



Dynamics of killer whale, bluefin tuna and human fisheries in the Strait of Gibraltar



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ARTICLE INFO

Article history:

Received 12 May 2015

Received in revised form 25 November 2015

Accepted 29 November 2015

Available online xxxx

Keywords:

Killer whales

Life-history parameters

Interaction

Abundance

Anthropogenic disturbance

ABSTRACT

A complex balance has arisen between the bluefin tuna, killer whales, and human activities in the Strait of Gibraltar. Recent changes in fishing effort have dramatically decreased tuna stocks, breaking this balance. Killer whales exhibit two strategies for feeding on tuna: active hunting and depredation on a drop-line fishery. From 1999 to 2011, a small community of 39 individuals was observed in the Strait in spring and summer. All individuals displayed active hunting and 18 of them also depredated on the fishery. These differences in foraging behaviour influenced life-history parameters. Adult survival for interacting and non-interacting individuals was estimated at 0.991 (SE = 0.011) and 0.901 (SE = 0.050), respectively. Juvenile survival could only be estimated for interacting individuals as 0.966 (SE = 0.024), because only one juvenile and one calf were observed among non-interacting individuals. None of the interacting calves survived after 2005, following the decrease in drop-line fishery catches. Calving rate was estimated at 0.22 (SE = 0.02) for interacting individuals and 0.02 (SE = 0.01) for non-interacting. Calving interval, which could only be calculated for interacting groups, was 7 years. The population growth rate was positive at 4% for interacting individuals, and no growth was observed for non-interacting individuals. These differences in demographic parameters could be explained by access to larger tuna through depredation. Consequently, we found that whales would need more tuna to cover their daily energy requirements while actively hunting. Therefore, our findings suggest an effect of artificial food provisioning on their survival and reproductive output. Urgent actions are needed to ensure the conservation of this, already small, community of killer whales. These include its declaration as Endangered, the implementation of a conservation plan, the creation of a seasonal management area where activities producing underwater noise (i.e. military exercise, seismic surveys or even whale watching activities) are forbidden from March to August, and the promotion of bluefin tuna conservation. Additionally, energetic requirements of this whale community should be taken into account when undertaking ecosystem-based fishery management for the Atlantic bluefin tuna stock. In the meantime, as marine predators are most sensitive to changes in fish abundance when prey abundance is low, we suggest an urgent short-term action. Artisanal fisheries, such as drop-lines, should be promoted instead of purse seiners in the Mediterranean Sea. This will help to maintain the survival and reproductive output of the whale community until showing clear signs of recovery and stability, and/or their prey stock recovers.

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1. Introduction

Atlantic bluefin tuna (*Thunnus thynnus*) (hereafter tuna) perform a gametic migration, entering the Mediterranean Sea through the Strait of Gibraltar (hereafter Strait) in late spring (Sella, 1928, 1929; Rodríguez-Roda, 1964). After spawning, tuna perform a trophic migration to the eastern North Atlantic in summer (de la Serna et al., 2004; Aranda et al., 2013). In the Strait, tuna have been caught for centuries using trap-nets, an artisanal fixed and passive gear (Doumenge, 1998).

In 1995, a new artisanal drop-line fishery was developed by Spain and Morocco in the Strait, to catch tuna on their trophic migration (Srouf, 1994; de la Serna et al., 2004). The Eastern tuna stock has been exploited by traditional fisheries for centuries, but in the 1960s industrial purse-seine and long-line fleets replaced the traditional fisheries (Fromentin and Powers, 2005). Thus, tuna stock has been mostly declining since then (ICCAT, 2011; Taylor et al., 2011).

Other natural predators feed on tuna such as large pelagic sharks and killer whales (*Orcinus orca*) (Fromentin and Powers, 2005). Killer whales have been observed in the Strait for centuries (Horozco, 1598; Richard, 1936; Aloncle, 1964). Stable isotope analyses suggest that their main prey is tuna (García-Tiscar, 2009). They have been assigned

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to a single population in conjunction with whales sampled in the Canary Islands based on nuclear DNA loci (Foote et al., 2011), and thus we define killer whales in the Strait as a community of whales. They occur in the Gulf of Cadiz in spring (Esteban et al., 2013), while they actively forage on tuna around the trap nets using the endurance–exhaustion hunting technique (hereafter active hunting) (Guinet et al., 2007), which consists of chasing tuna at high speed for around 30 min. According to the authors, this technique requires a high energy investment and the average catch is usually small–medium size. It was also suggested that killer whales likely rely on other techniques to catch larger individuals. During summer, the whales are also observed actively hunting in the central waters of the Strait (de Stephanis et al., 2008). However, a new operational and biological interaction has been described between killer whales and tuna drop-line fisheries, where killer whales depredate tuna from their baited hooks, for which they presumably invest less energy (Guinet et al., 2007; de Stephanis et al., 2008; Esteban et al., 2013). Esteban et al. (2015) described five pods in the Strait (A1, A2, B, C and D). While all pods have been seen actively hunting tuna, only two pods (A1 and A2) have learned to interact with the fishery.

The inclusion of artificial food provisioning and a presumably highly energetic resource, such as tuna from fisheries, may influence demographic parameters and reproductive output within these killer whales. Our aims were to elucidate the consequences of this interaction on life-history parameters for interacting and non-interacting killer whales, and compare their energy requirements for interacting or active hunting.

2. Materials and methods

2.1. Data collection

The study area is located between 5 to 6°W. It was surveyed between 1999 and 2011 for cetaceans. More details regarding the sampling protocols can be found in de Stephanis et al. (2008) and Esteban et al. (in press).

2.2. Life history parameters

Photo-identification techniques (e.g. Bigg et al., 1990; Ottensmeyer and Whitehead, 2003) were used as previously described (Esteban et al., 2015) to identify every individual observed in the study area. The photo-identification catalogue is available at www.cetidmed.com, where it is updated annually.

2.2.1. Abundance

Censuses were conducted annually from which a cumulative abundance count was calculated as the total number of individuals observed every year minus those determined to be dead. Killer whales' social structure in the Strait has been suggested to be a matrilineal social structure without individual dispersal from their natal group (Esteban et al., 2015); therefore, an individual was considered dead whenever it was not seen within its pod for three consecutive years or it was found stranded. We used photographs combined with direct observations to determine sex and approximate age of whales (Olesiuk et al., 1990). Following the definition of sexual maturity described in Olesiuk et al. (1990), we sorted the whales into three categories. Adults included males that were mature individuals presenting a prominent dorsal fin and females that were mature individuals accompanied by calves or juveniles; juveniles were individuals older than 1 year but still not mature and calves were individuals younger than 1 year. We also classified whales by their foraging behaviour as either interacting (hereafter INT) for whales observed depredating on the tuna drop-line fishery at least once, or non-interacting (hereafter NOT) for whales that were never observed depredating.

2.2.2. Mark-recapture analysis

As only some pods were seen every year (Appendix A, Table A.1), we used mark-recapture analyses to estimate (1) survival rate for the three

age classes (calf, juvenile, adult) and (2) population growth rate using the software MARK 7.1 (White and Burnham, 1999). All models were compared using QAIC_c (Quasi Akaike's information criterion adjusted for small sample bias) (Sugiura, 1978). The best model was selected by the lowest QAIC_c. Models within $\Delta\text{QAIC}_c \leq 2$ were considered to be well supported by the data (Burnham and Anderson, 2004). In case that $\Delta\text{QAIC}_c \leq 2$, model averaging was used (Buckland et al., 1997).

2.2.2.1. Survival rate. Multistate mark-recapture models (Hestbeck et al., 1991; Brownie et al., 1993) were used to examine differences in survival (S), the probability of an animal to be re-encountered (p) and state transition probabilities (ψ), among the two groups INT and NOT, contained in three different states: adult (A), juvenile (J) and calf (C). To assess the goodness-of-fit of the model, we used a parametric bootstrap procedure implemented in MARK 7.1 (White and Burnham, 1999). The variance inflation factor measuring possible over-dispersion in the data, \hat{c} , was estimated as the deviance estimate from the original data divided by the mean of the simulated bootstrapped deviances and applied to all models if > 1 (White, 2002). For both groups, we constrained state transitions ψ from A to C, A to J and J to C as 0 as these transitions are impossible. We also fixed transitions from C to J as 1, as calves become juveniles after one year of life by our definition. We fixed S_{CNOT} and S_{JNOT} to 1 because the only calf and juvenile in the NOT group were observed the in last year their pod was seen, so survival could not be estimated (Appendix A, Table A.1). For NOT, we fixed $p = 0$ for all age classes in 1999–2001 and 2008–2009 as no animals were seen during these years.

We considered various models to test for differences and similarities of survival between INT and NOT, and through time. We started with the full time-dependent model (t) (Appendix A, Table A.2, model 5). We then fitted a more parsimonious model by constraining all parameters to be constant in time (\cdot) (Appendix A, Table A.2, model 3). To investigate calf survival for INT, we tested it as constant over the study period (model 3), time-dependent (model 4) and fixed to 1 in 1999–2005 and to 0 in 2006–2010, following our direct observation of no calf surviving their first year of life after 2005 (model 1). Finally, we tested for similarities or differences of adult survival between INT and NOT (models 1 and 2).

2.2.2.2. Population growth rate. The population growth rate was modelled with a Pradel model with survival (ϕ) and lambda (λ) (Pradel, 1996). Lambda was modelled as constant in time, and was tested as equal or different for INT and NOT individuals. For NOT, we fixed $p = 0$ in 1999–2001 and 2008–2009 as no animals were seen during these years.

2.2.3. Reproductive rates

Calving rates for INT and NOT were calculated as the total number of calves born during a given year out of the total number of reproductive females available that year within the group. Females calving a year before were omitted for that year, because of their 12 month lactation period (Olesiuk et al., 2005; Kuningas et al., 2013). Females were assigned as reproductive females whenever they were known to be in their reproductive years (10–46 years old) (Olesiuk et al., 2005) or the years they were seen with a calf. A Mann–Whitney–Wilcoxon test was performed to compare annual calving rates between INT and NOT.

Calving interval was calculated as the interval at which the same female gives birth to successive viable calves (Olesiuk et al., 1990) (Appendix A, Table A.3). Fecundity rate was calculated as the reciprocal of calving interval (Olesiuk et al., 1990).

2.3. Interaction between tuna drop-line fisheries and killer whales

Available data on tuna catches by drop-line fisheries in the Strait were compiled from data of the regional Government of Andalusia (Spain) and data from Malouli Idrissi et al. (2013) for Morocco.

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