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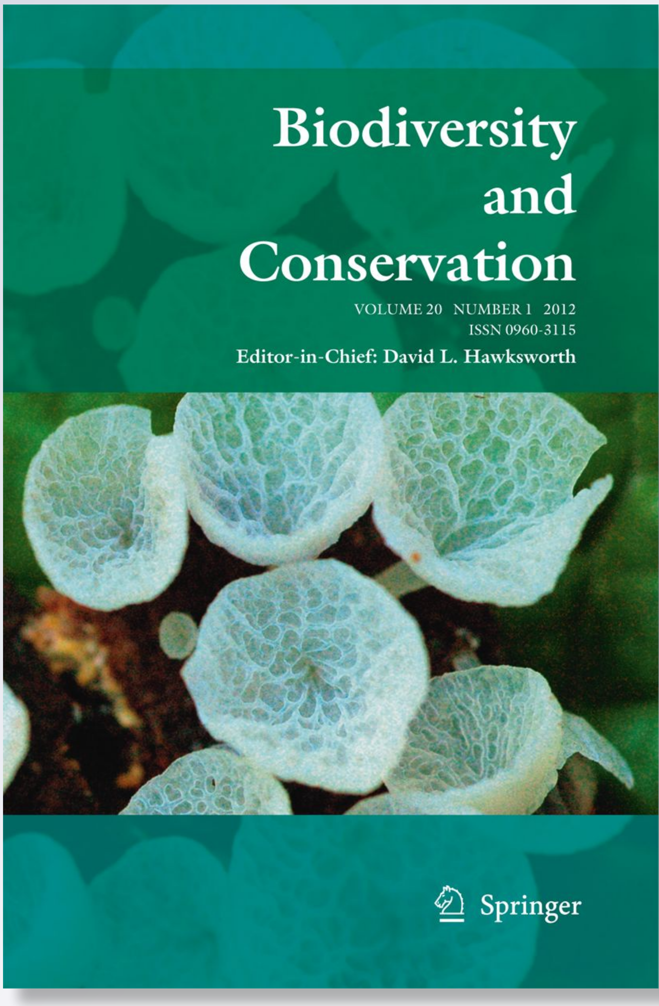

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ORIGINAL PAPER

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Abstract Fisheries are recognised as a major threat to sea turtles worldwide. Oceanic driftnets are considered the main cause of the steep decline in Pacific Ocean populations of the leatherback sea turtle *Dermochelys coriacea*. The world's largest leatherback population nests in West Africa and migrates across the Atlantic Ocean to feed off the South American coast. There, the turtles encounter a range of fisheries, including the Brazilian driftnet fishery targeting hammerhead sharks. From 2002 to 2008, 351 sea turtles were

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incidentally caught in 41 fishing trips and 371 sets. Leatherbacks accounted for 77.3% of the take ($n = 252$ turtles, capture rate = 0.1405 turtles/km of net), followed by loggerheads *Caretta caretta* (47 individuals, capture rate = 0.0262 turtles/km of net), green turtles *Chelonia mydas* (27 individuals, capture rate = 0.0151 turtles/km of net) and unidentified hard-shelled turtles (25 individual, capture rate = 0.0139 turtles/km of net) that fell off the net during hauling. Immediate mortality (i.e., turtles that were dead upon reaching the vessel, excluding post-release mortality) was similar among the species and accounted for 22.2 to 29.4% of turtles hauled onboard. The annual catch by this fishery ranged from 1,212 to 6,160 leatherback turtles, as estimated based on bootstrap procedures under different fishing effort scenarios in the 1990s. The present inertia in law and enforcement regarding gillnet regulations in Brazil could result in the reestablishment of the driftnet fishery, driving rates of leatherback mortality to levels similar to those observed in previous decades. This development could potentially lead to the collapse of the South Atlantic leatherback population, mirroring the decline of the species in the Pacific. In light of these potential impacts and similar threats to other pelagic mega fauna, we recommend banning this type of fishery in the region.

Keywords *Dermochelys coriacea* · *Caretta caretta* · *Chelonia mydas* · Incidental capture · Sea turtle conservation · Gillnet fisheries · Bycatch

Introduction

Sea turtles come into contact with a range of anthropogenic threats throughout their long lives (National Research Council 1990; Lutz and Musick 2003; Gilman et al. 2010), placing them among the most conservation-dependent marine taxa (Hamann et al. 2010). Threats at sea include ingestion of plastic debris (Bugoni et al. 2001; Derraik 2002; Guebert-Bartholo et al. 2011), organic and inorganic pollutants (Guirlet et al. 2008; van der Merwe et al. 2010; Lazar et al. 2011), entanglement in lost nets and floating debris (Northridge 1991), boat collision (Campbell 2002; Spotila 2004) and incidental catch in fisheries (Wallace et al. 2010; Alfaro-Shigueto et al. 2011). Incidental capture in fisheries is regarded as the main driver of decline in many sea turtle populations (National Research Council 1990; Wallace et al. 2010). Fisheries such as bottom-trawling for shrimp, purse seine, bottom and drift gillnets and bottom and pelagic longlines have the highest capture rates (Oravetz 1999; Lewison et al. 2004; Domingo et al. 2006a; Wallace et al. 2010). Bycatch and mitigation in fisheries are top-priority issues for sea turtle conservation, but effective mitigation strategies will be based on robust data about threats (Hamann et al. 2010).

Drift gillnets are highly efficient as fishing gear (Kotas 2004), but their low selectivity causes capture of non-target species, including cetaceans, sea turtles, seabirds and unwanted fishes, and gives these nets the name "wall of death" (Northridge 1991; Hall et al. 2000). In 1989, the United Nations urged that attention be paid to the potential ecological effects of this fishery and especially to the incidental catch of mammals, seabirds and salmon (Huppert and Mittleman 1993). In 1991, the UN General Assembly approved Resolution 46/215, which urged parties to pronounce a moratorium on the high-seas driftnet fishery (Burke et al. 1994; Bache and Evans 1999). In the same year, the European Union (EU) banned driftnets over 2,500 m long in the Mediterranean Sea. In 2002, the EU banned driftnets entirely (Tudela et al. 2005); however, the driftnet fishery continues off the coasts of Albany, Algeria, Spain, France, Greece, Italy, Malta, Monaco,

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Morocco and Turkey (Tudela et al. 2005; Cambiè et al. 2010; Lucchetti and Sala 2010). The impact of gillnets, particularly driftnets, on marine species such as seabirds and sea turtles is poorly documented (Gilman et al. 2010).

In the south-western (SW) Atlantic Ocean, driftnet fishery occurs only in Brazil (Domingo et al. 2006b), where it started in 1986. This fishery targets mainly hammerhead sharks (*Sphyrna lewini* and *S. zygaena*) of different age classes (Zerbini and Kotas 1998) for the commercial sale of fins and meat to Asian and domestic markets, respectively (Kotas et al. 2008). In 1998, Federal Ordinance IBAMA N° 121 limited the use and transport of bottom and drift gill nets over 2.5 km long and banned "finning", the practice of removing the sharks' fins, then releasing their bodies at sea; however, vessels from the ports of Itajaí, Navegantes and Porto Belo, in Santa Catarina state, south Brazil, deployed nets up to 7,846 m long between 2005 and 2006 (Kotas et al. 2008), demonstrating the challenge of enforcing such legislation. In 2007, an initiative was brought to discuss and reorganise gill net fisheries in Brazil. In 2010, this initiative resulted in the suspension of Ordinance IBAMA N° 121, thus permitting unrestricted fishing with driftnets.

Five sea turtle species nest in Brazil (Marcovaldi and Marcovaldi 1999). All are globally threatened by extinction (IUCN 2008) and included in the Brazilian list of threatened species as "Vulnerable" (loggerhead *Caretta caretta* and green turtle *Chelonia mydas*), "Endangered" (olive ridley *Lepidochelys olivacea* and hawksbill *Eretmochelys imbricata*), or "Critically Endangered" (leatherback *Dermochelys coriacea*) (MMA 2003). Capture of all five species has been recorded by different fisheries in Brazil throughout the year; including pelagic longlines (Kotas et al. 2004; Domingo et al. 2006a; López-Mendilaharsu et al. 2007; Sales et al. 2010), corrals (Bahia and Bondioli 2010), shrimp trawling (Silva et al. 2010) and coastal gillnets (Marcovaldi et al. 2001; Lima et al. 2010). Although high-seas driftnet fisheries are also reported to entangle sea turtles in the SW Atlantic Ocean, captures were not quantified, and thus, the real impact of these fisheries on sea turtle populations is unknown (Domingo et al. 2006b).

The incidental mortality of leatherback turtles in gill nets and on longline hooks has caused the steep decline of several populations around the world (Chan and Liew 1996; Sarti et al. 1996; Eckert and Sarti 1997; Spotila et al. 1996). In the Pacific Ocean, steep population declines in the 1980s and 1990s (22% per annum in the largest Mexican population—Sarti et al. 1996) may have been caused by mortality in the distant Chilean driftnet fishery in the southern Hemisphere (Frazier and Montero 1990; Sarti et al. 1996; Eckert and Sarti 1997). Spotila et al. (2000) reported a 95% decline in Pacific leatherback populations from 1975 to 2000 with an estimated 1,500 females killed in driftnet and longline fisheries during the 1990s. Frazier and Montero (1990) estimated that approximately 250 leatherbacks were caught annually in driftnets off Central Chile. Eastern Pacific driftnet fisheries are apparently still a threat, as 101 leatherbacks were killed between 2000 and 2003 in Peru (Alfaro-Shigueto et al. 2007), and it is estimated that approximately 70 leatherbacks are captured annually in driftnet fisheries operating out of only three Peruvian ports (Alfaro-Shigueto et al. 2011).

In the Atlantic Ocean, the largest leatherback rookeries are found off Gabon and Congo in Africa (Fretey 2001; Billes et al. 2005; Billes et al. 2006; Fossette et al. 2008; Witt et al. 2009), with adults moving to South Africa, east Equatorial waters and the SW Atlantic Ocean after nesting (Witt et al. 2011), and off of French Guiana and Suriname in South America (Fretey and Girondot 1996), with adults moving towards the North Atlantic after nesting. The number of females nesting every year in the single Brazilian rookery in the southeastern state of Espírito Santo is estimated to be between 1 and 18 (Thomé et al. 2007). The SW Atlantic Ocean is a critical foraging area for Gabonese populations (Witt

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et al. 2011). Billes et al. (2006) reported the capture of four tagged leatherbacks from Africa, including one on a Brazilian pelagic longline and one in a coastal gillnet. Recent satellite tracking studies confirmed the connection between the nesting grounds of leatherback turtles in the Gulf of Guinea, Africa, and foraging grounds in the SW Atlantic Ocean (Fossette et al. 2010; Witt et al. 2011).

In both the Atlantic and Pacific Oceans, a reduction in the incidental capture of leatherback turtles in southern fisheries is crucial if we are to avoid the extinction of this ancient reptile (Spotila et al. 2000; Kaplan 2005). Data on bycatch of sea turtles in gillnets are scarce globally (Gilman et al. 2010; Lucchetti and Sala 2010). In the SW Atlantic Ocean, particular attention should be paid to the capture of sea turtles, especially leatherback sea turtles, in Brazilian driftnet fisheries (Domingo et al. 2006b).

This study provides the first estimates of sea turtle bycatch in the driftnet fishery off the southeastern and southern Brazilian coasts under different fishing effort scenarios. Additionally, the spatial and temporal distribution of both fishing effort and sea turtle bycatch, as well as the operational characteristics influencing captures, are described.

Methods

Fishery description

The main target species of the commercial driftnet fishery fleet are the hammerhead sharks *S. lewini* and *S. zygaena*. The fleet operates in southeastern and southern Brazil from ports in Ubatuba (São Paulo state), Navegantes, Itajaí, Porto Belo, Laguna and Passo de Torres (Santa Catarina state) and Torres and Rio Grande (Rio Grande do Sul state) (Kotas et al. 2007). In this study, we sampled vessels from Ubatuba and Itajaí, the ports where most of the vessels using driftnets originate (Fig. 1). These vessels are wooden, with nets between 2,000 and 7,408 m long, made of twisted multifilament nylon or monofilament nylon, and with stretched mesh sizes ranging from 12 to 40 cm.

Most vessels deploy driftnets only in austral spring (September to December) and summer (December to March), when captures of sharks are high, and switch gear and target species in other seasons; however, the number of vessels in this fishery is highly variable, depending on the yield from the first months of the season (Zerbini and Kotas 1998; Kotas et al. 2007).

Study area

Driftnet fishing was recorded over the continental shelf off southern and southeastern Brazil (Fig. 1). The currents in this area are influenced by the Cabo Frio (Rio de Janeiro state) and Cabo de Santa Marta (Santa Catarina state) upwellings, where cold, nutrient-rich waters rise to the surface, and by the Subtropical Convergence (at approximately 35°S), where the warm, high-salinity, nutrient-poor waters of the Brazil Current meet the cold, low-salinity, nutrient-rich waters of the Malvinas/Falkland Current (Olson et al. 1988; Campos et al. 2000). The mixed waters of these currents flow eastward (Stramma and England 1999), playing a key role in the physical and biotic processes in the region (Campos et al. 2000). These processes sustain important fishing stocks and a range of top predators (Seeliger et al. 1998).

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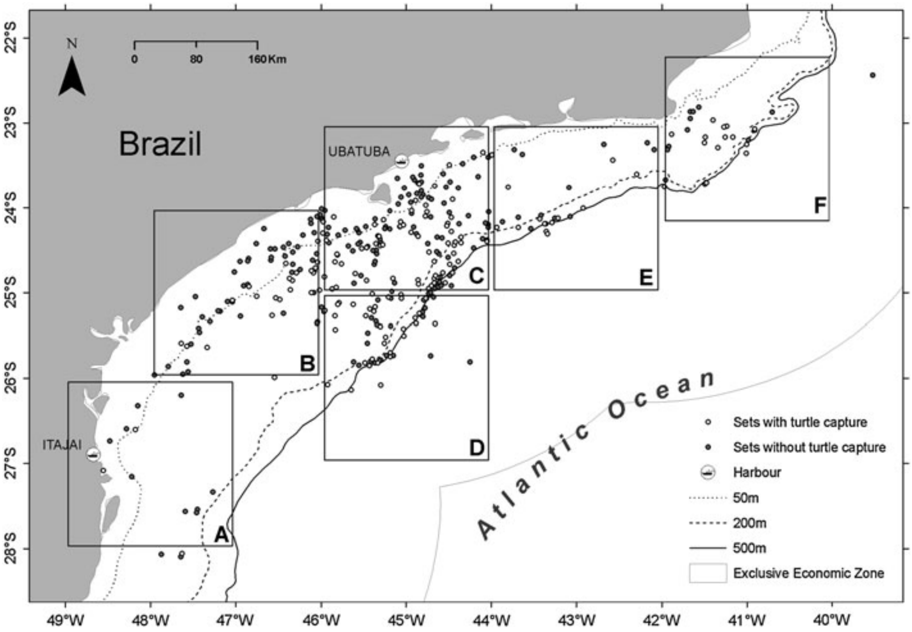

Fig. 1 Distribution of driftnet sets carried in 2002, 2003, 2005, 2006 and 2008 by the Brazilian fleet operating out of Itajaí and Ubatuba

Data gathering

Data were recorded by the captains of eight fishing vessels (seven from Ubatuba and one from Itajaí) from 2002 to 2006 and in 2008. Captains volunteered for data collection and were trained in sea turtle identification and determination of turtle condition (dead, alive or unknown). Photographs were used to confirm species identification. "Unknown" conditions usually describe turtles that fell off of the net before being hauled onboard. Date, time and coordinates of setting and hauling, number, identification and condition of entangled sea turtles were recorded in standardised datasheets. The number of vessels in the study at a given time varied because of (i) the point at which cooperation was established between fishermen and TAMAR staff in Ubatuba (2001) and Itajaí (2005); (ii) the large number of vessels from different fisheries and varying landing points in Itajaí; (iii) frequent changes in fishing-boat captains; (iv) the illegal nature of the fishery in terms of net length and catch of protected species.

Set depths were obtained from the General bathymetric chart of the ocean (GEBCO) at http://www.bodc.ac.uk/data/onlinedelivery/gebco/ using ARCGIS software v. 9.2.

Data analysis

Capture rates of sea turtles in gill net fisheries are usually reported as the number of individuals caught per linear km of net, not taking net height into account. As net height varies considerably (from 6 to 15.75 m in the current study), taller nets could potentially account for higher capture rates (Gilman et al. 2010). For comparison, capture rates were also calculated as turtles caught per km^2 of net (Table 1). This comparison was evaluated

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Table 1 Captures of leatherbacks *Dermochelys coriacea*, loggerheads *Caretta caretta*, green turtles *Chelonia mydas* and unidentified hard-shelled sea turtles and comparison between capture rates per km (number of turtles/km linear) and km^2 of net (number of turtles/km^2 of net) in the SW Atlantic Ocean, in 2002, 2003, 2005, 2006 and 2008

		Leatherback turtle	Capture rate	Loggerhead turtle	Capture rate	Green turtle	Capture rate	Unidentified hard-shelled turtle	Capture rate
Net length (km)	1793.9	252	0.1405	47	0.0262	27	0.0151	25	0.0139
Net area (km^2)	24397.8		0.0103		0.0019		0.0011		0.0010

$N = 371$ sets

by calculating the Spearman correlation between capture rates calculated by linear km of net and net height, using the mean capture rates per vessel and per set (Fowler et al. 1998). As no correlation was found between capture rates (turtles/km) and net height, all analyses were performed using the number of turtles entangled/linear km of net.

The correlation between water depth and capture rate for each turtle species was tested using Pearson's correlation coefficient. As residuals were not normally distributed and homoscedastic, differences in the mean capture rate among the six $2° \times 2°$ areas (Fig. 1), seasons, years and vessels were assessed through the non-parametric Kruskal–Wallis test. Dunn's post hoc test was used when a difference was found. The Mann–Whitney test with Bonferroni correction for multiple comparisons (Zar 2010) was used to test differences in the mean capture rates between pairs of species. Differences among sea turtle species in the proportion of turtles that were dead were evaluated through a χ^2 test. Differences were regarded as significant if $P < 0.05$. Tests were performed using the software Minitab v. 15 (Minitab Inc., Philadelphia, USA) and BioEstat v. 5.0 (Ayres et al. 2007).

The total number of sea turtles caught during a hypothetical year was estimated for the three most commonly entangled species, taking three scenarios of fishing effort into account: a low-effort scenario with 40 vessels, which is similar to the average number of vessels recorded during the study period; a medium-effort scenario with 55 vessels, which is likely a more realistic representation of the period; and a high-effort scenario with 110 vessels, which is similar to the numbers recorded in the mid-1990s (Zerbini and Kotas 1998). Non-parametric bootstrap sampling of *b* boats taken randomly with replacement was performed 10,000 times. The confidence interval for turtle captures was estimated using the 2.5th and 97.5th percentiles of the bootstrap replicates (Manly 1997) in Matlab. Data are presented as the mean number of turtles caught per year ± one standard deviation and as the lower and upper percentiles for all three species. Estimation was performed on a yearly basis because leatherbacks are stranded year-round along the southern and south-eastern Brazilian coast (Barata et al. 2004).

Results

Between 2002 and 2008, the monitored driftnet fishery operated in a vast area (from 22° to 28°S and 35° to 48°W) in isobaths ranging from 18 to 2304 metres in depth (mean = 189.5, sd = 286.2 m, $N = 364$ sets). Most sets (76.4%) occurred over the

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continental shelf, within the Brazilian economic exclusive zone (200 nautical miles), but they occasionally occurred in international waters (Fig. 1).

A total of 351 sea turtles were incidentally caught in 41 fishing trips and 371 sets (Fig. 1). Leatherbacks were responsible for the bulk of bycatches (252 individuals, 77.3% of sea turtles caught, capture rate = 0.1405 turtles/km), followed by loggerheads (47 individuals, 14.4%, capture rate = 0.0262 turtles/km), and green turtles (27 individuals, 8.3%, capture rate = 0.0151 turtles/km). Twenty-five hard-shelled turtles could not be identified to the species level (Table 1).

Most fishing sets resulted in no captures of leatherback (70.6%), loggerhead (91.4%) or green turtles (94.6%). Among the 109 sets with reported leatherback entanglements, 45.9% had one and 23.9% had two turtles per set. Two unusual sets caught ten turtles (at 24°46′3″S; 45°35′55″W and 25°08′18″S; 45°48′16″W). A third set caught 18 leatherbacks (at 24°12′670″S 45°56′15″W) (Fig. 2). Among the 32 sets with captures of loggerheads, 25 caught a single turtle, and a maximum of five turtles were caught in a single set (at 24°07′19″S; 43°12′62″W). Only 20 sets caught green turtles, usually a single specimen (15 sets), with a maximum of three individuals in a single set.

Capture rates (turtles/km) calculated per vessel were not correlated with net height for any of the three commonly caught turtle species or for all turtle species pooled, including unidentified sea turtles (Spearman rank correlation, leatherback $R = 0.37$, $P = 0.36$; loggerhead $R = 0.27$, $P = 0.51$; green $R = -0.14$, $P = 0.73$; all species $R = 0.27$, $P = 0.52$, all tests d$f = 6$) and calculate per set (Pearson correlation coefficient, leatherback $R = 0.04$, $P = 0.48$; loggerhead $R = 0.03$, $P = 0.57$; green $R = 0.008$, $P = 0.89$; all species $R = -0.002$, $P = 0.97$, all tests d$f = 344$ sets).

The proportion of turtles hauled onboard dead was similar across species ($\chi^2 = 1.03$, $P = 0.6$, d$f = 2$), with 29.4, 23.4 and 22.2% for leatherback, loggerhead and green turtles, respectively (Table 2).

Capture rates were not correlated with water depth for any of the three sea turtle species (R values from −0.04 to 0.009, P values from 0.47 to 0.86, d$f = 365$ for all species). The capture rate of leatherbacks in Area D was significantly higher than in Area B (Kruskal–Wallis test $H = 19.2$, $P = 0.002$, d$f = 5$; Dunn's test $Z = 3.71$, $P < 0.05$, Fig. 1). For

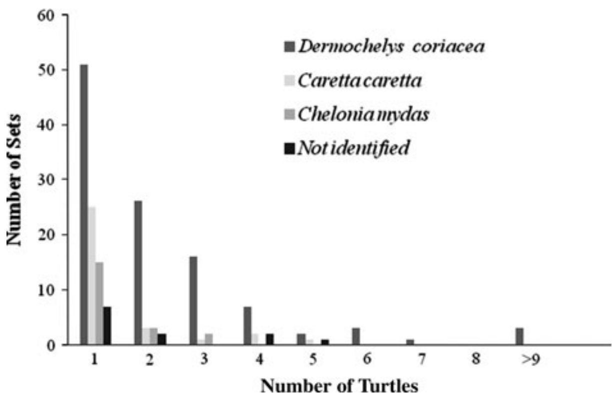

Fig. 2 Numbers of leatherbacks (*Dermochelys coriacea*), loggerheads (*Caretta caretta*), green turtles (*Chelonia mydas*) and unidentified sea turtles caught per set in 2002, 2003, 2005, 2006 and 2008, excluding sets with no captures. $N = 371$ sets

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Table 2 Number of leatherbacks (*Dermochelys coriacea*), loggerheads (*Caretta caretta*), green turtles (*Chelonia mydas*) and unidentified hard-shelled sea turtles hauled dead or alive by the Brazilian driftnet fishery fleet operating in the SW Atlantic Ocean

Condition	Leatherback turtle (%)	Loggerhead turtle (%)	Green turtle (%)	Unidentified hard-shelled turtle (%)	Total (%)
Alive	175 (69.4)	35 (74.5)	20 (74.1)	24 (96)	254 (72.4)
Dead	74 (29.4)	11 (23.4)	6 (22.2)	0 (0.0)	91 (25.9)
Unknow	3 (1.2)	1 (2.1)	1 (3.7)	1 (4.0)	6 (1.7)
Total	252	47	27	25	351

The descriptor "Unknown condition" refers to animals that fell out of the net before being hauled in

Table 3 Number of leatherbacks (*Dermochelys coriacea*), loggerheads (*Caretta caretta*), green turtles (*Chelonia mydas*) and unidentified hard-shelled sea turtles caught per year (with capture rates) by the Brazilian driftnet fishery fleet operating in SW Atlantic Ocean

Year	Leatherback turtle	Capture rate	Loggerhead turtle	Capture rate	Green turtle	Capture rate	Unidentified hard-shelled turtle	Capture rate	Total
2002	62	0.1096	16	0.0283	9	0.0159	12	0.0212	99
2003	121	0.2325	21	0.0404	5	0.0096	6	0.0115	153
2005	56	0.1170	10	0.0209	7	0.0146	6	0.0125	79
2006	4	0.0312	0	0	3	0.0234	0	0	7
2008	9	0.0892	0	0	3	0.0297	1	0.0099	13
Total	252	0.1405	47	0.0262	27	0.0151	25	0.1957	351

loggerhead and green turtles, however, capture rates did not differ across areas (Kruskal–Wallis $H = 1.26$, $P = 0.94$ d$f = 5$; $H = 0.79$, $P = 0.98$, d$f = 5$, respectively).

Leatherback and green turtles were caught in all years, but loggerheads were not caught in 2006 and 2008 (Table 3). The capture rate did not vary across years for leatherbacks (Kruskal–Wallis $H = 8.83$, $P = 0.07$, d$f = 4$; capture rate min. = 0.0312 turtles/km in 2006 and max. = 0.2325 turtle/km in 2003), loggerheads (Kruskal–Wallis $H = 0.06$, $P = 0.97$, d$f = 2$; min. = 0 turtles/km in 2006 and 2008, excluded from the statistical analysis, max. = 0.0404 turtles/km in 2003) or green turtles (Kruskal–Wallis $H = 0.43$, $P = 0.98$, d$f = 4$, min. = 0.0096 turtles/km in 2003, max. = 0.0297 turtles/km in 2008).

Sea turtles were incidentally caught throughout the year, reaching a minimum during the winter months (June to September) (9.2% of individual turtles identified to the species level) and a maximum during the austral spring (40.1% of individuals identified to the species level) (Table 4). Capture rates did not vary across seasons for any of the three species (Kruskal–Wallis, $H = 3.31$, $P = 0.34$ for leatherbacks; $H = 1.79$, $P = 0.67$ for loggerheads; $H = 0.47$, $P = 0.93$ for green turtles, all d$f = 3$). Overall, capture rates were high during the austral spring (September to December), summer (December to March) and autumn (March to June) for leatherbacks, the summer and autumn for loggerheads, and the spring and summer for green turtles (Table 4).

Captures rates of leatherback turtles varied significantly by vessel, from zero in two vessels to 0.0422 turtles/km in one vessel (Kruskal–Wallis $H = 14.72$, $P = 0.01$, d$f = 5$), but no such pattern was observed for loggerhead and green turtles (Kruskal–Wallis $H = 4.27$, $P = 0.37$, d$f = 4$; $H = 2.26$, $P = 0.52$, d$f = 3$, respectively).

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Table 4 Number of leatherbacks (*Dermochelys coriacea*), loggerheads (*Caretta caretta*), green turtles (*Chelonia mydas*) and unidentified hard-shelled sea turtles caught per season (along with capture rates) by the Brazilian driftnet fishery fleet operating in the SW Atlantic Ocean in 2002, 2003, 2005, 2006 and 2008

Season	Leatherback turtle	Capture rate	Loggerhead turtle	Capture rate	Green turtle	Capture rate	Unidentified hard-shelled turtle	Capture rate	Total
Summer	82	0.1401	25	0.0427	9	0.0154	5	0.0085	121
Autumn	43	0.1585	13	0.0479	1	0.0037	7	0.0258	64
Winter	17	0.0950	2	0.0112	1	0.0056	0	0	20
Spring	110	0.1451	7	0.0092	16	0.0211	13	0.0171	146
Total	252	0.1405	47	0.0262	27	0.0151	25	0.1957	351

Seasons: summer (Dec to Mar), autumn (Mar to Jun), winter (Jun to Sep) and spring (Sep to Dec)

Table 5 Estimated annual bycatch of leatherbacks (*Dermochelys coriacea*), loggerheads (*Caretta caretta*) and green turtles (*Chelonia mydas*) by the Brazilian driftnet fishery fleet operating in the SW Atlantic Ocean under different fishing scenarios

	Scenario 1 40 vessels	Scenario 2 55 vessels	Scenario 3 110 vessels
Leatherback turtle	1212.1 ± 497.3 (360.0–2280.0)	1669.0 ± 684.2 (515.6–3148.8)	3299.0 ± 1329.8 (990.0–6160.0)
Loggerhead turtle	233.9 ± 95.5 (80.0–440.0)	323.8 ± 131.2 (110.0–611.9)	645.8 ± 263.0 (220.0–1210.0)
Green turtle	134.8 ± 72.7 (20.0–295.0)	183.2 ± 99.0 (27.5–398.7)	371.5 ± 198.1 (55.0–811.2)

Values are the number of turtles ± one standard deviation (2.5th and 97.5th percentiles of the confidence interval)

Results from the bootstrap analysis suggest that a minimum of 1,212 leatherbacks, 233 loggerheads and 134 green turtles were caught per year during the study period (Scenario 1 in Table 5). Taking into account the proportion of dead turtles hauled onboard, it is likely that a minimum of 356 leatherbacks, 55 loggerheads and 30 green turtles were killed per year. Under 1990s conditions (Scenario 3), a mean of 3,299 leatherbacks may have been captured each year (97.5% CI = 990–6,160); of which, 291 to 1,811 were dead. These estimates do not consider post-release mortality and turtles that fell off the net before hauling.

Discussion

Many leatherback turtles have been incidentally caught in Brazilian coastal (e.g., coastal gillnets and shrimp trawls—Marcovaldi et al. 2006) and offshore (e.g., trawls—Domingo et al. 2006b; and pelagic longlines—Sales et al. 2008, 2010) fisheries, and an unknown number are caught and killed annually in the Brazilian driftnet fishery. This bycatch is likely to have a significant impact, even on the West African population, which is the largest in the world with 15,730 to 41,373 nesting females (Witt et al. 2009). Bycatch may drive the tiny population nesting in Brazil to extinction. The numbers of catches in driftnet fisheries reported here could not have been drawn exclusively from the small Brazilian population (Thomé et al. 2007), indicating that driftnets and other fisheries are probably

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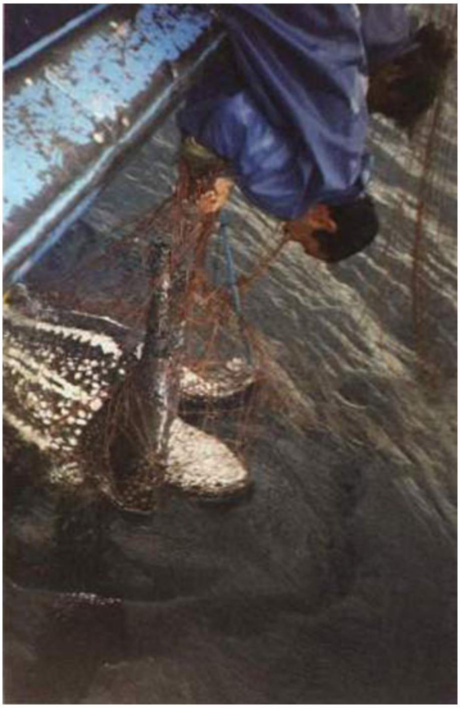
capturing leatherback turtles from the African nesting populations as well; these latter populations are vital to the conservation of these species in the Atlantic. Targeted molecular analyses are nonetheless needed to confirm this assertion. In contrast, this hypothesis is consistent with the recent observation that one out of three migration groups of Gabonese nesting leatherback turtles regularly crosses the Atlantic Ocean, swimming to the Subtropical waters off Brazil, Uruguay and Argentina (Witt et al. 2011). Additionally, female leatherback turtles have been observed in the SW Atlantic Ocean after having been tagged at Gabonese nesting beaches (Billes et al. 2006). Despite the apparently minor importance of the Brazilian population to the conservation of leatherbacks in the Atlantic Ocean, its protection could potentially increase the genetic diversity and sex-ratio balance of the species (Dutton et al. 1999). Thomé et al. (2007) reported an upward trend in the number of nests, probably as a result of the conservation effort invested at this nesting ground. This work minimises the effects of fishery-associated mortality at later phases of the turtle life cycle. Based on pictures taken by fishermen, the leatherbacks not hauled onboard were either large immatures close to nesting size or adults (Fig. 3); they (especially the females) thus have high intrinsic reproductive value to their populations. In addition to the threat they pose to the tiny Brazilian population, driftnets are a serious threat to the largest East African leatherback rookeries, which are key populations of the species.

In most sets, no turtles were caught. This pattern holds in several different fisheries, such as pelagic longlines (Gilman et al. 2006; Pradhan and Leung 2006; Sales et al. 2010), drift gillnets (Gallaway 2001) and trammel nets (Cambiè 2011). The capture of sea turtles is rare (but see a different situation in Alfaro-Shigueto et al. 2011 for small-scale fisheries). Sets with more than six individuals captured were observed only during the summer. This

Fig. 3 Large immature leatherback turtle (*Dermochelys coriacea*) incidentally captured by Brazilian driftnet fishery



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outcome is probably related to feeding aggregation; during this period, the nutrient-rich waters of the South Atlantic Central Water (SACW) are close to the continental shelf (Castro-Filho et al. 1987), increasing productivity and encouraging the development of the gelatinous prey (Rocha et al. 2007), the dominant organisms in the macrozooplankton assemblage in southern Brazil during spring/summer (Mianzan and Guerrero 2000). Gelatinous macrozooplankton constitute the main food items of leatherbacks over continental shelves and oceanic areas (Bjorndal 1997; Spotila 2004; Gulko and Eckert 2004; Dodge et al. 2011).

Capture rates were not correlated with net height for any turtle species, suggesting that in this study, net height did not influence captures. Studies on the diving behaviour of these three species have shown that they spend most of their time in the top 100 m of the water column (leatherback 93%; loggerhead 40%; and green >80%—Hays et al. 2001; Polovina et al. 2003; James et al. 2005, 2006; López-Mendilaharsu et al. 2009). Turtles occupy the entire vertical profile of the net, suggesting that they are vulnerable to captures throughout the net profile. Thus, the finding of no correlation between net height and capture rates is surprising, especially as net height is highly variable (2.6 times in this study and between 4.5 and 27 m., i.e., six times, in Zerbini and Kotas 1998). In two ports in the Trinidad coastal surface drift gill net fishery, however, the capture of leatherbacks decreased by 11 and 75% when the net profile was reduced from 10 to 5 m height (Gearhart et al. 2009). Such differences could be attributed to changes in diving and swimming behaviours during nesting (Trinidad) or foraging and migration (Brazil), but this issue deserves further investigation.

The proportion of turtles suffering immediate mortality did not vary according to species, suggesting that all species are similarly vulnerable. Mortality rates ranged from 22 to 30% and were similar to those observed in the driftnet fishery in Trinidad, where fishermen reported leatherback mortality rates from 10 to 34% (Lum 2006); in contrast, the observed values were much lower than the 69% leatherback mortality associated with bottom-trammel nets in the Mediterranean Sea (Cambiè 2011) or the 69.4% leatherback mortality observed in Tunisian bottom gillnets (Echwikhi et al. 2010). The higher mortality associated with bottom fisheries compared with driftnets likely results from the restriction of the turtles' ability to surface and breathe. The recorded driftnet-associated mortality is likely underestimated in this study, as dead turtles may fall-off the net during hauling and because considerable post-release mortality may occur. Additionally, fishermen may deliberately under-report mortality rates. Fishermen operating with trammel gillnets in Italy observed a fall-off rate in loggerheads as high as 14.3% during hauling (Cambiè 2011). Although post-release mortality is difficult to determine, severe physiological disruptions and injuries incurred while entangled in gillnets may result in undocumented deaths (Snoddy and Williard 2010). These authors used blood parameters and satellite tracking of gill-netted green turtles and Kemp's ridley *Lepidochelys kempii* turtles and estimated the post-release mortality to be between 7.1 and 28.6%. It is worth noting that the entangled turtles in this study soaked for only 4 h, while the soak time of driftnets studied here was frequently as long as 12 h. The immediate driftnet-associated mortality recorded in Brazil is higher than in other fisheries in the area. On the pelagic longlines, for example, 1.7, 4.0 and 4.4% of leatherback, loggerhead and green turtles, respectively, are hauled dead (Sales et al. 2008). Thus, although fewer leatherback and loggerhead turtles are caught in driftnets in comparison with longlines, mortality is probably higher in the former fishery type.

The temporal and spatial distribution of effort in the driftnet fishery is closely related to the profitability of the main target species (i.e., hammerhead sharks), as demonstrated in

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previous studies (Kotas 2004; Kotas et al. 2007). Distribution is also related to the at-sea autonomy of vessels (Zerbini and Kotas 1998). For the Ubatuba-based fleet, there was a clear offshore expansion. The mean set depth in the 1990s was 91.1 m (Kotas et al. 2005) versus 187 m in the 2000s (this study), suggesting that fishing trip lengths, and consequently costs, expanded in the search for target species (Kotas et al. 2005). Furthermore, this driftnet fishery fleet follows the hammerhead sharks from the slope where copulation occurs to their coastal spanning grounds in summer and back to offshore waters (Klippel et al. 2005; Kotas et al. 2008). While the wide longitudinal distribution of sets follows the migration of hammerhead sharks, the higher capture rates of leatherback turtles in Area D (offshore) compared with the adjacent Area B (neritic) is probably related to the concentration of leatherbacks on the shelf break area (López-Mendilaharsu et al. 2009).

The absence of seasonal variation in capture rates for the three sea turtle species suggests that turtles are vulnerable to nets throughout the year, which makes an annual estimation of captures under different fishing effort scenarios important. We carried out estimation on a hypothetical annual fishing season, but the bulk of the driftnet fishing effort occurs from September to March and minor effort during other months. This observation is also consistent with records of leatherbacks stranded along the Brazilian coast throughout the year (Barata et al. 2004). Additionally, satellite tracking studies demonstrate that leatherbacks fitted with satellite transmitters at sea after being captured in a pelagic longline, coastal gill net or driftnet, as well as females nesting on Brazilian coast, commute between foraging grounds in the SW Atlantic Ocean for long periods, migrating over the continental shelf and shelf break (López-Mendilaharsu et al. 2009; Almeida et al. 2011). In contrast, the driftnet fishing effort is higher during the austral spring and summer (September to March) as a result of the higher fishing yield (Kotas 2004; Kotas et al. 2007; Kotas et al. 2008), and thus, more turtles are caught during these seasons. Moreover, captures on pelagic longline gear increase in the SW Atlantic during the summer and autumn (Pons et al. 2010 and references therein), while longline fishing effort increases in the winter over the shelf break (see Bugoni et al. 2008 for year-round fishing effort), where capture rates of leatherback and loggerhead turtles are high.

Similar to the vessels used for driftnet fishing off the coast of Italy (Cambiè et al. 2010), Brazilian driftnet vessels are typically multi-gear, operating in three distinct fisheries over the year: surface driftnet for hammerhead sharks, mid-water gillnet for sand sharks *Carcharias taurus*, and bottom gillnet for white croaker *Micropogonias furnieri* (Zerbini and Kotas 1998; Kotas et al. 2005), highlighting the seasonal nature of the surface driftnet fishery. These characteristics complicate the assessment of the impact of driftnet fishery on non-target species (Kotas et al. 2005; Vooren et al. 2005; Marigo and Giffoni 2010; this study). Even the target hammerhead sharks may be affected by fishing pressure (Vooren et al. 2005).

There was a steep decline in the number of vessels using driftnets in the SW Atlantic Ocean in recent years, with no vessels operating in 2009 (UNIVALI/CTTMar 2010). The reduction of the fleet probably resulted from the reduction in the catch of target species and consequently in profitability. Additionally, the IBAMA Normative Instruction N°166/2007 halted the issue of new driftnet fishing licenses and established a two-year deadline by which vessels were to replace driftnets with other gear. Despite fleet reduction and fishery collapse, the current lack of regulation of gillnet fisheries could lead to a regrowth of the driftnet fishery, with potentially severe impacts on sea turtles, as demonstrated by the estimated catches under different scenarios. Recently, the end of the above IBAMA Normative allowed three vessels from Itajaí and Porto Belo to return to driftnet fishing (L. Maçaneiro, Projeto TAMAR, personal communication).

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Regardless of the scenario used to estimate the number of turtles caught in the driftnet fishery, the overall capture of leatherback and loggerhead turtles is high. For each of the three scenarios, the mean estimates were 1,212, 1,669 and 3,299 leatherbacks and 234, 324 and 646 loggerheads. For leatherbacks in particular, mortality known to be caused by driftnets is comparable to the estimated number of turtles killed in the Pacific Ocean ($\sim$2,000 per year, Eckert and Sarti 1997), which led to the collapse of several populations (Sarti et al. 1996; Spotila et al. 1996), or to the 3,000 adult leatherbacks caught yearly in Trinidad (Lum 2006). If the current inertia in lawmaking and enforcement of gillnet regulations in Brazil results in the regrowth of the driftnet fleet, Atlantic leatherback populations may incur losses comparable to those that occurred in the Pacific Ocean.

Some mitigation measures to address the incidental capture of sea turtles in gillnets have been tested recently. Wang et al. (2010) found that green sea turtle bycatch in bottom gillnets could be reduced by 60 and 40%, with similar fish target catches, using chemical light sticks and LED lights, respectively. Similarly, Gearhart et al. (2009) found that lowering the net profile from 10 to 5 m. leads to a significant reduction in the catch of leatherbacks in the coastal driftnet off Trinidad; however, the efficiency of visual deterrents, gear modifications or other mitigation measures in the high-seas driftnet fisheries targeting sharks in Brazil (Kotas et al. 2008), tuna in the North Atlantic Ocean (Rogan and Mackey 2007) or swordfish and tunas in the Mediterranean (Tudela et al. 2005; Cambiè et al. 2010) has yet to be tested. It is improbable that any single mitigation measure will significantly reduce catches of all sea turtle species, maintain catches of these threatened reptiles at acceptable levels, and concomitantly prevent entanglement of whales, dolphins and other threatened megafauna while remaining commercially profitable. High-seas driftnets have long been recognised as having unacceptable impacts. Thus, consistent with the United Nations, the European Union and a range of studies (e.g. Northridge 1991; Rogan and Mackey 2007) we also recommend its banning in the SW Atlantic Ocean.

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