



Bycatch of loggerhead sea turtles: insights from 14 years of stranding data

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ABSTRACT: We present a detailed analysis of sea turtle strandings ($n = 619$) over a 14 yr period (1993 to 2006) from the Valencian Community (eastern Spain). Loggerhead turtles *Caretta caretta* accounted for 98.1 % of recorded strandings. Although we detected an increasing trend in the annual number of strandings over the study period, we expect that this was driven primarily by the increase in detection effort. Loggerhead turtles were mainly juveniles (mean \pm SD curved carapace length from notch to tip [CCL] = 53.6 ± 12.6 ; range: 16 to 80.2 cm, $n = 312$) and strandings were far more frequent in summer months (69.6 % June to September). We believe that interaction with longline fisheries was by far the main cause of observed stranding (>28 % of all cases, 43.5 % of 393 turtles with likely mortality cause identified). Turtles showing signs of interaction with longlines were, on average, larger (CCL = 57.5 ± 10.4 ; range: 29.8 to 80.2 cm, $n = 116$) than turtles stranded due to other causes (CCL = 51.4 ± 13.3 ; range: 16 to 79 cm, $n = 196$; t -test: $t = -4.49$, $p < 0.001$) and were more frequent in summer months, when longline fishing effort off the coast was highest. Recent reductions in longline effort may have led to a decrease in recent years in the proportion of stranded turtles with evidence of longline interaction. Although inferences from stranding data must be subject to a number of caveats, when considered over wide spatio-temporal extents and in conjunction with other data sources, they can offer useful insights into the geographic ranges, seasonal distribution and life history of marine species of conservation concern.

KEY WORDS: Loggerhead sea turtle · *Caretta caretta* · Fisheries bycatch · Strandings · Spanish Mediterranean · Conservation

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INTRODUCTION

Many species of large marine vertebrates, such as marine mammals, sea turtles or seabirds are subject to bycatch in commercial fisheries operations, and these captures may occur in substantial numbers (Lewison et al. 2004a). Assessment of the impact of bycatch, however, is difficult (Hall et al. 2000, Lewison et al. 2004b). Worldwide, sea turtles are caught primarily by trawl, pelagic longline and coastal gillnet fisheries (Hays et al. 2003, Lewison & Crowder 2007, Peckham et al. 2007). In the Mediterranean Sea, thousands of turtles are estimated to be incidentally captured or entangled by longline fisheries each year (Kemf et al. 2000, Lewison et al. 2004a). Although sea turtle bycatch has been reported from other fisheries

throughout the Mediterranean (Godley et al. 1998, Carreras et al. 2004, Casale et al. 2005), bycatch in longline fisheries has been cited as the most important source of anthropogenic mortality for sea turtles (Gerosa & Casale 1999), particularly in the western basin of this sea (Camiñas & De la Serna 1995, Laurent et al. 2001, Camiñas et al. 2006).

Two key points generally compromise bycatch studies: (1) data limitations and (2) spatial coverage of bycatch information (Lewison et al. 2004b). With respect to data limitations, direct measurements of bycatch require on-board observation programmes, which are costly and require independent and well-trained observers. Although additional data can be derived from catch books/interviews with fishermen, these often need validation (Carreras et al. 2004).

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Concerning spatial scale, fishing effort is globally distributed and many large marine vertebrates, including sea turtles, have ocean-wide distributions. Therefore, information must often be compiled from smaller scale studies to form large-scale perspectives of the impact of incidental catch on non-target species. Although integrated studies based on observer data and interviews provide some data on the number of turtles caught and post-capture mortality rates (Carreras et al. 2004, Carranza et al. 2006, Mejuto et al. 2006 and references therein), the sample sizes in these studies are typically small (i.e. with low statistical power), and the amount of observed gear is dwarfed by the overall fishing effort of the fishery in question (Lewison et al. 2004b). Most importantly, very few studies are able to provide accurate post-capture mortality rates (Chaloupka et al. 2004, Sasso & Epperly 2007), as not every interaction with longline gear is lethal.

Additional insight can be gathered from stranding data. We consider strandings as turtles washed ashore dead or alive or found floating dead or alive in coastal waters. Although inferences from these data are subject to a number of caveats, when integrated over wide spatio-temporal extents, stranding data can provide information about geographic ranges, seasonal distribution and life history of marine vertebrates, including sea turtles, particularly for life stages underrepresented in the literature (Witt et al. 2007). For instance, Snover et al. (2007) use data of stranded turtles to ascertain and validate loggerhead turtle growth rates. Furthermore, when integrated with careful post-mortem examination, these data can also allow inferences to be made on patterns of bycatch (Panagopoulos et al. 2003).

The location of the Valencian Community (eastern Spain) is of special interest in relation to loggerhead sea turtle *Caretta caretta* distribution and conservation for the Atlantic and Mediterranean basins. Although adults from Mediterranean nesting stocks appear limited to the eastern basin (Godley et al 2003, Broderick et al. 2007, Zbinden et al. 2008), the western Mediterranean hosts juveniles from both Mediterranean and Atlantic rookeries (Laurent et al. 1998). These 2 stocks appear to remain segregated. Cape La Nao (Fig. 1) is likely the mainland boundary between groups, with the Mediterranean stocks remaining in the northern Balearic Sea and the Atlantic individuals in the Algerian basin, with limited exchange (Cardona et al. 2005, Carreras et al. 2006, Revelles et al. 2007a). Here we present a detailed spatio-temporal analysis of the patterns of marine turtle stranding along the coastline of the Valencian Community from 14 yr of data, to seek insights into patterns of bycatch and other anthropogenic impacts in the region.

MATERIALS AND METHODS

Sea turtle stranding data were collected between 1993 and 2006 in the Valencian Community by a stranding network involving public and private institutions, including public rescue centres and aquaria in the region. This network records dead or injured turtles stranded on beaches, or turtles found floating dead or in a weakened condition. It is coordinated via a 24 h telephone hotline.

The Valencian Community is a region with a high degree of development and tourism along its coast and is comprised of the 3 coastal provinces of, from north to south, Castellón, Valencia and Alicante; totalling 419 km of coastline that extends from 40° 31' N, 0° 31' E to 37° 51' N, 0° 45' W (Tomás et al. 2003a). Cape La Nao is located in the north of Alicante province (38° 44' N, 0° 14' E, Fig. 1) and divides the coast studied into 2 stretches of 257 and 162 km, north and south, respectively. Most stranding data were reported to the coordinators of the network with no precise geographical position provided. For this reason, we collated stranding data by municipality, except for those with the longest coastlines, Valencia City in Valencia Province and Elche in Alicante Province; for these municipalities we created 3 and 2 subareas, respectively. Although we recognise that there is some variation in the length of coastline among sections, we feel that this level of data aggregation allows sufficient spatial resolution to elucidate patterns of distribution within the study area.

Animals found alive were moved to public rescue centres and subjected to veterinary examination and rehabilitation; the carcasses found were brought, where possible, to the research installations of the University of Valencia (UV) for necropsy. We measured turtles (curved carapace length notch to tip [CCL] and curved carapace width [CCW]), identified them to species level, and determined the likely cause of death based on a complete external and internal examination. A trained team performed necropsies of fresh carcasses following a detailed protocol. If no severe lesions were detected in the presence of evidence of anthropogenic impacts (such as ingested longline gear, marks of nets around flippers or neck), these were assigned as the stranding cause. In cases when there was evidence of multiple lesions, the most severe and recent was assigned as the primary cause of stranding. Interaction with longline fishery was assigned as the stranding cause when branch lines and/or hooks were found upon examination. We were not able to identify the cause of stranding in all the cases, as we did not have access to all animals due to logistical constraints.

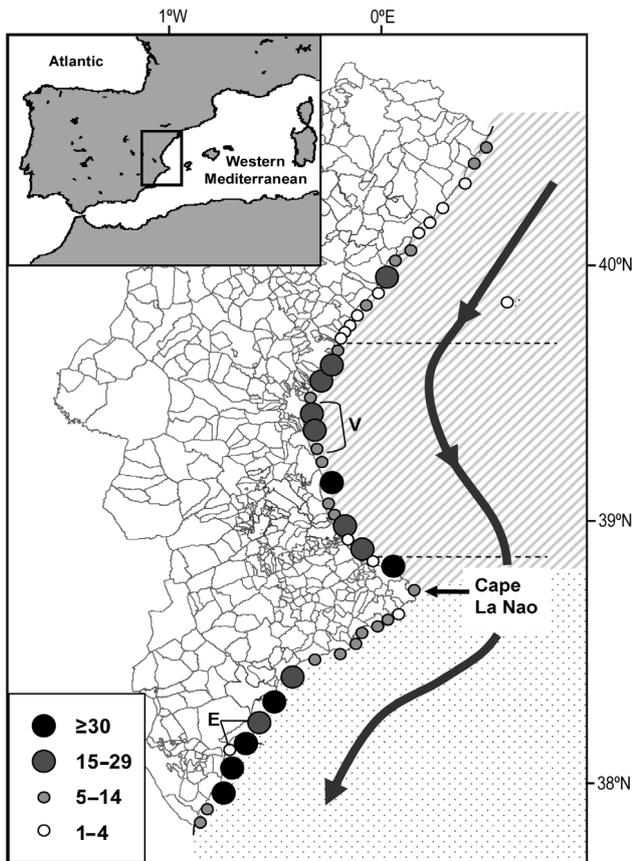


Fig. 1. *Caretta caretta*. Study area and distribution of strandings per municipality in the Valencian Community for the period 1993–2006. Black dashed lines represent the coastal limits of provinces (north to south: Castellón, Valencia, Alicante). Grey arrow represents main surface current (Millot 1999). Dotted and hatched areas represent the areas occupied mainly by Atlantic and Mediterranean loggerhead turtles, respectively, according to Carreras et al. (2006). V: municipality of Valencia; E: municipality of Elche (see 'Materials and methods'). Strandings at sea correspond to those registered in the Columbretes Archipelago

RESULTS

Total strandings, species and life stages

Between 1993 and 2006, we recorded a total of 619 sea turtles stranded (46.7 ± 28.2 annually, mean \pm SD; range: 12 to 124). Strandings were recorded throughout the study area in almost every coastal municipality (Fig. 1). There was no significant difference in the total of stranding records between the regions south (301) and north (318) of Cape La Nao (chi-squared test, $\chi^2 = 0.467$, $df = 1$, $p = 0.49$). However, overall density of strandings was higher south of the cape (south: 1.9 turtles km^{-1} ; north: 1.2 turtles km^{-1}). In each of these 2 areas we observed an increasing north-south pattern in stranding occurrence (Fig. 1).

With the exception of 2001, there was an apparent general increase in strandings observed between 1993 and 2006, but since 2000 the annual number of stranded turtles appears to have been relatively stable (Fig. 2a), at a mean density of 0.13 stranded turtles km^{-1} of coast yr^{-1} .

The vast majority of stranded turtles (98.1%) recorded were loggerheads. The remaining 1.9% comprised Kemp's ridleys *Lepidochelys kempii* ($n = 2$) (Tomás et al. 2003b, Tomás & Raga 2007) and leatherbacks *Dermochelys coriacea* ($n = 2$) (Tomás et al. 2003a), while another 8 carcasses were not definitively identified to species. To date, only 1 green turtle *Chelonia mydas* has been reported from the Valencian Community, and this was prior to the present study (Raga & Salinas 1990).

Mean CCL for the stranded loggerheads was 53.6 ± 12.6 cm ($n = 312$; range: 16 to 80.2). The size distribution of these turtles is larger than expected and includes animals of all size classes from neritic juvenile to adult whether from Mediterranean (Broderick et al. 2003) or Atlantic populations (Bolten 2003, Kamezaki 2003). There was no significant difference in mean size among years (Fig. 2b), but we did find significant differences among months (ANOVA, $F_{11,300} = 3.32$, $p < 0.001$). Turtles recorded in summer months (June through August) were significantly larger than those recorded in spring (March through May, see Fig. 4a). Based on the apparent separation of stocks proposed by Carreras et al. (2006), we tested for geographic differences in loggerhead turtle size. We found no difference between those turtles north (54.2 ± 12.4 cm, $n = 183$) and south (52.1 ± 12.9 cm, $n = 129$) of Cape La Nao (t -test: $t = 1.75$, $p = 0.08$).

Seasonality

Stranding records show a clear seasonal pattern, with records increasing in summer (July to August) (ANOVA, $F_{11,146} = 8.534$, $p < 0.001$; Tukey post hoc comparison, $p < 0.01$, see Fig. 4b). Although subject to annual variation (Fig. 2c), neither the mean (Spearman's rank correlation, $r_s = -0.31$, $p = 0.29$) nor the median day of strandings per year ($r_s = -0.51$, $p = 0.052$) changed over time.

Causes of stranding and mortality

We investigated the likely cause of stranding of 393 turtles (63.5% of the total) and found that 312 (50.4% of the total) had stranded as a result of anthropogenic interaction. Of these, interactions with longline fishery were the most frequent (12.2 ± 11.5), accounting for

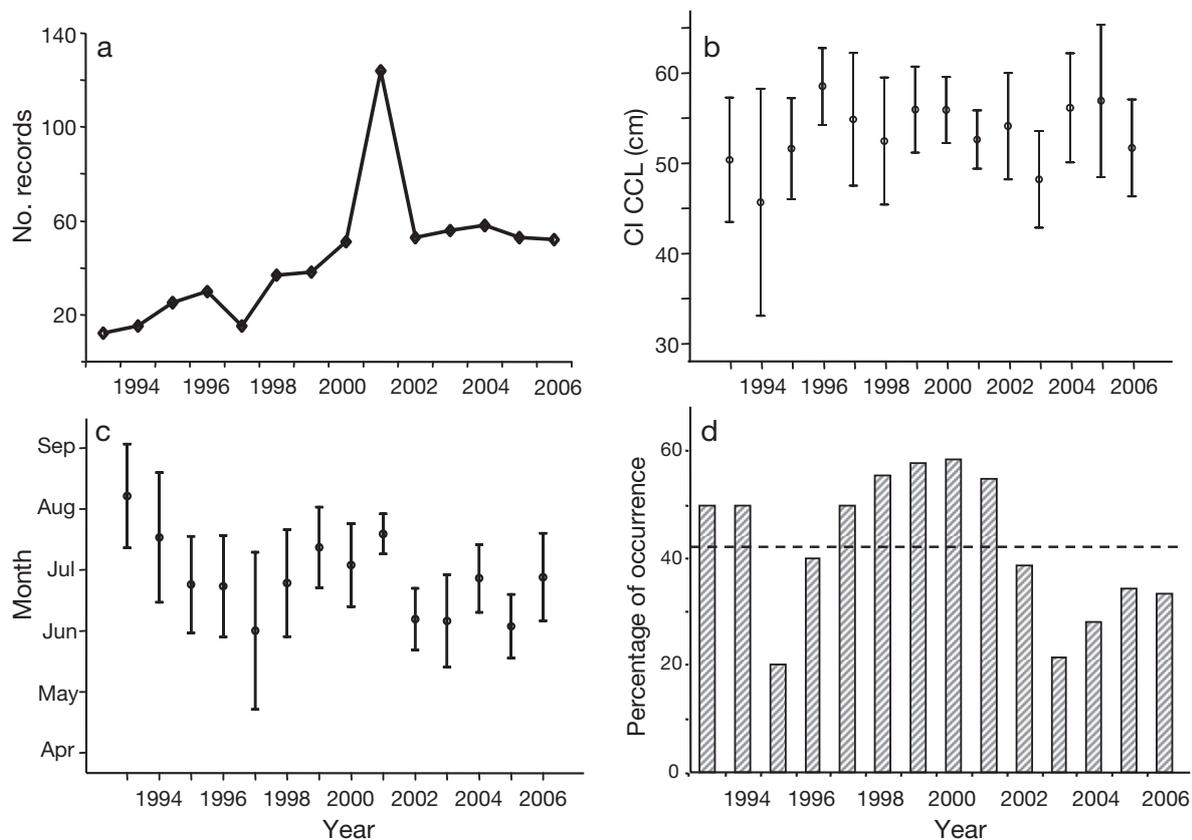


Fig. 2. *Caretta caretta*. Annual variation in (a) number of stranding records, (b) mean \pm 95% CI of curved carapace length (CCL), (c) mean \pm 95% CI monthly distribution of records, and (d) proportion of records assessed as resulting from longline interaction. Dashed line in (d) represents the annual mean percentage of occurrence of longline interaction

27.6% of the total records (43.5% of the records with an identified cause of stranding). However, in each of the most recent 5 yr the proportion of turtle strandings with an identified cause attributed to longline interactions was below the 14 yr average ($42.6\% \pm 13.2$, Fig. 2d).

No significant difference was found in the relative proportion of strandings attributed to longline interaction between the northerly ($n = 84$) and southerly ($n = 85$) areas off Cape La Nao ($\chi^2 = 0.006$, $df = 1$, $p > 0.05$). Of the 171 turtles stranded due to longline interaction, 49 were rehabilitated in rescue centres and released back to the sea after tagging.

Carapace length of stranded turtles attributed to longline fisheries interactions was between 29.8 and 80.2 cm (mean CCL = 57.5 ± 10.4 cm, $n = 116$), but 90% of these turtles were between 40 and 74 cm (Fig. 3). Stranded turtles with signs of longline interaction were significantly larger than those without clear signs of longline interaction (mean CCL = 51.4 ± 13.3 cm, $n = 196$; $t = -4.49$, $p < 0.001$). Most of the turtles with signs of longline interaction were stranded in summer months (June to September, Fig. 4c).

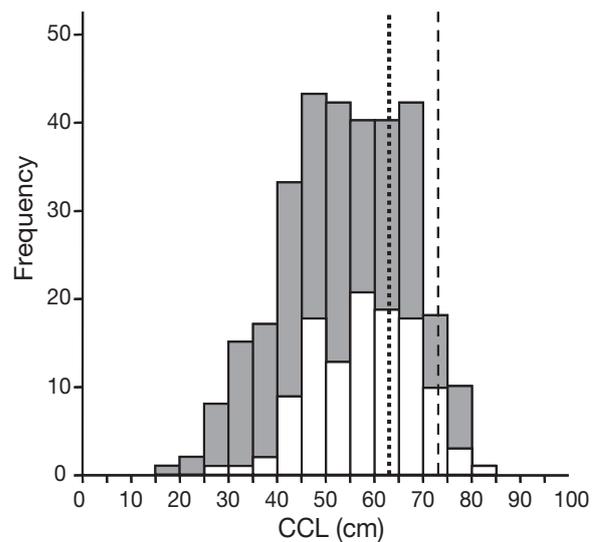


Fig. 3. *Caretta caretta*. Curved carapace length (CCL) of all measured turtles (grey bars, $n = 327$) and turtles interacting with longline fishery (white bars, $n = 116$) for 1993–2006. Dotted line: minimum carapace length for Mediterranean stocks (Broderick et al. 2003); dashed line: minimum carapace length for Atlantic stocks (Kamezaki 2003)

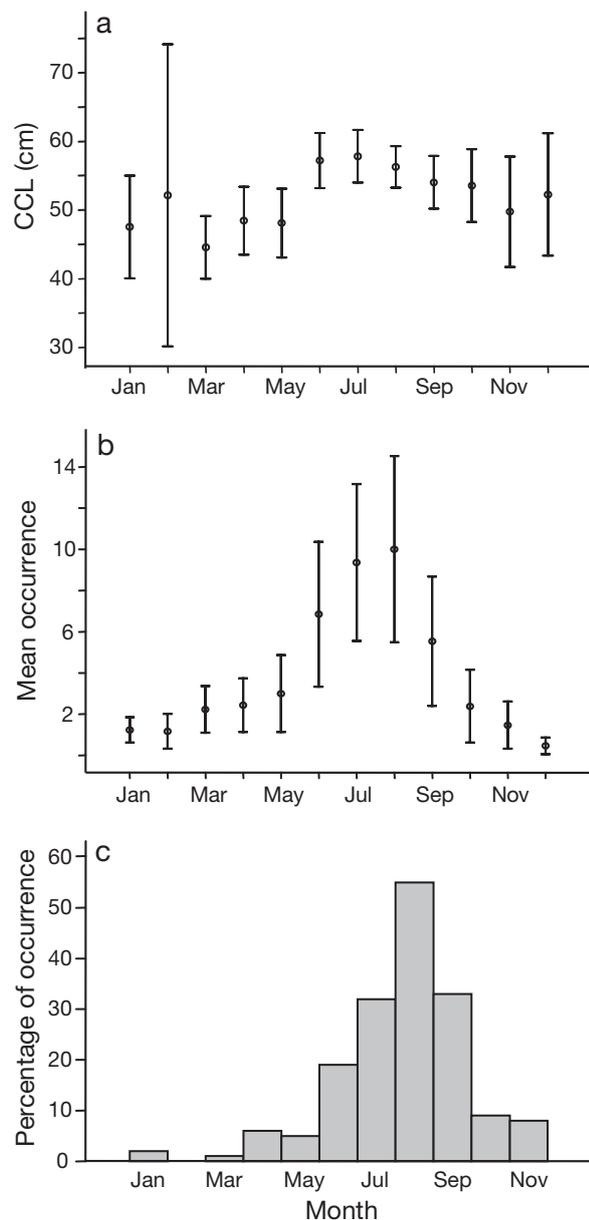


Fig. 4. *Caretta caretta*. Monthly variation in turtle records. (a) monthly mean \pm 95% CI curved carapace length (CCL) of loggerhead turtles, (b) mean \pm 95% CI proportion of annual occurrence of strandings in each month, (c) proportion of each monthly total assessed as having interacted with longline fisheries

Each of the anthropogenic causes of strandings other than longline fishing constituted less than 10% of the records. However, combined they accounted for a total of 145 strandings (23.9% of all records). Other fisheries that were recorded interacting with sea turtles accounted for 5.3% of records with known stranding cause.

In addition, 9% of stranded turtles in 2006 presented evidence of boat strikes.

DISCUSSION

Interaction with fisheries is a matter of concern for sea turtle conservation in the Mediterranean. Lewison et al. (2004a) suggest that the Mediterranean is a global hotspot for longline fishery interaction with loggerhead turtles. These authors estimated that 60 000 to 80 000 loggerhead turtles were captured by pelagic longline fisheries in the Mediterranean Sea in 2000. However, their analysis is based on homogeneous bycatch rates across the region, which may not be the case (Camiñas et al. 2006, Báez et al. 2007). Estimates of turtles captured yearly by the Spanish longline fishery, the largest of several fishing fleets in the western Mediterranean range, put loggerhead bycatch at between 2000 and 20 000 individuals (Laurent et al. 2001, Mejuto et al. 2006 and references therein).

Our analyses suggest that, at minimum, half of the total number of turtle strandings over the period of our study were caused by interactions with human activities at sea, mainly fisheries. This number is consistent with other reports elsewhere in the Mediterranean (Casale et al. 2007). Elsewhere in Spain, 48% of 147 stranded loggerhead turtles found alive along the coast of Andalusia in 2000 showed clear signs of interaction with fishing gear, 73% of these were demonstrated to have interacted with longline fisheries (Aula del Mar 2000).

For the Pacific Ocean, Lewison et al. (2004b) estimated that adult turtles have an approximately 50% chance of being hooked or entangled in longline fishing gear once each year. When a turtle is hooked by longline gear, depending on where the hook is embedded, they can face substantial risks. Most often, hooked turtles are not hauled aboard as fishermen often cut the line as soon as they are able to identify the species captured. If the hook and line are ingested, they may produce intussusception and other gut pathologies that can be lethal (Oros et al. 2005). However, data on post-hooking mortality, the likelihood that a turtle hooked by longline gear will die after it is released, are sparse. Although pre-release mortality in the western Mediterranean has been found to be very low, and most of the turtles are captured alive (Carreras et al. 2004, Camiñas et al. 2006), there have been few assessments of post-release mortality (Chaloupka et al. 2004). For turtles captured by longline fisheries in the Mediterranean, Casale et al. (2007) propose post-release mortality $\geq 30\%$, underlining the potentially important impact that longline bycatch may have on marine turtle stocks regionally.

The analysis of size class distribution of stranded loggerhead turtles demonstrates several interesting features. First, we observed that longline bycatch appears to be more prevalent in large juvenile and adult-sized

turtles—age classes which may have a higher reproductive value and are therefore of great importance for the conservation of the species (Crowder et al. 1995, Lewison & Crowder 2007). Similar size range has been reported in longline bycatch studies in the northeast Spanish Mediterranean (Pont & Alegre 2000, Camiñas & Valeiras 2001) and in other parts of the Mediterranean (Deflorio et al. 2005). This suggests that large juvenile loggerheads in the western Mediterranean exploit pelagic habitats (Laurent et al. 1998, Revelles et al. 2007a) and may not adhere strictly to the typical ontogenic pattern of habitat use proposed for the species (Bolten 2003). This is supported by the importance of pelagic feeding in large juvenile loggerhead revealed in dietary studies in the western Mediterranean (Tomás et al. 2001, Revelles et al. 2007b). Moreover, satellite tracking studies of this species have now also highlighted pelagic habitat use by large juveniles (Revelles et al. 2007a, McClellan & Read 2007) and adults (Hatase et al. 2002, Hawkes et al. 2006), indicating an ever-more complicated picture of multiple life-history strategies (Witzell 2002, Godley et al. 2008).

Longline fisheries extend throughout the Spanish Mediterranean and surrounding waters (Báez et al. 2006, Mejuto et al. 2006, Camiñas et al. 2006). However, we recorded fewer strandings in the northern province (Castellón). This could be explained by the surface currents pattern proposed by Millot (1999; see our Fig. 1) and strandings distribution may be driven by the dominant north-south current along the Spanish coast. The perpendicular orientation of Cape La Nao to the main surface current would facilitate the concentration of strandings at the northern aspect of this cape. The southern part of the study area is west of a very important fishing zone known as 'El Patíbulo' (located approximately between 38°00' N, 00°30' E and 38°30' N, 01°00' E), which is subject to high fishing effort not only by longliners, but also by trawlers and other gears. This concentration of fishing activity may be responsible for the level of human-related strandings in Alicante province.

In the Spanish Mediterranean, swordfish longline fisheries operate from spring to autumn, but concentrate effort in summer months (June to September; Pont & Alegre 2000, De la Serna et al. 2004, Camiñas et al. 2006). Based on the seasonality of fishing effort, it is not surprising that stranding of turtles, particularly of turtles showing signs of longline interaction, mirrors this seasonality. De la Serna et al. (2004) reported a slight increase of fishing effort and a slight decrease of total captures of target species for the period 1998 to 2001 in the Spanish longline fishing fleet. A modification of the longline fishing gear introduced in 2003 in the fleet resulted in a considerable reduction of mean number of hooks per vessel (Báez et al. 2006, Camiñas

et al. 2006). A considerable reduction in total captures of swordfish and tuna (both species targeted by longline) in the water surrounding the study area in 2004 and 2005 are reported compared to the period 1994 to 2003 (Generalitat Valenciana 2005). This is consistent with the general down-sizing of the whole Spanish fishing fleet and the reduction of captures of almost all fish species in the recent years, both in the studied region (Generalitat Valenciana 2005) and in the rest of Spain (MAPA 2007). This information would possibly explain the reduction of the impact of this fishery on loggerhead turtles in recent years. However, it is as yet too early to establish long-term trends in the fishery and its impact on sea turtles in the area. The possibility exists that some high seas fisheries will decline, perhaps due to overexploitation, before effective bycatch mitigation can be enacted.

Other fisheries

Other types of fishing gear, such as trawlers, static nets or trammel nets, are responsible for bycatch of non-targeted marine vertebrates in the Mediterranean (Carreras et al. 2004, Casale et al. 2005, Karamanlidis et al. 2008, this Theme Section [TS]), and further afield (Epperly et al. 2007, Slooten 2007, Chilvers 2008, this TS). Compared to longlines, relatively little research effort has been directed at assessing the impact of other such gear on sea turtles (Godley et al. 1998, Casale et al. 2005). Interaction with other fisheries has been highlighted in areas near the studied region (Bertolero 2003, Carreras et al. 2004). It is possible that the impact of other kinds of fishing methods, such as trawling, will be underestimated in the studied area, since assessment studies are scarce and the available information comes only from vessel logbooks and occasional reports at a regional level.

Insights into natural history

Most stranded individuals were within the ranges of typical juvenile sizes reported for the Mediterranean (Margaritoulis et al. 2003) and the Atlantic populations (Bolten 2003). However, adults may also be present in our sample; in the Mediterranean, nesting females of less than 65 cm CCL have been reported (Broderick et al. 2003, Margaritoulis et al. 2003 and references therein). Similar size distributions have been found in strandings in Catalonia, north of our study area (Pont & Alegre 2000, Bertolero 2003). Although genetic studies suggest that turtles either side of Cape La Nao are from different stocks (Carreras et al. 2006), we did not find a difference in size of turtles north and south of

Cape La Nao. This supports the findings of Carreras et al. (2006), that there were no size differences between turtles with Atlantic and Mediterranean haplotypes in the Spanish Mediterranean.

North of Cape La Nao, we report a decrease in number of strandings with increasing latitude. However, at-sea studies have reported a high density of turtles in the waters surrounding the northern part of the study area throughout the year (Gómez de Segura et al. 2003, 2006). Furthermore, no differences in density were found north and south of Cape La Nao in aerial surveys performed between 2001 and 2003 (Gómez de Segura et al. 2006). Thus, abundance and distribution of strandings are more likely linked to interaction with fisheries and the predominant current patterns than to the abundance and distribution of regional stocks. However, in some cases stranding data can help in the detection of dramatic variations in the composition of regional stocks of marine vertebrates and changes in their seasonal distribution patterns (e.g. Aguilar 2000). For instance, the substantial increase in strandings in June 2001 was likely related to a massive influx of turtles into the study area (Tomás et al. 2003a).

The present study provides data on turtle strandings throughout the year, but the majority of these strandings occur in the spring, summer and early autumn (between April and October). This is consistent with other studies on the Spanish Mediterranean coast (e.g. Bertolero 2003). However, aerial surveys have shown no differences in density between seasons (with the exception of spring 2001, Gómez de Segura et al. 2006). Here, the seasonality in strandings is most likely explained by the increase of fishing activity in summer months (Carreras et al. 2004, Camiñas et al. 2006). The fact that no significant seasonality has been detected in cetacean stranding pattern, also recorded by the same network in the same region (University of Valencia Marine Zoology Unit unpubl. data), supports this explanation. Cetaceans exhibit substantially lower levels of longline fishery interactions than turtles.

CONCLUSIONS

Compared to studies at nesting beaches, studies of sea turtles at sea are subject to many methodological and economic constraints. Data from stranding networks can be biased due to seasonal and spatial variation in recording, and interannual variation in surveying (see Hart et al. 2006 and Witt et al. 2007). Despite these limitations, long-term stranding studies can give insight into sea turtle biology, population dynamics and, particularly, threats at sea. However, in order to provide accurate information, stranding networks must employ repeatable protocols and constant search effort

over designated areas. Although the numbers of records in the earlier years of our study period were likely influenced by changes in detection and search effort, our data set provides valuable information on anthropogenic mortality sources in this region.

Fisheries and other disturbances in the study region affect turtles from different population stocks: the Atlantic and the Mediterranean (Laurent et al. 1998, Carreras et al. 2006 and references therein). Because the Mediterranean stock is far smaller than the Atlantic stock (see Broderick et al. 2002, Margaritoulis et al. 2003), a more detailed assessment of turtle–fishery interaction in the Mediterranean that is able to identify stock source of affected individuals is essential.

Interactions with longlines were found to be the main cause of observed turtle mortality. However, we also found evidence that the potential impact of longlining on western Mediterranean loggerhead turtles may have decreased in recent years. Future integrated studies, including at-sea surveys of sea turtle abundance, fisheries effort through direct or indirect methods (Witt & Godley 2007), together with stranding records collected by an effective and standardized network, will help to offer insight into the biology of sea turtles and the role of different anthropogenic threats on this species in the area.

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