.g., bluefin tuna, other tunas, elasmobranchs which are caught in these traps) could be attained if several traps were re-deployed and established as scientific monitoring tools (Addis et al., 2008, 2012) for future data collection purposes. New deployments for monitoring purposes would allow detection of changes in future and the potential identification of their causes, and the evaluation of the effectiveness of new marine ecosystem and fishery management policy in achieving objectives. Such deployments would have to be done within the framework of bluefin tuna fishery management regulations, including authorisation and cooperation from relevant national and international authorities and stakeholders (e.g., ICCAT, fishing industry). Furthermore, given that several traps are already deployed annually in the northeast Atlantic (Spain, Portugal, Morocco) and that there is genetic evidence which suggests that Mediterranean swordfish can be found outside its stock management area (i.e., west of the Strait of Gibraltar to 10° W; Smith et al., 2015), analyses of the genetic composition of trap-bycatched swordfish in the Mediterranean and NE Atlantic could improve understanding of meta-population structure and mixing.

In summary, we have analysed new—neglected—time series data for a charismatic, commercially important, highly migratory top predator in the Mediterranean Sea. Our study adds to a growing literature which highlights the value of historical reports and archives for generating new insights into marine ecological dynamics (Caswell et al., 2020; Engelhard et al., 2016; Schwerdtner Máñez et al., 2014; Thurstan, 2022). Here we have shown that such sources provide information not only about the target species of fisheries but also bycatch species. Furthermore, the detailed time-space resolution and biological documentation of the catch records has enabled us to investigate ecological properties and phenomena which are impossible with studies at coarser resolution (e.g., removals of species of unspecified size at annual or longer time scales). For example, the high time-space resolution of data recovered here has allowed quantification of migration phenologies for the first time for swordfish in the Mediterranean. These data will allow future studies to investigate

the roles of exploitation, oceanographic and climatic conditions on both the catch rates (as done already for bluefin tuna at one of our sites; Addis et al., 2013) and phenologies (Asch, 2019). In addition, the individual- and group-level size measurements allow investigation of which lifehistory stages have been exploited (e.g., juveniles, adults), the spatial use of coastal habitats by such lifehistory stages, and how body size varies over years and seasonally in relation to exploitation and ocean-climate variability. Such investigations can now be conducted over longer periods of time and before the onset of many intensive human impacts than could be done before recovery of these data.

More generally, it is likely that reports of similar biological and time-space resolution as for these trap data are available for many other (targetted and bycatch) species and regions in the Mediterranean Sea and global ocean. Extraction, digitisation and analysis of data from such reports could provide many new insights into population and ecosystem processes at scales and/or for time periods beyond the reach and budget of contemporary sampling and documentation, and much faster than waiting for new data to accumulate 1 year at a time. New knowledge generated from such work is needed to help estimate the vulnerability of populations and species to anthropogenic and natural drivers of change, forecast how they might respond to those drivers in future and improve the ecological basis for sustainable management of marine populations, species and ecosystems.

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## CONFLICT OF INTEREST

The authors have no competing interests.

## DATA AVAILABILITY STATEMENT

All raw data used in analyses in this investigation have been deposited at Pangaea.de: <https://doi.org/10.1594/PANGAEA.928996>. <https://doi.org/10.1594/PANGAEA.928980>. <https://doi.org/10.1594/PANGAEA.928981>. The data are available open access to the public (MacKenzie et al., 2021).

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