

## ***Biomass removal by dolphins and fisheries in a Mediterranean Sea coastal area: do dolphins have an ecological impact on fisheries?***

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### ABSTRACT

1. Dolphins are often claimed to compete with fisheries, including through removal of substantial biomass. To calculate the biomass removed by fisheries and the degree of resource overlap with dolphins in a coastal area of Greece, estimates of dolphin abundance based on photographic capture–recapture were combined with an assessment of fishing effort and catch.

2. The estimated total biomass consumed annually by local dolphin populations – 15 short-beaked common dolphins and 42 common bottlenose dolphins – was 15.5 and 89.8 tonnes, respectively. The total biomass removed by the local fishing fleet (307 fishing boats) was 3469.2 tonnes, i.e. about 33 times greater than that removed by dolphins.

3. Dolphins removed 2.9% of the total biomass, fisheries 97.1%. Nine purse seiners (representing only 3% of the active fishing fleet) were responsible for 31.9% of biomass removal. Similarity of biomass composition between dolphins and fisheries was expressed by a Pianka index of 0.46 for common dolphins and 0.66 for bottlenose dolphins.

4. Overlap differed according to fishing gear. Common dolphin overlap was higher with purse seiners (0.82), and lower with beach seiners (0.31), bottom trawlers (0.11) and trammel boats (0.06). There was virtually no overlap with longliners (0.02). Bottlenose dolphin overlap was higher with trammel boats (0.89) and bottom trawlers (0.75), and lower with longliners (0.38), purse seiners (0.24) and beach seiners (0.18). There was minimal overlap (0.12) between the two dolphin species.

5. This study suggests that ecological interactions between dolphins and fisheries in this coastal area have minor effects on fisheries. Conversely, prey depletion resulting from overfishing can negatively affect dolphins. Fisheries management measures consistent with national and EU legislation are proposed to ensure sustainability and to protect marine biodiversity. Copyright © 2010 John Wiley & Sons, Ltd.

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### INTRODUCTION

There has been much debate as to whether dolphins compete with fisheries and reduce fishery landings. While direct (or ‘operational’) interactions involving depredation of fishing gear by dolphins are relatively well documented (Northridge, 1991; Read, 2005), the claim that dolphins negatively affect fishery yields through indirect (also termed ‘biological’ or ‘ecological’) interactions is mostly unsupported. To date, no robust scientific investigation confirms that present-day dolphin populations reduce fishery catches by removing biomass that would otherwise be available to fishermen

(Punt and Butterworth, 1995; Northridge and Hofman, 1999; Lavigne, 2003; Plaganyi and Butterworth, 2005).

Ecosystem damage resulting from overfishing and habitat degradation in the Mediterranean Sea (Briand, 2000) has likely exacerbated the perception that dolphins reduce fishery yields (Reeves *et al.*, 2001). Dolphins in Mediterranean coastal waters are often blamed for ‘competing’ with fisheries and deemed responsible for reduced catches. This attitude generates conflict and intentional kills may occur in retaliation, with potentially significant impact on local populations of dolphins living in continental shelf waters, particularly short-beaked common dolphins *Delphinus delphis*

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and common bottlenose dolphins *Tursiops truncatus* (hereafter common dolphin and bottlenose dolphin, respectively; Bearzi *et al.*, 2004a, 2008c). It is therefore important to discriminate between depredation of fishing gear and generalized claims of dolphins reducing fishery yields. Depredation in Mediterranean coastal waters normally involves bottlenose dolphins and it affects primarily gill and trammel nets, with an overall economic impact that, when carefully assessed, may be modest even within areas of acute conflict (Lauriano *et al.*, 2004; Brotons *et al.*, 2008), with some exceptions (Bearzi *et al.*, 2008c). When it comes to competition, the available evidence suggests that Mediterranean dolphins are unlikely to affect coastal fishery yields to a significant extent (Coll *et al.*, 2007).

Conversely, prey depletion resulting from overfishing is seen as one of the main ongoing threats to dolphins living in Mediterranean coastal waters (Bearzi *et al.*, 2003, 2008b, c; Cañadas and Hammond, 2008). Depletion of dolphin prey is worrying, considering the threatened status of coastal dolphin populations. Common dolphins – once one of the most common cetacean species in the Mediterranean Sea – have declined throughout the region since the 1960s (Bearzi *et al.*, 2003). Their Mediterranean population is listed in Appendix I and II of the Convention on the Conservation of Migratory Species (Bonn Convention - CMS) and classified as Endangered in the IUCN Red List of Threatened Animals. Bottlenose dolphins are also thought to have declined in the Mediterranean, and their population is listed in Appendix II of CMS and considered Vulnerable according to IUCN Red List criteria (Bearzi *et al.*, 2008c).

To contribute to the identification of appropriate fisheries management and marine conservation action, we provide a quantitative estimate of the total biomass removed by cetaceans and fisheries in a coastal area, and evaluate the trophic overlap between fisheries and the only two cetacean species regularly found in the Inner Ionian Sea Archipelago, Greece. This area is considered of high conservation importance, being included by the Hellenic Ministry of the Environment in the Natura 2000 network under the European Commission (EC) Habitats Directive. The aim of the Natura 2000 network is to assure the long-term survival of Europe's most valuable and threatened species and habitats. In addition, the Agreement on the Conservation of Cetaceans in the Black Sea, Mediterranean Sea and contiguous Atlantic area (ACCOBAMS), ratified by Greece, identified this area as one where conservation and management actions should be developed and implemented immediately (Bearzi *et al.*, 2004b). The waters of the Inner Ionian Sea Archipelago are an important spawning site for European pilchard *Sardina pilchardus* (Somarakis *et al.*, 2006a, b; Machias *et al.*, 2007) and European hake *Merluccius merluccius* (Politou *et al.*, 2006; Politou, 2007), making this area a candidate for special protection based on EC Regulations for the sustainable exploitation of fishery resources in the Mediterranean (EC, 2006). Moreover, the area is home to species included in Annex II to the Habitats Directive, such as bottlenose dolphins, Mediterranean monk seals (*Monachus monachus*) and loggerhead sea turtles (*Caretta caretta*; Bearzi *et al.*, 2006), providing further basis for conservation action.

Preliminary information on biomass removal by fisheries and common dolphins was included in a former contribution (Bearzi *et al.*, 2008b). Here, we expand and complement that information by: (1) estimating the number of bottlenose

dolphins, the biomass removed by these animals and their degree of trophic overlap with fisheries; (2) employing a substantially improved method to estimate the body mass of both dolphin species and their daily food intake; and (3) using a more homogeneous and larger sample of fisheries landings.

## METHODS

### Study area

The study area situated in eastern Ionian Sea coastal waters covers approximately 1100 km<sup>2</sup> of sea surface (Figure 1). The sea floor is mostly 50–250 m deep, with rocky coasts and shallows covered by *Posidonia* meadows. Waters are transparent and oligotrophic (Gotsis-Skretas and Ignatiades, 2007), and river and agricultural runoff is insignificant (Pitta *et al.*, 1998).

### Dolphin abundance estimates

Surveys at sea were conducted from a 5.8 m inflatable craft with rigid hull powered by 100 HP four-stroke outboard engine. The survey coverage totalled 2846 km under favourable conditions ('on effort', Bearzi *et al.*, 2008b) and 59 survey days, from June to September 2007. Common dolphin numbers were estimated based on field counts, as described in Bearzi *et al.* (2008b), capture–recapture analyses being precluded by low density of animals and insufficient recaptures. Bottlenose dolphin numbers in the same area were estimated through photographic capture–recapture. Photo-identification was performed following Würsig and Jefferson (1990), with digital cameras equipped with 70–200 mm f2.8 zoom lenses. Individual identification relied on long-term natural marks on the dolphins' dorsal fins (Würsig and Würsig, 1977), based on high-resolution digital colour photographs. On each encounter with a group of dolphins, as many photographs as possible were taken of all individuals present. Photos were selected using consistent criteria (i.e. entire dorsal fin visible, fin perpendicular to camera, high sharpness and resolution, no water spray masking fin profile), based on recommendations provided by Read *et al.* (2003). Following such selection, the catalogue included 3903 dorsal fin photos. These images were then matched and the identified animals included in a database. Only individuals with distinctive dorsal fin profiles, bearing marks suitable for reliable long-term identification of either side of the fin, were used for capture–recapture analyses. Patterns of site fidelity implied by photo-identification data assisted in the selection of the most appropriate model for capture–recapture analyses.

Schwarz and Arnason's parameterization of the Jolly–Seber open population model (Schwarz and Arnason, 1996), with data pooled by month, was used to estimate bottlenose dolphin numbers using Program MARK 5.1 (White and Burnham, 1999). This model provides abundance estimates while allowing entries and losses in the population under study, consistent with the available information (Bearzi *et al.*, 2005). Four conditional forms of the Jolly–Seber model were fitted to the data, and the appropriate model for inference was selected using the Akaike Information Criterion corrected for small-sample sizes (AICc; Burnham and Anderson, 1998). As capture–recapture estimates relied on natural markings to

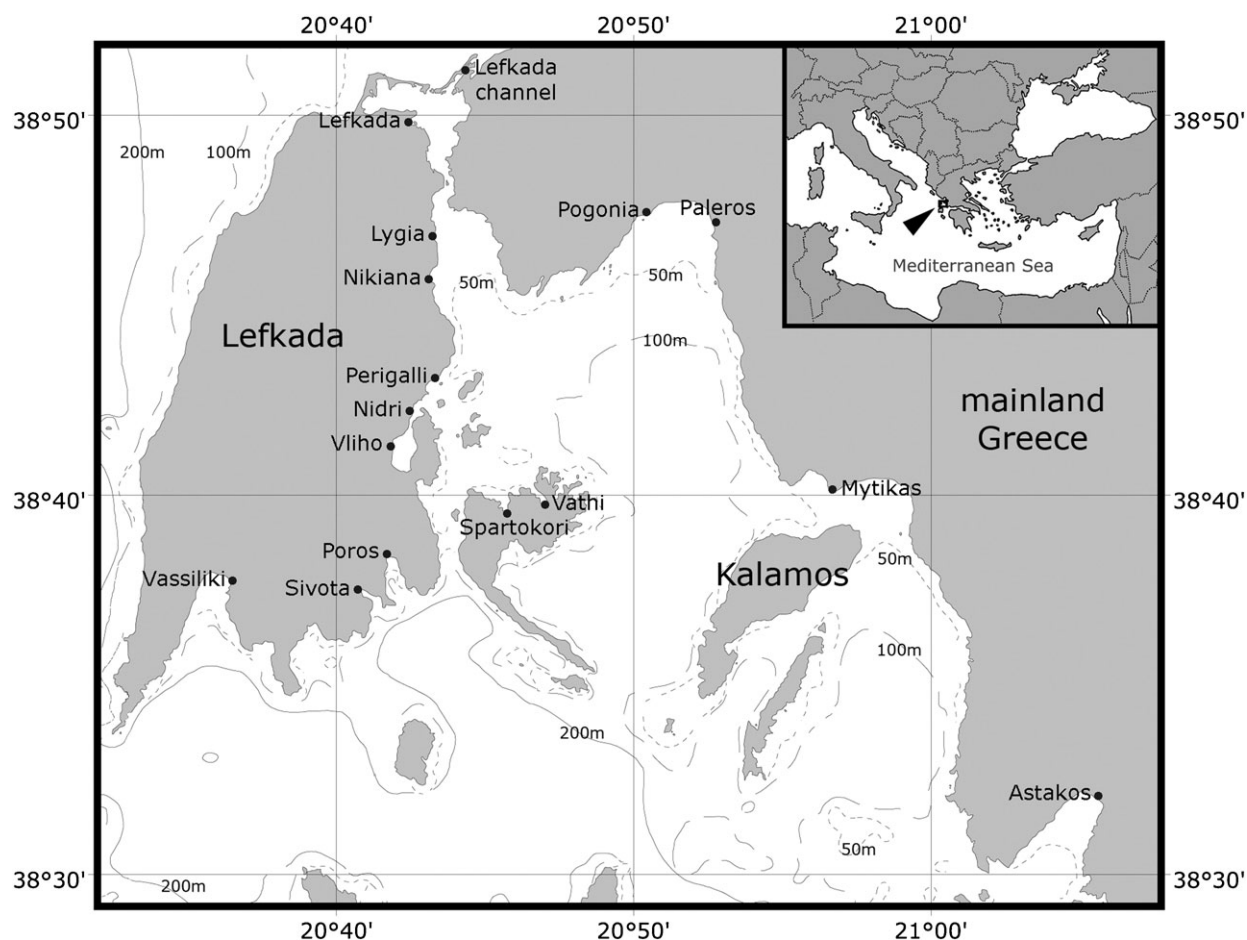


Figure 1. Study area with bathymetric contours lines and locations of fishing ports and other mooring/landing sites monitored on a monthly basis. The location of the study area relative to the eastern Mediterranean basin is shown in the inset.

identify individuals, they refer exclusively to the population of marked animals. To include the unmarked portion and estimate total abundance, the proportion of unmarked individuals (which also included calves) was computed based on the number of photographs of marked and unmarked dorsal fins obtained daily (Williams *et al.*, 1993; Bearzi *et al.*, 2008a, b).

### Monitoring of fishing effort and landings

Between January and December 2007, the local fishing fleet was monitored in all 16 ports and other mooring and landing sites within the study area shown in Figure 1, once per month on days of bad sea state conditions and/or on Sundays (i.e. on days with a high probability that the whole fishing fleet would be moored). A total of 343 fishing boats were catalogued, including 'active' and inactive boats (see below). Classification of fishing boats and gear was based on Nedélec and Prado (1990). The activity status of each boat was recorded visually every month based on fishing gear on board or near the boat's mooring place, fishermen working on board, boat conditions, and direct inquiries. The local fishing fleet – i.e. the boats recorded as 'active' in any given month during the study period – included 12 purse seiners, 24 beach seiners, 9 bottom trawlers, 49 longliners, and 213 trammel boats (Table 1). These numbers include the entire resident fishing fleet operating in

the area shown in Figure 1 (a reasonable proxy of fishing effort, considering that fishing in Greece is preferably conducted near home ports to reduce operating costs while obtaining higher profits; Tsitsika and Maravelias, 2008). Boats shorter than 4 m ( $n = 21$ ) were not considered as their impact was assumed negligible based on direct observations of their landings. For the same reason, 15 boats 4–7 m long equipped with gas-powered lamps and tridents/harpoons were also excluded from the analyses. Some boats had multiple gears, and switched from one fishing method to another depending on seasonal closures. For instance, most beach seiners operated as trammel boats during months of beach seining closure. Therefore, 'active' boats were classified according to the gear used in each month of sampling (Table 1).

Landings of purse seiners, beach seiners and bottom trawlers were monitored between January and December 2007. In total, 24 landings were recorded for purse seiners (mean = 2.7 per month of activity,  $SD = 2.00$ ), 21 for beach seiners (3.5,  $SD = 3.56$ ), and 16 for bottom trawlers (2.0,  $SD = 1.07$ ). Total catch by species was recorded by trained researchers based on number of full boxes landed (boxes were routinely divided by species before landing). Full boxes were estimated to average 10 kg irrespective of species (an assumption routinely used for market purposes at landing sites, confirmed by direct observations). Days of activity per month per boat for purse seiners, beach seiners and bottom

Table 1. Fishing fleet operating in the study area: boat kind, mean number of active boats, months of activity recorded between January and December 2007, and boat length

Boat kind	Mean number of active boats	Number of active months	Boat length (m)
Purse seiner	7.0	9.5	15–25
	1.0	6	
	1.0	5	12
Beach seiner	24.0	6	8–12
Bottom trawler	5.9	8	20–25
Trammel	1.0	6	
	22.7	12	4–5
	48.8	12	5–7
	14.5	6	7–10
	82.5	12	
	2.3	6	10–12
Longliner	2.3	12	
	16.4	12	4–7
	10.0	12	7–9
	4.3	12	9–10

For instance, the first line means that – on average – 7 purse seiners of 15–25 m were actively fishing for a total of 9.5 months, one purse seiner of 15–25 m was actively fishing for 6 months, and one purse seiner of 12 m was actively fishing for 5 months. Pooling of length categories followed literature information.

trawlers were based on Kapantagakis *et al.* (2001), also taking into account periods of seasonal fishing closure (Kapantagakis, 2007). Average catch for trammel boats and longliners was estimated as 3.42 tonne yr<sup>-1</sup> for boats 4–7 m long, and 6.31 tonne yr<sup>-1</sup> for boats 8–1 m long (Stergiou *et al.*, 2007c). Percentage catch contribution of the most abundant species for longliners and trammel boats operating in the Ionian Sea was based on Stergiou *et al.* (2007a). Discard rates were assumed to be 14% for purse seiners (Tsimenides *et al.*, 1995), 28% for beach seiners (Stergiou *et al.*, 1996), 39% for bottom trawlers (Machias *et al.*, 2001) and 9.8% for netters and longliners (Stergiou *et al.*, 2002; Tzanatos *et al.*, 2007). Biomass removed annually by fisheries, calculated separately according to fishing gear, was the product of average daily catch per boat (including discards), days of activity per month, number of boats recorded as ‘active’ in each month of sampling and months of fishing activity.

## Resource overlap

The Pianka niche overlap index (a measure of qualitative resource overlap between two species; Pianka, 1973; Pusineri *et al.*, 2004; Bearzi *et al.*, 2008b) was used to assess the similarity of biomass composition between dolphin prey and fishery catches, and between the prey of the two dolphin species:

$$\alpha = \frac{\sum_i P_{ij} \cdot P_{ik}}{\sqrt{\sum_i (P_{ij})^2 \cdot \sum_i (P_{ik})^2}}$$

where  $P_{ij}$  is the percentage of prey item  $i$  of predator  $j$ , and  $P_{ik}$  is the percentage of prey item  $i$  of predator  $k$ . The index ranges between 0 and 1, and the similarity is higher the closer the index is to 1.

While the precise diet composition of dolphins in the study area is not known, their dietary preferences were inferred from: (1) information on common dolphin and bottlenose dolphin diet in neritic Mediterranean habitats (reviewed in Bearzi *et al.*, 2003, 2008c); (2) observations of dolphin feeding behaviour and diving patterns in the study area (Bearzi *et al.*, 2005; Tethys Research Institute unpublished data); (3) analysis of fish scales

sampled during surface feeding by common dolphins in the study area (Bearzi *et al.*, 2006); and (4) stomach contents of dead animals in the study area (Bearzi, 2000; Tethys Research Institute unpublished data). Based on this information, the average diet of common dolphins was assumed to be composed of 80% Clupeidae and Engraulidae, 5% small Carangidae and Belonidae, 5% Gadidae, 5% cephalopods, and 5% other families. The average diet of bottlenose dolphins was assumed to be composed of 35% Merluccidae, Mullidae and Gadidae, 20% Sparidae, Mugilidae and Moronidae, 15% Congridae, 15% cephalopods, 2.5% Clupeidae, Engraulidae and Scombridae, 2.5% small Carangidae and Belonidae and 10% other families. ‘Cephalopods’ included primarily Loliginidae, Octapodidae, Ommastrephidae, Sepiolidae and Sepidae.

To account for the controversies surrounding the most appropriate methods to estimate food consumption by marine mammals (Kaschner, 2004; Leaper and Lavigne, 2007), average daily food consumption of common dolphins and bottlenose dolphins was estimated using four different methods:

- (1)  $IB = 0.123M^{0.8}$  (Innes *et al.*, 1987)
- (2)  $IB = 0.482M^{0.524}$  (Leaper and Lavigne, 2002; Kaschner, 2004)
- (3)  $IB = 0.035M$  (Tamura, 2003; Kaschner, 2004)
- (4)  $IB = 0.1M^{0.8}$  (Trites *et al.*, 1997)

where  $IB$  is the ingested biomass (kg day<sup>-1</sup>) and  $M$  the body mass in kg. The results of these estimates were averaged, and the confidence interval was expressed as two times the standard deviation.

Adult body mass was computed using the following formulae, where  $L$  is the body length (in cm):

Body mass of common dolphin (kg) =  $7.5814(L-140)^{0.5345}$  (Kastelein *et al.*, 2000)

Body mass of bottlenose dolphin (kg) =  $17.261e^{0.0156(L-100)}$  (Kastelein *et al.*, 2002)

Body lengths of adult individuals stranded along the Ionian Sea coasts of Italy (i.e. a few hundred kilometres from the study area) were extracted from the Italian Strandings Database (<http://mammiferimarini.unipv.it/>). Body lengths below the mean body length minus SD (i.e. common dolphins below 1.39 m and bottlenose dolphins below 1.66 m) were not included in the computation, as these values probably referred to young animals including nursing individuals. The median adult length was 1.98 m for common dolphins (mean = 1.94, 95% CI = 1.87–2.02,  $n = 26$ ) and 2.40 for bottlenose dolphins (mean = 2.43, 95% CI = 2.29–2.57,  $n = 35$ ). This resulted in an average body mass of 66.4 kg for common dolphins (CI = 55.2–70.6) and 153.3 kg for bottlenose dolphins (CI = 93.1–244.8). Average per-capita daily food consumption was therefore 3.7 kg for common dolphins (CI = 2.4–4.1) and 6.1 kg for bottlenose dolphins (CI = 5.4–6.9).

## RESULTS

### Dolphin abundance

The model that best fitted the data was model phi(t)p(.) in which capture probabilities are constant and survival

probabilities vary over time, also model  $\phi(\cdot)p(\cdot)$  and model  $\phi(\cdot)p(t)$  provided good descriptions of the data (Table 2) and therefore model averaged estimates were used (Anderson, 2008). Capture–recapture analyses resulted in a point estimate of 30 marked animals (95% CI = 27.4–31.8). The mean proportion of unmarked bottlenose dolphins in the

population was 0.28 (95% CI = 0.19–0.37,  $n = 31$ ). A total population size of 42 animals was therefore obtained by adding this mean proportion of unmarked individuals to estimates based on marked animals. Abundance of common dolphins in 2007 was estimated in a previous study as 15 animals (Bearzi *et al.*, 2008b). Two bottlenose dolphin and two common dolphin calves were excluded from the calculation of biomass removal.

Table 2. Selection of appropriate open-population model using the Akaike Information Criterion (AICc) corrected for small sample sizes

Model	AICc	$\Delta$ AICc	AICc weight	$np$
$\phi(t)p(\cdot)$	98.47	0	0.433	6
$\phi(\cdot)p(\cdot)$	99.70	1.23	0.234	5
$\phi(\cdot)p(t)$	100.15	1.68	0.188	6
$\phi(t)p(t)$	100.65	2.18	0.145	7

$\Delta$ AICc = difference between AICc and minimum AICc obtained; AICc weight = relative weight or strength of the model, used for model averaging if models are very similar in weight;  $np$  = number of parameters in the model.

### Impact of fishing and resource overlap

Table 3 reports the mean landings and catch composition of purse seiners, beach seiners and bottom trawlers. Estimates of the total biomass removed by the local fishing fleet by fishing gear and aggregated catch are shown in Table 4, taking into account discards and days of inactivity (see Methods). The estimated total biomass removed by local fisheries was 3469.2 tonnes. An average of nine active purse seiners (representing only 3% of the total active fleet) were responsible for 31.9% of

Table 3. Mean biomass per landing of purse seiners, beach seiners and bottom trawlers, and their species composition

Boat kind	Biomass per landing (kg)	SE	$n$	Range (kg)	Species landed	Percentage of total catch	
Purse seiners	754.6	202.76	24	0–4030	Atlantic horse mackerel	<i>Trachurus trachurus</i>	26.6
					European pilchard	<i>Sardina pilchardus</i>	23.0
					bogue	<i>Boops boops</i>	15.4
					round sardinella	<i>Sardinella aurita</i>	11.3
					Atlantic mackerel	<i>Scomber scombrus</i>	10.6
					picarel	<i>Spicara smaris</i>	7.6
					Atlantic bonito	<i>Sarda sarda</i>	1.9
					European squid	<i>Loligo vulgaris</i>	1.7
					European barracuda	<i>Sphyraena sphyraena</i>	1.0
					skipjack tuna	<i>Katsuwonus pelamis</i>	0.8
Beach seiners	108.8	19.01	21	29–410	picarel	<i>Spicara smaris</i>	62.2
					European pilchard	<i>Sardina pilchardus</i>	20.8
					bogue	<i>Boops boops</i>	5.9
					European squid	<i>Loligo vulgaris</i>	5.7
					European barracuda	<i>Sphyraena sphyraena</i>	1.5
					striped red mullet	<i>Mullus surmuletus</i>	1.1
					common pandora	<i>Pagellus erythrinus</i>	0.6
					annular seabream	<i>Diplodus annularis</i>	0.1 to 0.4
					gilthead seabream	<i>Sparus aurata</i>	
					white seabream	<i>Diplodus sargus</i>	
greater amberjack	<i>Seriola dumerili</i>						
Atlantic mackerel	<i>Scomber scombrus</i>						
salema	<i>Sarpa salpa</i>						
common cuttlefish	<i>Sepia officinalis</i>						
saddled seabream	<i>Oblada melanura</i>						
European seabass	<i>Dicentrarchus labrax</i>						
Bottom trawlers	508.5	88.27	16	195–1510	European hake	<i>Merluccius merluccius</i>	34.2
					Atlantic horse mackerel	<i>Trachurus trachurus</i>	18.6
					European squid	<i>Loligo vulgaris</i>	14.8
					blue and red shrimp	<i>Aristeus antennatus</i>	10.8
					striped red mullet	<i>Mullus surmuletus</i>	8.2
					bogue	<i>Boops boops</i>	4.2
					thornback ray	<i>Raja clavata</i>	2.2
					angler	<i>Lophius piscatorius</i>	1.7
					silver scabbardfish	<i>Lepidopus caudatus</i>	1.0
					<i>Octopus</i> sp.		1.0
					common pandora	<i>Pagellus erythrinus</i>	0.7
					spottail mantis shrimp	<i>Squilla mantis</i>	0.6
					European seabass	<i>Dicentrarchus labrax</i>	0.5
					greater amberjack	<i>Seriola dumerili</i>	0.5
					small-spotted catshark	<i>Scyliorhinus canicula</i>	0.1 to 0.4
					flathead mullet	<i>Mugil cephalus</i>	
gilthead seabream	<i>Sparus aurata</i>						
John dory	<i>Zeus faber</i>						
common cuttlefish	<i>Sepia officinalis</i>						

the total annual biomass removal. The estimated biomass consumed by common dolphins and bottlenose dolphins was 15.5 tonnes (CI = 11.4–19.6) and 89.8 tonnes (CI = 78.4–101.1), respectively, totalling 105.3 tonnes. Of the total biomass removed, cetaceans took an estimated 2.9%, fisheries 97.1% (Table 4, Figure 2). The likely similarity of biomass composition between dolphins and fisheries in the study area was expressed by an average Pianka index of 0.46

for common dolphins and 0.66 for bottlenose dolphins. There was minimal overlap (0.12) between the two dolphin species. Overlap differed considerably according to fishing gear (Table 5). It should be noted that these indices are based on the assumed percentage of prey in dolphin diet and on the observed percentage of various fish classes in fisheries landings. This does not deal with the absolute amounts of biomass removal. In other words, fisheries take much more fish than

Table 4. Estimated annual biomass removal (tonnes) of different fishing gear and the two dolphin species (Dd = common dolphins, Tt = bottlenose dolphins). Percentage of total removal is shown in the last row

	Purse seiners	Beach seiners	Bottom trawlers	Trammel	Longliners	Subtotal fisheries	Dd	Tt	Subtotal dolphins	Total
Clupeidae, Engraulidae, Scombridae	449.91	75.83	0	0	0	525.74	12.40	2.24	14.64	540.38
Sparidae, Moronidae, Mugilidae	154.57	28.97	34.18	261.09	5.06	483.87	0	17.95	17.95	501.82
Small Carangidae, Belonidae	265.53	0	107.54	0	0	373.07	0.78	2.24	3.02	376.09
Merluccidae, Mullidae, Gadidae	0	3.96	245.34	305.37	24.01	578.68	0.78	31.42	32.20	619.88
Congridae	0	0	0	0	0	0	0	13.47	13.47	13.47
Cephalopods	15.73	21.61	91.87	79.68	0	208.89	0.78	13.47	14.25	223.13
Fisheries: other families	113.99	231.37	99.42	176.97	115.57	737.32	0	0	0	732.32
Fisheries discards	139.96	101.29	225.56	80.66	14.17	561.64	0.78	8.98	9.76	571.40
Total annual biomass removal (tonnes)	1139.69	463.03	803.91	903.76	158.81	3469.21	15.52	89.77	105.29	3574.50
Total annual biomass removal (%)	31.88	12.95	22.49	25.28	4.44	97.05	0.43	2.51	2.94	100.00

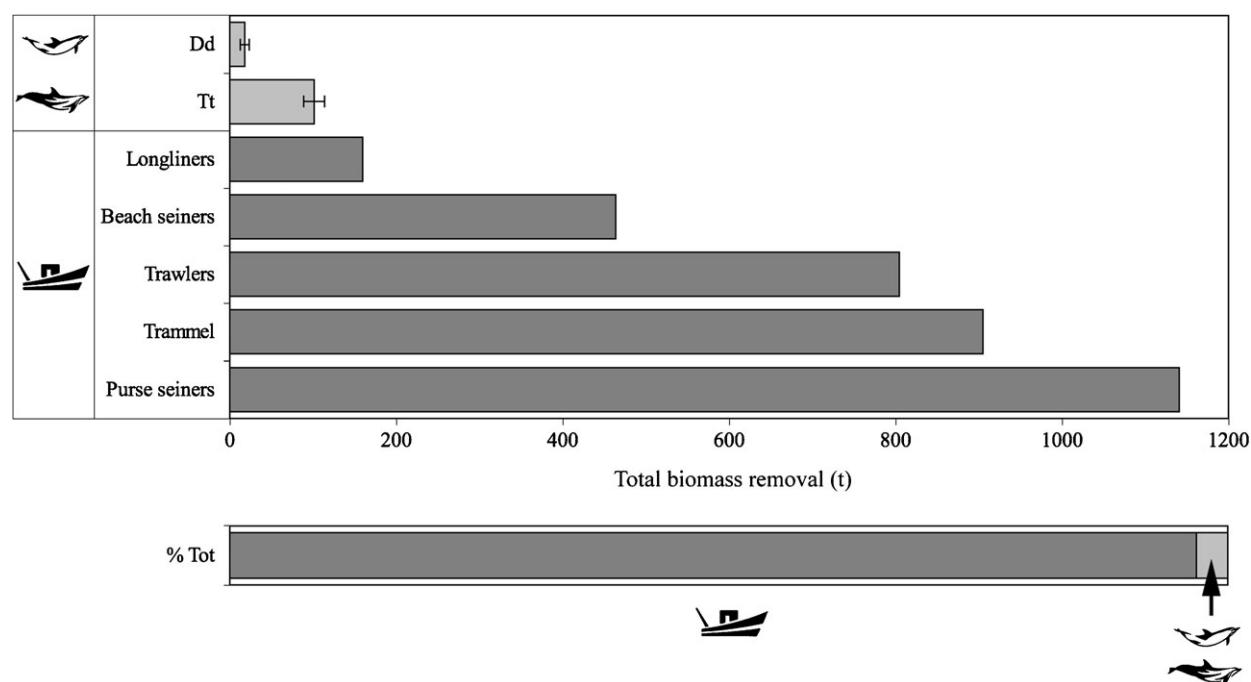


Figure 2. Estimated total biomass (tonnes) removed by common dolphins (Dd), bottlenose dolphins (Tt) and fisheries in the study area. Error bars for dolphins indicate SD resulting from the use of different methods to estimate food consumption by marine mammals. Percentage of total biomass removal (%Tot) is also shown: dolphins 2.9%, fisheries 97.1%.

Table 5. Pianka indices indicating the similarity of biomass composition between common dolphins (Dd), bottlenose dolphins (Tt) and fisheries operating in the study area. Trophic overlap between the two dolphin species is also shown

	Purse seiners	Beach seiners	Bottom trawlers	Trammel	Longliners	Total fisheries	Dd	Tt
Dd	0.82	0.31	0.11	0.06	0.02	0.46	—	0.12
Tt	0.24	0.18	0.75	0.89	0.38	0.66	0.12	—

dolphins but resource overlap, as expressed by the Pianka index, represents coincident species rather than amounts taken, which were separately considered in this study (Table 4).

## DISCUSSION

Excessive fishing pressure and the resulting worldwide decline in fish stocks and loss of marine biodiversity is a growing concern (Pauly *et al.*, 1998, 2002; Jackson *et al.*, 2001; Worm *et al.*, 2006). In the Mediterranean Sea, unsustainable fishing has resulted in profound direct and indirect ecological impacts (Sala, 2004), and has caused the decline of many fish stocks (Garcia *et al.*, 2005; Leonart, 2005). This is particularly true in continental shelf regions, where the large majority of fishing occurs (Roberts and Hawkins, 1999). Some of the overexploited fish stocks are important prey for dolphins living in Mediterranean coastal waters (Leonart, 2005), including common dolphins and bottlenose dolphins (Bearzi *et al.*, 2003, 2008c).

Overlap between dolphin prey and fishery target species does not imply direct competition (Briand, 2004). However, it is reasonable to infer competitive effects when key prey becomes scarce and remains subject to heavy fishing pressure (Trites *et al.*, 1997; Pusineri *et al.*, 2004). Under these conditions, not only are high degrees of overlap suggestive of ecological interactions, but these interactions may be particularly important when they involve threatened marine mammal species with restricted nearshore distribution and no easy access to alternative food sources (Plaganyi and Butterworth, 2005). While the assessment of trophic overlap is clearly a simple and preliminary approach to understanding complex food-web dynamics, alternative and more sophisticated modelling approaches are prone to bias and heavily reliant upon appropriate expertise as well as large reliable datasets (Plaganyi and Butterworth, 2005), which are simply unavailable in this and other Mediterranean cases. Scarcity of information on dolphin diet composition in the study area and more generally in the Mediterranean Sea called for assumptions that, albeit based on a comprehensive review of the available information, may affect the degree of overlap with some fishing gear. Conversely, figures of biomass removal by dolphins are probably robust, uncertainties about population size and energetic requirements being generally more important than uncertainty about diet composition when it comes to estimates of total food consumption (Pierce *et al.*, 2007).

In the Hellenic Seas, evidence of overfishing is ample and growing. The long-term increasing trends in Hellenic marine landings from 1964–1994, attributed to fleet modernization and geographic expansion of the fisheries over this period, have been followed since the mid-1990s by rapidly declining trends in landings and yields, suggesting that the fishing has been overly intense, i.e. unsustainable (Stergiou *et al.*, 1997, 2007b; Stergiou, 2005). In the eastern Ionian Sea, including the Inner Ionian Sea Archipelago, catch of both demersal and pelagic resources has declined steeply since the mid-1980s (Papaconstantinou *et al.*, 1988; Papaconstantinou and Stergiou, 1995; Stergiou *et al.*, 1997, 2007b). Considering the high and ongoing fishing pressure in the study area, a potential for 'exploitative competition' (Keddy, 1989; Plaganyi and Butterworth, 2005) exists between dolphins and local fisheries targeting their prey.

Information contributed by this study improves understanding of the current trophic impact of cetaceans and fisheries in a coastal Mediterranean ecosystem. This study shows that similarity of biomass composition between dolphin prey and catches by some fisheries was high (Table 5) and the total biomass removed by fisheries exceeded that removed by dolphins by a factor of 33 (Table 4). This suggests that (1) fisheries in the study area have by far the greatest impact on the ecosystem in terms of biomass removal, and (2) competitive effects are more likely to affect dolphins than fisheries. The disproportion of biomass removal (2.9% to dolphins and 97.1% to fisheries; Figure 3) further suggests that the damage caused by dolphins to fishery yields in terms of indirect (i.e. food-web) interactions is probably modest.

It must be noted that this study refers to a depleted population of common dolphins. Biomass removal by common dolphin in the mid-1990s, when their numbers were as high as 150 animals (Bearzi *et al.*, 2008b) can be assumed to have been an order of magnitude greater than in 2007. This would still represent a relatively low proportion of the total biomass removed by fisheries in 2007 (Figure 2). Figures of biomass removal by dolphins in the mid-1990s, however, should be compared with biomass removed by fisheries back then. There is no reliable information on biomass removal by fisheries in the study area in the mid-1990s, but fisheries landings throughout Greece are known to have peaked at that time (Stergiou, 2005; Stergiou *et al.*, 2007b). Bottlenose dolphins are not known to have declined in the Inner Ionian Sea Archipelago since the mid-1990s (Bearzi *et al.*, 2006).

While in the study area there were only nine active purse seiners (Table 1), this fishery had the greatest impact on fish biomass (Table 4). Purse seiners totalling 3% of the active fishing fleet and operating over a period up to 9.5 months were responsible for 31.9% of total biomass removal. High overlap was found between purse seine landings and common dolphin key prey, and this was linked to an observed 90% population decline of common dolphins over 10 years (Bearzi *et al.*, 2008b). Prey of bottlenose dolphins was primarily a target of trammel boats and bottom trawlers. The trammel fleet was considerably larger (173 active boats) and operated year round, except for 17 boats switching to beach seining for half of the year. Conversely, only seven bottom trawlers operated in the study area for up to 8 months. Overall, a relatively small number of purse seiners and bottom trawlers (16 boats, i.e. 6.6% of the total active fleet) were responsible for the majority (54.4%) of the total biomass removal. These two fisheries were held responsible for ecosystem damage and catch reduction by the local community of 'small-scale' fishermen. Illegal purse seining in shallow waters and on *Posidonia* beds was consistently reported by trammel fishermen and repeatedly observed by the authors across this study. Excess capacity of purse seiners is a recognized problem in Greece, resulting in stock depletion and reduced economic performance of the fleet; decreasing the purse seine fleet was therefore proposed as a reasonable management measure towards reducing overexploitation and attaining sustainability (Tsitsika *et al.*, 2008). Bottom trawling is known for its high discard rates (i.e. 39%, Machias *et al.*, 2001) and its negative impact on the seabed (Jones, 1992; Smith *et al.*, 2000). Landings of both purse seiners and bottom trawlers often included fish below the minimum legal size. Management measures targeting purse seiners and bottom trawlers would probably bring

considerable benefits to the ecosystem and allow for a more sustainable fishery. This study, however, shows that trammel boats and beach seiners also have a significant impact in terms of biomass removal (Figure 2). While beach seining had a relatively lower impact on dolphins in terms of resource overlap (Table 5), this fishery is an important cause of habitat degradation. Still legal in Greece at the time of writing, but banned in most EU Countries, beach seining often occurs on nearshore *Posidonia* beds and it is known to devastate the sea floor (Stergiou *et al.*, 1996; Katsanevakis *et al.*, 2010).

Based on the different impacts of various fishing gear highlighted in this study, relevant and feasible management measures that should be implemented by the Greek Government within the Natura 2000 area of the Inner Ionian Archipelago (Habitats Directive site GR2220003) include: (1) the enforcement of national legislation and of Council Regulation 1967/2006, and appropriate penalties for illegal fishing; (2) strict and immediate restrictions on purse seining and trawling to ensure that these fisheries are fully sustainable and do not harm the ecosystem and its biodiversity, including endangered dolphin populations; (3) prompt implementation of the ban of beach seining by 31 May 2010, as demanded by Council Regulation 1967/2006; and (4) temporal restrictions and/or the adoption of a larger mesh size (e.g. 36 mm) for all bottom-set nets of coastal fishermen in order to increase selectivity (current practice is 20–22 mm knot-to-knot minimum). In addition, measures should be taken to ensure that the present fishing capacity does not increase. Complaints by a minority of fishermen who may be negatively affected by management action must be weighed against increased ecosystem services and advantages for the wider society. Alternative opportunities (e.g. coastal fishing tourism) and/or compensation may be proposed to fishermen who are forced to leave their jobs, for example in the framework of the European Fisheries Fund (Regulation 1198/2006). The beach seining ban is unlikely to cause social or economic problems, considering that the beach seining season is already limited to 6 months and for the remainder of the year these boats fish with trammel nets. Thus, enforcement of the ban would result in the same boats ( $n = 24$ ) fishing with trammel nets year-round.

## CONCLUSION

The information produced by this study would allow identification of appropriate fisheries management action to ensure sustainability, protection of marine biodiversity and continued ecosystem services, as advocated by the EC Habitats Directive and, more recently, by the Marine Strategy Framework Directive. The latter Directive, in particular, recognizes that ‘the marine environment is a precious heritage that must be protected, preserved and, where practicable, restored with the ultimate aim of maintaining biodiversity and providing diverse and dynamic oceans and seas which are clean, healthy and productive’. Based on the existing political and legal commitments to preserve marine biodiversity in general, and cetaceans in particular, relevant local, national and regional governmental bodies must take action to reduce fishing pressure and limit the use of fishing gears that can have unintended harmful effects on the marine environment. Solutions to reduce current over-exploitation and protect endangered marine fauna are already provided for by national and EU legislation, and

fisheries management measures within Natura 2000 sites are mandated by the EU Common Fisheries Policy. Member States can take measures to minimize the effects of fishing on conservation of the marine ecosystem within 12 nautical miles from their coasts (6 nautical miles in the case of Greece). Council Regulation 1967/2006, which includes management measures for sustainable exploitation of fishery resources in the Mediterranean Sea, and calls to create Fishing Protected Areas, should be used as framework.

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