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**HISTORIA DE VIDA E VULNERABILIDADE DE TUBARÕES OCEÂNICOS
(ELASMOBRANCHII) DO ATLANTICO SUL.**

**RECIFE, AGOSTO
2015**



UNIVERSIDADE FEDERAL RURAL DE PERNAMBUCO
PROGRAMA DE PÓS-GRADUAÇÃO EM RECURSOS PESQUEIROS E
AQUICULTURA

HISTORIA DE VIDA E VULNERABILIDADE DE TUBARÕES OCEÂNICOS
(ELASMOBRANCHII) DO ATLANTICO SUL.

Rodrigo Risi Pereira Barreto

Tese apresentada ao Programa de Pós-Graduação em Recursos Pesqueiros e Aquicultura da Universidade Federal Rural de Pernambuco como exigência para obtenção do título Doutor.

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RECIFE, AGOSTO

2015

Ficha catalográfica

B273h Barreto, Rodrigo Risi Pereira
História de vida e vulnerabilidade dos tubarões oceânicos
(*elasmobranchii*) do Atlântico Sul / Rodrigo Risi Pereira
Barreto. – Recife, 2015.
165 f. : il.

Orientador(a): Rosangela Lessa.
Tese (Programa de Pós Graduação em Recursos
Pesqueiros e Aquicultura) – Universidade
Federal Rural de Pernambuco, Departamento de Pesca e
Aquicultura, Recife, 2015.
Referências.

1. Tubarão (Peixe) 2. Atlântico Sul, Oceano 3. CPUE
I. Lessa, Rosangela, orientadora II. Título
CDD 639

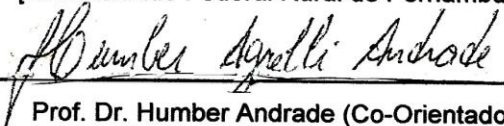
**UNIVERSIDADE FEDERAL RURAL DE PERNAMBUCO
PRÓ-REITORIA DE PESQUISA E PÓS-GRADUAÇÃO
PROGRAMA DE PÓS-GRADUAÇÃO EM RECURSOS PESQUEIROS E AQUICULTURA**

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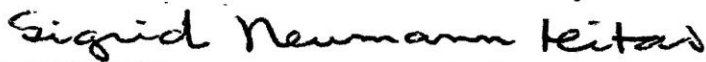
Rodrigo Risi Pereira Barreto

Tese julgada adequada para obtenção
do título de doutor em Recursos
Pesqueiros e Aquicultura. Defendida e
aprovada em 28/08/2015

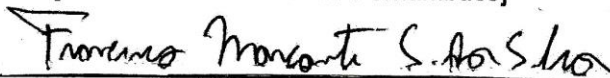
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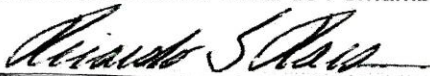
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Epigrafe

A força não provém da capacidade física.
Provém de uma vontade indomável.”

Mahatma Gandhi.

Dedicatória

A Fatima Motta Risi, Priscila Risi Pereira Barreto e Marilia Ferreira Lancellotti pelo amor incondicional. A Professora Rosangela Lessa pela paciência, exemplos e ensinamentos. Aos Professores Francisco Marcante, Humber Andrade, Boris Worm, Francesco Ferretti, Joanna Mills Flemming, Maria Lucia Araújo, Ricardo Rosa, Sigrid Neumann e Alberto Amorim. Aos meus tios (Edison, Silvana, Eliana, Waldemar e Luiz Gonzaga). Aos tantos amigos que me acompanham ou acompanharam pelo caminho da vida (felizmente não cabem todos aqui). Especialmente aos tubarões que estão no mar e meu pai (Élcio da Silva Pereira Barreto) que está no céu. Com amor.

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Resumo

Uma grande diversidade de tubarões vem sendo explorada pelas pescarias modernas, capturados regularmente como by-catch (ou captura incidental) e após a década de 90 como espécies alvo, principalmente devido ao alto valor de suas nadadeiras no mercado internacional. Este grupo de espécies é caracterizado por crescimento lento, maturação tardia, alta longevidade, baixas taxas de fecundidade e produtividade, conjunto de características que implicam em baixo potencial de reposição para a maioria das espécies, resultando em sérias implicações para a sustentabilidade das populações e também da pesca. Nações costeiras do Atlântico Sul (Brasil, Uruguai, África do Sul e Namíbia) têm desde a década de 50 cedido seus portos a frotas espinheleiras tradicionais (principalmente da Ásia e União Europeia) objetivando intercâmbio de tecnologia para desenvolver suas próprias frotas através de uma política conhecida como arrendamento. Declínios de espécies comerciais como por exemplo os atuns e os espadartes em outras localidades, assim como as divisões de cotas de pesca estabelecidas pela principal RFMO do Atlântico, a ICCAT, foram responsáveis pela introdução pelo menos 20 diferentes frotas que utilizam espinheis pelágicos. Espinheis pelágicos estão comprovadamente associados a declínios em abundância de inúmeras espécies de tubarões oceânicos, em diversas localidades do mundo. Entretanto informações sobre o Atlântico Sul são incipientes, com poucos estudos concentrados em espécies mais abundantes como, por exemplo, o tubarão azul. A presente tese reúne informações sobre captura e história de vida coletadas junto aos principais órgãos de monitoramento sediados no Brasil, com objetivo de identificar tendências populacionais para

as principais espécies capturadas no Atlantico Sul. No primeiro capítulo, abordamos a idade e crescimento do tubarão mako, em virtude de ser a única espécie (em meio as principais) cujo crescimento ainda nao tinha sido estudado na área do presente estudo. No capítulo 2, abordamos o histórico da pesca bem como as diferentes fases de exploração em nossa área de estudo, resultando na identificação de tendências na exploração e abundancia das principais espécies. O capítulo 3 aborda a vulnerabilidade deste grupo de espécies através do uso de diferentes métodos demográficos. Nossos resultados apontam que a maioria das espécies analisadas encontram-se depletadas e seriamente ameaçadas pela pesca de espinhel pelágico, que ainda nao dispõe de adequado monitoramento e controle no Atlantico Sul.

Abstract

A great diversity of sharks has been explored by modern fisheries, regularly caught as by-catch and after the decade of 1990's as target species, mainly due to the high value of their fins in the international market. This species group is characterized by slow growth, late maturity, high longevity, low rates of fertility and productivity that implies in low potential of replacement for most species, resulting in serious implications for the sustainability of populations and also for fishing. Coastal nations of the South Atlantic (Brazil, Uruguay, South Africa and Namibia) have historically (since the 50s) given its ports to traditional longline fleets (mainly from Asia and the European Union) in order to exchange technology to develop their own fleets through a policy known as leasing. Declines of commercial species such as tuna and swordfish elsewhere, moreover divisions of fishing quotas established by the ICCAT, main RFMO in Atlantic, were responsible for the introduction of at least 20 different pelagic longline fleets. Pelagic longline fishing are demonstrably associated with declines in abundance of many species of oceanic sharks in various locations around the world. However information on the South Atlantic is incipient, with few studies focusing on more abundant species such as the blue shark. This thesis gathers information about catches and life history collected from the main monitoring fishing agencies from Brazil, in order to identify population trends and conservation status for main species caught in the South Atlantic. In the first chapter, we address the age and growth of the shortfin mako shark, because growth of this specie had not yet been studied in our study area. In Chapter 2, we discuss the history of fishing as well the different phases of exploitation focused in western and central South Atlantic, resulting in the identification of trends in exploitation and thus, abundances of

key species. The third chapter addresses the vulnerability of this group of species through the use of different demographic methods. Overall our results show that most species analyzed are depleted and seriously threatened by pelagic longline fishing, which still lacks adequate monitoring and control in the South Atlantic.

1. Introdução e Revisão de literatura

Os condrictes, ou peixes cartilaginosos, têm sobrevivido por mais de 400 milhões de anos estando entre os taxa mais antigos do grupo dos vertebrados (Camhi, 2008). É considerado um grupo de grande sucesso evolutivo que apresenta todos os modos de reprodução existentes entre os vertebrados, podendo variar de planctívoros a predadores de topo, ocupando diversos nichos aquáticos (Priede et al., 2006; Snelson et al., 2008). Dentro deste grupo, estão os tubarões, que devido ao desenvolvimento do aparato mandibular e do sistema sensorial, tornaram-se predominantemente predadores, ocupando níveis tróficos superiores (Musick, 1999; Camhi et al., 1998). Como são espécies que ocupam o topo das teias tróficas, ocorrem em números relativamente menores do que os taxas inferiores (Walker, 1998; Stevens et al., 2000). Muitas espécies de tubarões desempenham um importante papel na manutenção dos ecossistemas marinhos, pois controlam taxas populacionais de presas em potencial, auxiliando no controle da qualidade genética das populações (Holden, 1974).

Uma grande diversidade de tubarões vem sendo explorada pelas pescarias modernas, capturados regularmente como by-catch, ou captura incidental, por diversas pescarias destacando os espinhéis, as redes de arrasto e as redes de emalhar, que usualmente objetivam capturar teleósteos de alta demanda comercial (Baum et al., 2003). Aliado a isto, a maioria das espécies de tubarões possui nadadeiras valiosas, comercializadas no mercado asiático como iguaria refinada, para compor as famosas sopas de nadadeiras, demanda que tem aumentado devido ao avanço da economia

asiática (Clarke et al., 2004; Dulvy et al., 2008). As nadadeiras possuem valor muito mais alto do que a carne propriamente dita e isto economicamente falando, torna-se um incentivo para que as frotas pesqueiras retenham as nadadeiras e descartem as carcaças ao mar, muitas vezes com o animal ainda vivo, uma prática criminosa conhecida como “*finning*” (Clarke et al., 2004).

Os tubarões são caracterizados por crescimento lento, maturação tardia, alta longevidade, baixas taxas de fecundidade e produtividade (Holden, 1974; Cortés, 2002). De modo geral os tubarões possuem particularidades biológicas que os aproximam em termos de vulnerabilidade muito mais aos répteis, aves e mamíferos do que aos peixes propriamente ditos (Hoenig e Gruber, 1990). Esse conjunto de características resulta em baixo potencial de reposição para a maioria das espécies, com sérias implicações para a sustentabilidade pesqueira, conferindo às populações de tubarões uma capacidade limitada de resiliência em caso de sobre-exploração (Smith et al., 1998, 2008; Stobutzki et al, 2002). Neste contexto, a subestimação dos registros das capturas de espécies do *by-catch*, associada ao finning, a perda de habitat e a degradação ambiental incluem os tubarões entre os grupos marinhos mais ameaçados da atualidade (Pierce, 2010).

Pesquisas recentes apontam que acentuados declínios populacionais tem ocorrido para diversas espécies de tubarões em todo mundo. Existem casos bem documentados de populações de tubarões que entraram em colapso como *Lamna nasus* no Atlântico Norte, *Galeorhinus galeus* na Califórnia e Austrália, *Cetorhinus maximus* na Inglaterra, *Squalus acanthias* no Mar do Norte e Columbia Britânica, *Pristis pectinata* na Florida e

Louisiana, e os grandes tubarões costeiros na costa oeste dos Estados Unidos, destacando *Carcharias taurus* e *Carcharhinus obscurus* (IUCN, 2008). Pescarias nacionais também atingiram níveis críticos para *Squatina guggenheim*; *S. occulta*; *Rhinobatos horkelii*; *Carcharias taurus*; *Galeorhinus galeus* ; *Mustelus schmitti*; *M. fasciatus*; *Carcharhinus plumbeus*, *C. porosus*, *C. galapagensis*, *Sphyrna tudes*, *S. tiburo*, *S. lewini*, *S. media*, *S. tudes*, *S. zygaena*, *Isogomphodon oxyrinchus*, *Galeorhinus galeus*, *Mustelus fasciatus* (Vooren, 1997; Lessa et al., 1999; Lessa et al. 2006ab; ICMBio 2014).

A União Internacional para Conservação da Natureza (IUCN, 2010) reporta que aproximadamente 30% da fauna de tubarões e raias estão em categorias de ameaça, ou quase ameaçadas, devendo-se considerar ainda que 47% das espécies descritas tem informação insuficiente para que se avalie o seu estado de conservação (IUCN, 2010). No Brasil o percentual de espécies em categorias de ameaça foi ainda maior (39%) sendo: 18% Criticamente ameaçadas CR; 5% ameaçadas EN; 16% vulneráveis VU; 1% regionalmente extintas; 9% próximas a ameaça NT; 18% em menor preocupação LC e 34% com informações insuficientes para avaliação, DD (ICMBio, 2014).

Esse histórico de não sustentabilidade tem levado organizações internacionais a elaborar estratégias de conservação. Em 1999 a FAO (Food and Agriculture Organization) introduziu o Plano de Ação Internacional para o Manejo e Conservação dos Tubarões (International Plan of Action for the Conservation and Management of Sharks- IPOA Sharks) com objetivo de programar ações efetivas no que diz respeito ao uso sustentável dos

tubarões capturados direta ou indiretamente por pescarias globais (FAO, 2000). O documento contextualizado no âmbito daquele plano fornece uma série de ações necessárias para efetiva conservação do grupo, de maneira clara e concisa suplementado pelo Manual de Técnicas de Gestão da Pesca de Elasmobrânquios (APEC, Musick e Bonfil, 2005). Em 2015 o Governo do Brasil também instituiu seu plano de ação que atualmente encontra-se em fase de implementação (ICMBio, 2014).

No que diz respeito às espécies oceânicas (aproximadamente 2,5% do total de 1160 espécies de condrictes conhecidas, Ebert, 2007), as informações sobre capturas e tendências, bem como abundância e história de vida das espécies são incipientes (Hazin et al. 2008); sendo as mesmas fundamentais no desenvolvimento de estratégias sustentáveis de manejo e conservação (Lessa et al. 1999). Os tubarões oceânicos são capturados no Atlântico, Índico e Pacífico tendo suas capturas reais não identificadas visto às altas taxas de descarte e também o *finning* (Clarke et al. 2015). Um estudo realizado no mercado de peixes de Hong-Kong constatou que aproximadamente 70% das nadadeiras comercializadas pertenciam a espécies oceânicas, capturadas em diversas localidades (Clarke et al., 2006a). O número estimado de tubarões comercializados neste mercado atinge a alarmante marca de 38 milhões de indivíduos por ano (estimado entre 28 e 73 milhões de indivíduos) correspondendo a 1,7 milhões de toneladas (Clarke et al., 2006b, Dulvy et al., 2008). Estas estimativas apontam que a captura de tubarões oceânicos supera em torno de 3 a 4 vezes os registros disponibilizados pela FAO, confirmando assim a

subestimação das capturas globais compartilhada entre a comunidade científica especializada (Clarke et al., 2006b; IUCN, 2007).

Os tubarões oceânicos, passam grande parte ou todo seu ciclo de vida em ambientes pelágicos oceânicos (Camhi, 2008). As regiões oceânicas são menos produtivas e dispõem de menor biomassa e menor diversidade do que os ambientes costeiros (Worm, et al., 2013). Isto é compensado em termos de produtividade, através da ocorrência de zonas de ressurgência, elevações, bancos e montes submersos, ambientes que podem variar sazonalmente, ou conforme as condições oceanográficas, influenciando nos padrões migratórios de diversas espécies, como por exemplo os atuns, espadartes, tartarugas e aves (Block, et al., 2001). Os tubarões oceânicos estão intimamente adaptados a estes ambientes mutáveis, através do desenvolvimento da capacidade de migrar através destes ambientes. Estudos comprovaram que espécies como *Prionace glauca* e *Isurus oxyrinchus* (Figs. 1.2 e 1.2) realizam migrações transatlânticas (Kohler e Turner, 2008). Outra divergência adaptativa dos tubarões oceânicos em relação às espécies demersais e costeiras, esta relacionada à produção de filhotes bem desenvolvidos, com tamanhos relativamente maiores, padrão desenvolvido talvez em virtude da escassez de alimentos e necessidade de grandes deslocamentos (Snelson et al., 2008).

Em 2002, aproximadamente 25% dos tubarões oceânicos capturados no mundo corresponderam às frotas que operam no oceano atlântico (Clarke et al. 2006), devendo-se considerar ainda que esta proporção certamente é subestimada, em virtude principalmente dos descartes/*finning* (Camhi, 2008; Clarke et al. 2015). A Comissão Internacional para Conservação do Atum no

Atlântico (ICCAT-International Commission for the Conservation of Atlantic Tunas) é o principal órgão envolvido nas avaliações, manejo e recomendações relativas a pesca de tubarões oceânicos, e no ano de 2002, aprovou uma emenda proibindo a prática do *finning* em todo o Atlântico (Hazin, 2004). A preocupação desta comissão na conservação dos tubarões data desde 1995, quando, através da resolução 95-2, incentivou os países membros a disponibilizarem dados sobre as capturas (Hazin, 2004). Desde então, as questões relativas aos tubarões capturados no Atlântico, são de responsabilidade da ICCAT, que em 2008 recomendou a proibição das capturas para o todas as espécies do gênero *Sphyrna* e para o tubarão raposa-olhudo (*Alopias superciliosus*) mais recentemente para *Carcharhinus longimanus* e *Carcharhinus falciformis* (ICCAT, 2010).

Brasil e Uruguai foram as primeiras nações costeiras a desenvolverem pesca com espinhel pelágico no Oceano Atlântico Sul (Domingos et al. 2014). Como o Brasil, Uruguai e África do Sul ocupam considerável área do oceano Atlântico Sul em termos de ZEE (Zona Econômica Exclusiva). Estes países historicamente têm adotado uma política de arrendamento de barcos de outras nacionalidades através de convênios com empresas e frotas distantes (principalmente da Ásia e União Europeia) a fim de desenvolver suas próprias frotas (Ministério da Pesca e Aquicultura – Brasil, website). O arrendamento é caracterizado por um acordo entre uma empresa de pesca estrangeira (locador) e uma empresa de pesca nacional (locatário). A empresa de pesca é responsável por recursos tecnológicos e operacionais (Ministério da Pesca e Aquicultura – Brasil, website). Estimulados por esta estratégia, inúmeras frotas estrangeiras foram atraídas para o Atlântico Sul ao longo nas últimas

décadas. Dados Brasileiros apontam a ocorrência de pelo menos 23 diferentes frotas desenvolvendo pesca com espinhel pelágico em sua ZEE. Algumas frotas como Japão, Espanha e China-Taipei inclusive flutuaram seus portos de desembarques entre nações costeiras do Atlântico Sul. Por exemplo em três anos (1969-1971), China-Taipei utilizou 15 diferentes portos no Brasil, Uruguai e África do Sul como pontos de desembarque: St. Maarten, Abidjan, Cidade do Cabo, Las Palmas, São Vicente, Monrovia, Tema, Dakar, Santa Cruz, Walvis Bay, Buenos Aires, Recife, Montevideo, Paranaguá e Tenerife (Yang & Yuan, 1973; Domingos et al 2014).

As avaliações do estado de conservação das populações de tubarões, primordialmente envolvem modelos da dinâmica populacional ajustados a índices de abundancia e análise das capturas, resultando em estimativas de quanto pode ser capturado para as populações se mantidos os níveis sustentáveis de exploração, bem como em quanto as capturas devem ser reduzidas, em caso de sobre-exploração, para que as espécies possam se recuperar (Campana et al, 2008; Walker et al, 2008). A utilização de métodos demográficos vem sendo utilizada em modelos estocásticos e determinísticos, combinando parâmetros reprodutivos e de idade e crescimento à taxas de mortalidade específicas (natural, pesca, equilíbrio, etc..) (Cortés et al., 2008; Smith et al., 2008). Tais métodos permitem identificar em que momento do ciclo de vida a espécie é mais vulnerável a determinada mortalidade e conseqüentemente em que momento devera ser protegida.

Mais recentemente, devido aos inúmeros problemas nos bancos de dados disponíveis (informação para poucas espécies, quantidade

imensurável de underreport e descartes), avaliações alternativas veem sendo conduzidas através da combinação de índices de abundancia e dados demográficos, método que em elasmobrânquios iniciou-se com as perspectivas de se estimar probabilisticamente o risco de extinção das populações/espécies exploradas pela pesca onde se destacam os trabalhos de Musick, 1999; Au and Smith 1997; Smith et al. 1998; Smith et al. 2008; Au et al. 2015; Milton et al., 2002; Stobutzki et al., 2002 e 2006; Cortes et al. 2002 e 2010. Estas estimativas são traduzidas em índices que permitem categorizar em quão vulnerável é a espécie em relação a determinada atividade de risco, no caso dos tubarões oceânicos as pescarias de espinheis.

Sendo assim a presente tese visa contribuir no conhecimento da historia de vida das principais espécies, bem como utilizar de metodologias usuais (CPUE) e alternativas (demografia) para aferir sobre o estado de conservação dos tubarões pelágicos capturados por espinheis no Atlântico Sul (Fig. 1). Dentre as espécies capturadas no Atlântico Sul (Fig. 2), cujo conhecimento da história de vida e dos níveis reais de captura é particularmente limitado, encontram-se as espécies capturadas no âmbito do Projeto Tubarões Oceânicos, fundamentado nas capturas da frota espinheleira nacional e arrendada sediada nos portos de Santos e Natal. Entre essas, incluem-se os tubarões da família Lamnidae (*Isurus oxyrinchus*, *I. paucus*, *Lamna nasus*), da família Carcharhinidae (*Prionace glauca*, *Carcharhinus longimanus* e *C. signatus*), Pseudocarchariidae (*Pseudocarcharias kamoharui*), Alopidae (*Alopias vulpinus* e *A. superciliosus*), Sphyrnidae (*Sphyrna mokarran* e *S. lewini*) e a raia

Pteroplatytrygon violacea. Todas as espécies citadas acima, com exceção de *P. violácea*, se encontram listadas em categorias de ameaça segundo os critérios IUCN e avaliações recentes (IUCN, 2010).

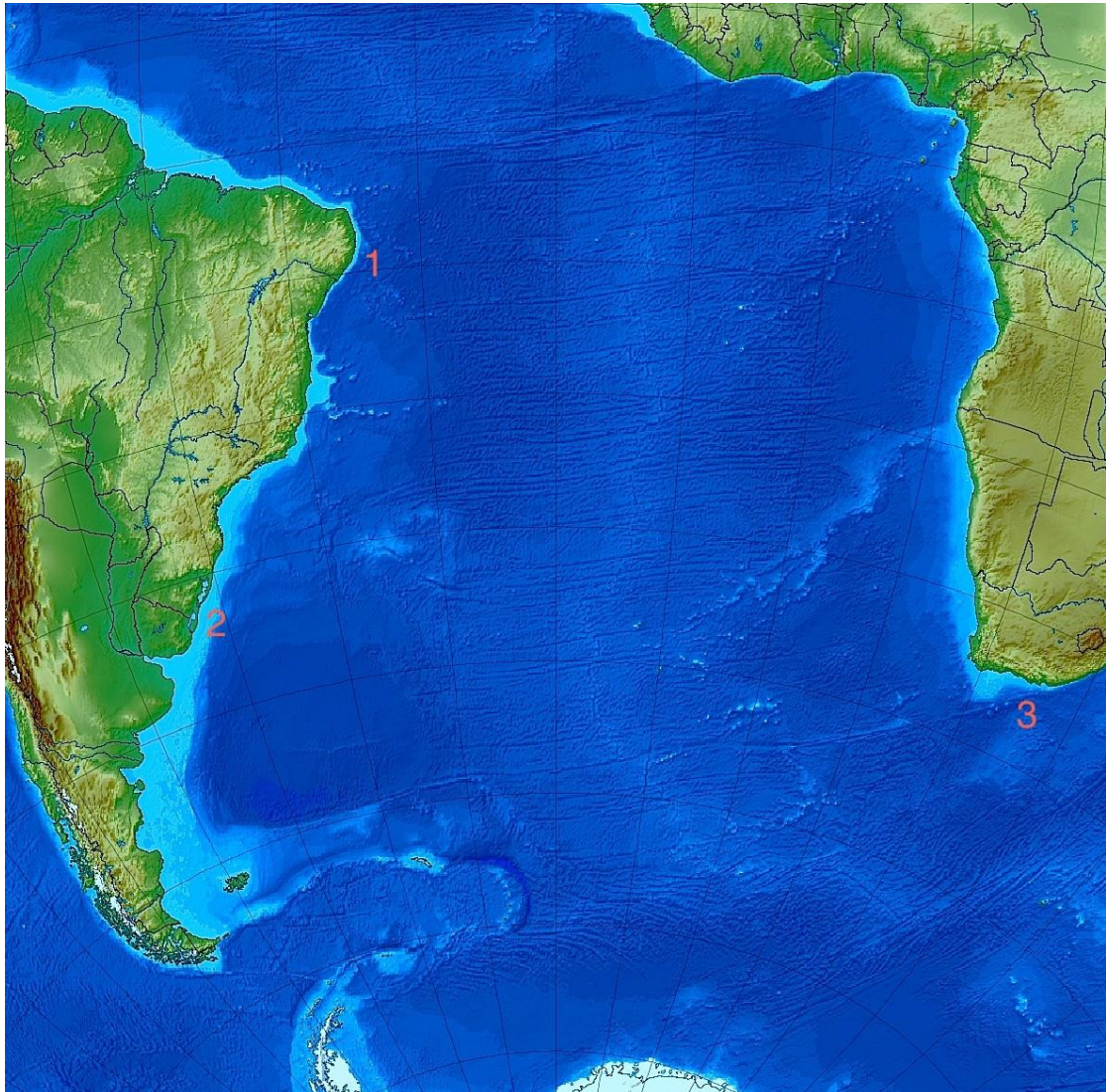


Figura 1. Oceano Atlântico Sul e principais nações costeiras ligadas a pesca de espinhel pelágico, que também serviram como portos de desembarque para frotas estrangeiras. 1 – Brasil, 2 – Uruguai, 3 – África do Sul. Fonte: Google Maps.



Figura 2. Principais espécies de tubarões capturadas por espinheis pelágicos no oceano Atlântico Sul e analisadas no presente estudo. 1 – *Prionace glauca* (tubarão azul); 2 – *Isurus oxyrinchus* (tubarão mako); 3 – *Carcharhinus longimanus* (tubarão galha-branca-oceânico); 4 – *C. falciformis* (tubarão lombo-preto); 5 – *Alopias superciliosus* (tubarão raposa olhudo); 6 – *C. signatus* (tubarão toninha); 7 – *Sphyrna lewini* (tubarão martelo) e 8 – *S. mokarran* (tubarão martelo).

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2. Age, growth and spatial distribution of the life stages of the shortfin mako, *Isurus oxyrinchus* (Rafinesque, 1810) caught in the western and central Atlantic.

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Abstract

Age and growth estimates for the shortfin mako, *Isurus oxyrinchus*, derived from vertebral centra of 467 individuals (129 males, 104 females and 234 of undetermined sex) ranging from 60 to 296 cm fork length (FL) are reported. Age structure was composed mostly of individuals aged between 0 and 12 years, for both males and females, with only one large female being older than 12 years (25 BP, 23+ years). Age at maturity was estimated at 7.5 years for males and 21.1 for females. The spatial distribution revealed that newborns were only observed below the latitude of 30° S and near the shoreline. Young individuals were more frequent than newborns and adults, being observed between latitudes 15° N and 40° S and longitudes 15° and 45° W. Sub-adult males were significantly more frequent than females. Males were distributed over a wide area, whereas females were observed only in the central part of the ocean. The population of shortfin mako sharks caught by commercial fisheries in the South Atlantic is predominantly made up of immature individuals. In this context, a better understanding of the species is required in order to implement a management plan for the population in the Atlantic. The current study is the first to analyze the spatial distribution of the phases of the life cycle of this species in the western and central Atlantic.

Introduction

The *Isurus oxyrinchus* (shortfin mako shark) is a temperate and tropical highly migratory pelagic species that inhabits preferably oceanic regions, in practically all seas from about 50 ° N to 50 ° S (up to 60 ° in some regions, i.e. northeast Atlantic) [1, 2, 3]. This shark uses a heat-exchanging circulatory system to keep internal temperature above that of the environment, and is among the most active and powerful fishes and probably the fastest shark (4, 5, 6]. Consequently, the species can perform migratory movements that encompass large oceanic extensions, being listed by the UNCLOS as a highly migratory species [1, 7, 8].

The wide distribution range of the species renders it susceptible to several fisheries, the most representative being the industrial longline oceanic fisheries which target tunas and billfishes [1, 2]. The shortfin mako is the second species of shark most caught in longline fisheries [9,10], it is highly valued for their fins and, unlike other sharks, also for their meat [11, 12]. Despite having often been reported as bycatch (or incidental), some fleets have targeted shortfin mako sharks [13]. Furthermore, because of their physical strength it is one of the most prized species in recreational fisheries, with countless fishing tournaments worldwide.

As a consequence, considerable declines in abundance have been detected over various parts of its distribution range; the pelagic longline industrial fishery is considered the biggest threat to the conservation of this species [6, 14, 15]. In 2009, the International Union for the Conservation of Nature (IUCN) ranked the species as vulnerable (VU) based on inferred declines worldwide, inadequate management and continuous fishing pressure

[6]. A major concern is that historically catches have not been properly recorded, a fact that must be taken into account [16].

In the Atlantic Ocean industrial longline fisheries respond for approximately 25% of reported global shark catches [6] [15]. The application of the ecological risk assessment approach (ERA, based on biological productivity and susceptibility), showed that shortfin mako sharks are likely to be the second most vulnerable shark to longline fisheries in the Atlantic.

The International Commission for Conservation of Atlantic Tunas (ICCAT) assessed the conservation status of pelagic sharks using data-poor methods, which led to recommendations for no retention, catch or commercialization of some species; among them all species of genus *Sphyrna*; and species *Alopias superciliosus*; *Carcharhinus longimanus*; and *C. falciformis*. Surprisingly, the shortfin mako was not included among the species recommended by the ICCAT for protection.

Lamnidae sharks are known to be among the most vulnerable species due to their high longevity, late maturity and low fertility rates [1]. However, there is a lack of biological information (and catch data) on the shortfin mako, with few studies focusing on their life history, showing contrasting results [17]. Thus, regarding reproduction, a large difference in size at sexual maturity has been reported, in comparative studies from the Atlantic and the Pacific [18, 19, 20, 21, 22]. There is no information for the South Atlantic with exception of a preliminary study using 4 pregnant females in southwest Brazil [23].

Regarding age and growth, the situation is even more complicated. Some authors assume an annual pattern of band pair (*BP*) deposition [17, 24, 25, 26, 27, 28, 29, 30, 31], while others report a biennial pattern [18, 32, 33].

Both annual and biennial patterns have been validated in studies conducted in the North Atlantic and North Pacific. Some authors [17, 29, 30] showed, through bomb radiocarbon and chemical tagging techniques, that age ring deposition occurs annually. Wells [33] showed (using chemical tagging) that juvenile shortfin mako sharks (< than 200 cm FL) deposit 2 age rings per year.

Age estimation is essential for any assessment and management of stocks as it allows the calculation of growth and other vital rates such as longevity, age at maturity and natural mortality [34]. Despite contrasting results for this species worldwide, there is only one available study for the South Atlantic [31]. Information on reproductive parameters and habitat use is even more rare. The species is considered by the International Union for Conservation of Nature as vulnerable (IUCN) in the Atlantic, though this estimate took into account studies carried out only in the northern hemisphere.

The aim of this study is to provide information on age and growth of the species in the South Atlantic, using data from the western and central parts of the ocean. In order to elucidate other important aspects, particularly regarding the use of the habitat, we additionally present the first analysis of the spatial distribution of the life stages of shortfin mako sharks, caught by commercial longline fisheries in the South Atlantic (Fig. 1).

Methods

Sample

From 2005 to 2011, shortfin mako sharks were sampled by onboard observers of the Brazilian pelagic longline chartered fleet (from Spain, Panamá, Honduras, Morocco, Portugal, United Kingdom), based in Northeast Brazil (Fig. 1). Fork length (measured from the tip of the jaw to the center of the tail indentation, following the body curve), sex and geographical coordinates (i.e. latitude and longitude) of individuals were recorded in fisheries logbooks and a block of five vertebrae was removed from a subsample. Vertebrae were collected through an agreement between the Special Secretariat of Aquaculture and Fisheries (SEAP) and the commercial fleet engaged in catching large pelagic fish for commercialization purposes (Project: Biology and Ecology of Sharks and Billfishes of the South Atlantic Ocean, n.046 / 2009). Also, the protocols for biological sampling in Brazilian EEZ has been conducted in accordance with Brazilian regulations for wildlife research and approved by the Instituto Chico Mendes de Conservação da Biodiversidade of the Brazilian Ministry of the Environment (SISBIO permit no. 49663-1). The species currently are not assessed as endangered in Brazilian waters.

Histograms of length frequency (10 cm classes) were built to visualize size structure and sex bias was analyzed through the use of the χ^2 -test. Length frequencies were also presented for males and females as an empirical cumulative distribution function (ECDF) [35], which is the proportion of individuals that are less than each observed length [36]. Size-frequencies of both sexes were compared using a two-sample *Kolmogorov-Smirnov* test.

Lengths mentioned hereafter refer to fork length (*FL*, cm) and all statistical tests were considered significant at an alpha level of 0.05.

Age and growth

Vertebrae were cleaned and fixed in 4% formaldehyde for 24h and then stored in 70% alcohol. For the analysis, each vertebra was embedded in polyester resin and sectioned to a thickness of approximately 0.3 mm using a low speed metallographic saw [34, 37]. Growth band pairs (*BP*), consisting of one wide band (opaque) and one narrow band (translucent) were counted and measured using a stereomicroscope at a magnification of 10x [37, 38]. The software Image Pro-plus was also used as an auxiliary tool. The distances from the focus of the vertebra to the outer margin of each band pair (band pair number, *BP_n*) and to the edge of the section (vertebral radius, *VR*, mm) across the corpus calcareum, were recorded [37, 38]. Only transmitted light was used so that the narrow bands would appear light, and wide bands would appear dark.

Sections were read twice at different times, without knowledge of the individual size or previous count. Whenever counts differed between the two readings, a third reading was performed by readers using the Image Pro-plus software. The age bias plot [38, 39, 40] and the average percentage of error (*APE*) [41] were used to analyze the reliability between readings [39]:

$$AE = 100\% \times \frac{1}{N} \sum_{j=1}^N \frac{|X_{ij} - \bar{X}_j|}{\bar{X}_j};$$

in which *N* is the number of vertebrae; *R* is the number of readings of individual *j*; *X_{ij}* is age *i* determined for individual *j*; and *X_j* is the mean age calculated for individual *j*.

All age groups with *APE* values greater than 10% were read again. In cases where counts differed between readers a third reading was performed for consensus. If no consensus was reached then the vertebrae were discarded from further analysis. Proportionality between the structure (vertebrae radius, *VR*) and individual size (*FL*) was calculated for the sexes separately and compared using ANCOVA.

After excluding newborns with only the birth mark on the vertebrae, the periodicity of *BP* deposition was analyzed through the monthly marginal increment ratio (*MIR*) in order to identify the period in which the narrow band is formed and a new *BP* begins to form [42]. The following equation was used: $MIR = (VR - R_n) / (R_n - R_{n-1})$; in which *VR* is the vertebra radius; *R_n* is the distance from the core to the last band pair; and *R_{n-1}* is the distance to the penultimate band pair. Monthly mean and standard deviation ($\pm$ *SD*) values were analyzed using the *Kruskal-Wallis* test [43].

Despite the tendency to use multi model approaches to fit growth data [44, 45], most fisheries stock assessment models rely only on von Bertalanffy growth function (*VBGF*) estimates [46]. Thus, in our study we chose to only use the traditional form of the *VBGF* [47, 48], as was recently recommended [46]: $L_t = L_{inf} (1 - e^{-k(t-t_0)})$ in which *L_t* = length at age *t*; *L_{inf}* = maximal asymptotic length; *k* = growth coefficient; *t₀* = theoretical age at which fish has zero length. The Kimura's likelihood test was then used to compare growth parameters (isolated and combined) between males and females.

Due to the small sample size and the lack of individuals in larger size classes (for females and males), lengths at previous ages were back calculated from centrum measurements for both sexes. As the linear fit to the

relationship between VR and length does not pass through the origin, the “size-at-birth-modified” Fraser-Lee method was employed [34, 49, 50]: $L_i = L_c + [(VR_i - VR_c)(L_0 - L_c)/(VR_0 - VR_c)]$; where: L_i = length at BP_i , L_c = length at capture, VR_c = centrum radius at capture, and VR_i = centrum radius at BP_i ; L_0 = length at birth and VR_0 = vertebrae radius at birth.

In order to identify the age composition for the entire sample the age-length key (ALK) was used [51, 52]. Length classes for shark sizes were fixed at 10 cm (FL) and contingency tables were used to plot the frequency of the individuals from specific age classes in all specific length classes [52]. Using this same method, age at maturity was estimated from size at maturity (203 and 275 cm respectively for males and females) [21]. Additionally, age at maturity was also indirectly estimated by converting median lengths at maturity from the literature to age at maturity using our $VBGF$ equations. Longevity (ω) was considered to be the age at which 99% of the theoretical maximal size is reached, estimated using the Fabens algorithm: $7.21 \ln 2/k$; as this is the most appropriate model to estimate longevity for elasmobranchs [37].

Life stages and spatial distribution

According to [53], the life span of female shortfin makos can be represented as: YOY (0-1 year), juveniles (1–17 years), subadults (17–19 years), pregnant adults (1 year), parturient adults (1 year), and resting adults (1 year; only used when a 3-year reproductive cycle is considered [54]); and for males as: neonates (0–1 year), juveniles (1–6 years), sub-adults (6–13 years), and adults (13 years and older). We adapted these definitions to

identify, through our age composition, the different stages of the life cycle for the species. After determining the geographical coordinates in which individuals were caught, we then plotted the identified life stages on a map.

Results

A total of 1,325 individuals (385 females, 498 males and 442 sex undetermined) were reported (Fig. 2). The ratio of males to females (1.29:1) was significantly different from 1:1 ($X^2 = 14.2061$, $p = 0.0001638$). Size was taken from 824 individuals, values ranging from 76 to 296 cm (Fig. 2). Through the empirical cumulative distribution function (ECDF) we identified that males and females showed similar length ranges (Fig. 3), however, the *Kolmogorov-Smirnov* test detected significant differences in ECDF between sexes ($D = 0.2464$, $p\text{-value} = 2.464\text{e-}07$).

For determining growth parameters, 467 vertebrae were analyzed from 129 males, 104 females and 234 individuals whose sex was not recorded (Fig. 4). The first distal narrow band to the focus was interpreted as the pre-birth mark (*PBM*, not considered for age assignment) and the second as the birth mark (*BM*), equivalent to age 0+. Mean radius of *PBM* and *BM* respectively were 4.52 (s.d. = 0.64) and 2.02 (s.d. = 0.29).

The relationship between *FL* and *VR* was slightly curvilinear (particularly for males), indicating allometric growth of vertebrae, so data were log transformed to allow linear regression (Fig. 5). ANCOVA detected significant effects of both *VR* and sex in *FL*, but no significant interaction, suggesting that the slope of the regression between *VR* and *FL* is similar for both sexes. Sex in turn, had a significant effect on the *FL*, which in this case can be interpreted

as a significant difference in intercepts between the regression lines of males and females.

The average percentage error (*APE*) calculated between readings was 1.39% for the entire sample and the variation among the classes was 0 % at age 0+ (1 *BP*) and 3.19% at age 23+ (25 *BP*). It is important to note that the majority of age classes ranged from 0 to 12 years, with only a single individual observed as having 25 *BP* (Fig. 6). The monthly analyses of the marginal increment ratio (*MIR*) carried out on 448 individuals (*YOY* were excluded), do not show significant differences across medians ($H = 12.46$, $d.f = 11$, $p = 0.33$). The lowest values were identified to occur in May (Fig. 7). Since *MIR* analyses were inconclusive about periodicity, age was assigned by assuming an annual pattern.

Based on *BP* counts, the observed age range for females was 0 to 23 years (73 to 296 cm *FL* respectively) and 0 to 11 for males (79 to 250 cm *FL* respectively). Observed age structure, however, was mostly composed of individuals between 0 and 12 years, for males and females. We found only one large female over 12 years of age (25 *BP*, 23+ years). Growth parameters estimated for males, females and pooled sexes through the use of the *VBGF* are available in Table 1, and growth curves (with 95% confidence bands) for females and males can be found in Figs. 8 and 9 respectively. We observed that the estimated *S.E.* of L_{inf} and t_0 for females was larger than that for males, while *S.E.* of K for females and males were similar (Table 1). Kimura's likelihood ratio test estimated a difference in the growth curves for males and females considering all parameters used ($\chi^2 = 31.63$, $p < 0.0001$),

(Fig. 10 and Table 2). The estimated growth equations are as follows:

$$\text{Males } L_{\infty} = 328.75 \{1 - e^{-K(t-t_0)}\}; \text{ Females } L_{\infty} = 407.66 \{1 - e^{-K(t-t_0)}\}.$$

Table 1. Growth parameters estimated for shortfin mako sharks caught by the Brazilian chartered longline fleet (reported by onboard observers), from 2005 to 2011, using the von Bertalanffy growth equation.

Sex	Sample size (n)		Linf (FL, cm)	K (year ⁻¹)	t0
Females	104	Estimate	407.66	0.04	-7.01
	104	S.E	97.69	0.02	1.32
Males	129	Estimate	328.75	0.08	-4.47
	129	S.E	40.84	0.02	0.73
Pooled	235	Estimate	294.90	0.08	-5.88
	235	S.E	19.26	0.01	0.55

Table 2. Kimura's likelihood hypothesis and results test for growth parameters estimated for male and female shortfin mako sharks.

Test	Hypothesis	Chisq.	p
Ho vs H1	Linf (M) = Linf (F)	1.66	0.198
Ho vs H2	K (M) = K (F)	2.88	0.090
Ho vs H3	t0 (M) = t0 (F)	3.86	0.049
Ho vs H4	M (all) = F (all)	31.63	0.000

Maximum observed ages suggest longevity of 23 and 11 years for females and males, respectively, while using the Fabens algorithm [37] estimates were 28 and 23 (females and males respectively). Age at maturity was estimated at 7.5 years for males and 21.1 for females. The age length keys for males and females which allowed the identification of age structure of the entire sample are shown in Figs. 11 and 12. Age composition for the

entire sample (as we identify differences between sexes; sex undefined individuals were discarded, $n = 442$) is shown in Fig. 13.

The spatial distribution of the phases of the life cycle (Fig. 14) revealed that the YOY (for both males and females) were only observed below the latitude 30 S and near the shoreline. Young individuals (both males and females) were more frequently observed over a wide area of the western and central part of the South Atlantic Ocean. Male sub-adults were significantly more frequent than females and were distributed over a wider area, while females were observed only in the central part of the ocean. One single adult male was observed in the central part of the ocean. Unfortunately, a large number of individuals were discarded from the spatial analysis because onboard observers did not record sex and/or lengths consistently, ($n = 1056$). Furthermore, latitude and longitude coordinates were mostly available for non measured individuals.

Discussion

The vertebrae from shortfin mako sharks proved useful for ageing since high reproducibility between readings was detected. However, vertebrae of older specimens that are more prone to reading errors (due to the overlapping of growth band pairs as a result of the approximation to asymptotic length) were rare in our sample. We believe that our sample was sufficient to estimate growth parameters for the species, given that size structure was similar to that obtained in other studies (Table 3) which used the same species and growth parameters within identical bounds. The *VBGF* provided an acceptable description of the overall pattern for males and females. Data

was better fitted for ages up to 12 years for females and up to 11 years for males (Figs. 8 and 9). At older ages confidence intervals became wider. Males ranged from 0+ to 11+ years, growing faster but reaching smaller sizes than females (for which ages varied from 0+ to 23+). Considering the maturity ages obtained in this study (7.5 for males and 21.5 for females, using the inverse *VBGF* sex specific equations; and 8 years for males and >12 for females, using the *ALKs*), the results showed that the population of the shortfin mako sharks caught by pelagic longlines in the western and central part of the Atlantic is predominantly composed of young individuals.

Between 2005 and 2011, observers reported catches of 1,326 individuals of which 884 (66%) had their sex identified, 535 (40%) had biometric information collected and 468 (35%) had vertebrae collected (from this amount, only 234 individuals had sex identified (50%), and were used in the growth analysis). Onboard observers reported that this species, unlike most other sharks, is highly valued for their meat (besides the fins), so, in order to preserve the carcasses for commercial purposes, the masters of fishing operations have often prevented the collection of biological samples. In all, from the 1,326 individuals reported, only 234 (17%) were able to be used in the analysis.

Table 3. Growth studies conducted with the shortfin mako sharks by several authors (Adapted from Natanson et al. 2006).

Study	Sex	Size range (FL, cm)	Linf (FL, cm)	k	t0	n	Area	Age at maturity	Oldest aged	BP/years	Longevity
Pratt and Casey (1983)	Male	69-238	302	0.266	-1	49	Western NA	3	4.5	2	10
	Female		345	0.203	-1	54		7	11.5	2	14
Cailliet and Bedford (1983)	Male	80.6-293	298	0.072	-3.75	44	Pacific, California	7	17	1	38
	Female										
Chan (2001)	Male	66-274	267	0.312	-0.95	24	Pacific, Australia	-	7	2	9
	Female	74-314	349	0.155	-1.97	52		-	10	2	17
Hsu (2003)	Male	72.6-250.9	321.8	0.049	-607	133	China	13	23.6	1	3
	Female	72.6-314.9	403.62	0.04	-5.27	174		18	30.6	1	-
Ribot-Carballal et al. (2005)	Male	68.6-264	375.4	0.05	-4.7	109	Pacific, Baja	7	18	1	55
	Female							15			
Bishop et al. (2006)	Male	100-347	302.2	0.052	-9.04	145	Pacific, New Zealand	8	29	1	48
	Female	-	820.1	0.013	-11.3	111		20	28	1	219
Natanson et al. (2006)**	Male	72-260	253.3	0.125	L0	118	Western NA	8	29	1	21
	Female	64-340	365.6	0.087	L0	140		18	32	1	38
Cerna and Lincandeo (2009)	Male	70-258*	268.07*	0.087	-3.58	243	Pacific, Chile	-	25	1	-
	Female	69-300*	295.73*	0.076	-3.18	304		-	25	1	-
Semba et al. (2009)	Male	73- 265*	255*	0.16	L0	128	Western and central NP	8	14	1	-
	Female	73-330*	340*	0.09	L0	147		20	19	1	-
Doño et al. (2014)	Male	81-250	416	0.035	-6.18	116	Western SA	8	18	1	-
	Female	101-330	580	0.021	-7.52	126		18	28	1	-
This study	Male	79-250	328.74	0.08	-4.47	129	Western and central SA	7.5	11	1	23.2
	Female	73-296	407.65	0.04	-7.08	109		21.1	23	1	28.2

* FLs were estimated from study-specific conversion equations; ** Use of Gompertz GF;

* FLs were estimated from study-specific conversion equations; ** Use of Gompertz GF;

Although our growth models have shown a good statistical fit – mainly between 0 and 11 years of age – and the reproducibility index between readings has been low, we identified some inconsistency between observed ages and lengths, particularly for females at ages 1, 3, 4 and 5 (Fig. 8) and males at ages 2 and 7 (Fig. 9).

Our study was inconclusive in relation to the BPs deposition pattern by using the indirect validation method (*MIR*). As suggested by [54], the low sample size associated with the long sample period probably influenced our results due to variability in the periodic deposition of bands that should not always happen in the same time scale. As in the study of Dono et al. (2014), we assume for the western Atlantic an annual pattern of growth band deposition, in light of other studies conducted in other areas of the Atlantic. This question, however, still remains uncertain in the South Atlantic, suggesting the necessity of new and more detailed studies for the species as pointed out by Dono et al. (2014).

Over the last years several authors have analyzed the growth of the shortfin mako sharks and the results of these studies differ considerably from one another (Table 3). The main sticking point is in relation to the pattern of deposition of growth bands. Pratt and Casey (1983) and Chan et al. (2001), for example, assumed biennial band deposition through the use of indirect validation techniques, and thus their oldest ages were considerably (2-fold) different from studies that have assumed an annual pattern (Table 3). This pattern was recently corroborated by Wells et al. (2013), in Southern California, through the injection of chemical markers in juvenile shortfin mako sharks. Authors found that individuals between 0 and 5 years of age, deposit two complete bands a year. In contrast, the vast majority of authors had assumed an annual pattern [24, 25, 26, 27, 28, 31]. Studies that used indirect validation methods to infer about periodicity have found significant results (1 BP deposition by year), as described in [25, 26, 28]. The period (month) of BP deposition varied among them. Finally, studies performed by several authors [17, 29, 30] through the application of the radiocarbon bomb technique (one of the most reliable direct validation methods) found that BP deposition in shortfin mako sharks occurs annually.

Dono et al. (2014) recently reported a similar size structure for the species. Although the authors used an area that was more concentrated in the South (Southern Brazil and Uruguay) and found a female that was bigger than our biggest female (Table 3), the presence of large and mature females was generally uncommon, as in our study. This may be an indication of a pattern for the western Atlantic, given that Campana et al. (2005) also reported a similar range of sizes in the northwest Atlantic. However, although

the size structure was similar, age structure reported by Dono et al. (2014), particularly for larger male individuals, differed from those reported in our study. While authors reported that the largest male (250 cm) and largest female (330 cm) were 18 and 28 years of age, respectively, we found that our biggest male (250 cm) and our biggest female (296 cm) were 12 and 23, respectively (Table 1). Also, differently from [31], we identified the *PBM* (pre-birth mark) in all vertebrae analyzed ($n = 468$). In both studies, however, the predominance of young and sub-adults in relation to YOYs and adults is clear. According to Goldman et al. (2012), the lack of individuals of lower and higher age classes may affect the estimation of reliable growth parameters.

Considering the enormous variability in relation to the growth parameters estimated for the species by different authors, in different study areas and time scales, it is virtually impossible to identify which studies more accurately reflect the growth pattern for the species. Although minor variations are acceptable, species must have a similar general pattern. As a highly migratory species, which performs large-scale movements to accomplish stages of their life cycle, mako sharks are present in several types of habitat [57]. These changes in habitat may be the cause of the variability found in growth parameters for this species.

Our study found that the part of the population of mako sharks that is exploited by commercial fisheries in the South Atlantic is predominantly made up of immature individuals, thus unable to reproduce (Fig. 14). We conclude that there is an immediate need to better understand habitat use and behavior for this species, considering that large mature females are rare – a result corroborated by Dono et al. (2014) in the same geographical area. According

to a recent ecological risk assessment study, shortfin makos are expected to be more vulnerable than other species caught by commercial longlines in the Atlantic [15]; and studies focused on abundance have shown that their population is currently depleted. Because life history parameters are fundamental to any kind of management of fish stocks, we call upon the scientific community, particularly the authors who have worked with shortfin mako sharks, to the need of an integrated appraisal to elucidate why so many differences have been reported.

Acknowledgments

We shall also like to thank the Ministry of Environment (MMA) and the Ministry of Fisheries and Aquaculture (MPA) of Brazil, and the onboard observers that collected data. Special thanks to CAPES, CNPq and FACEPE for the fundings.



Fig. 1. Sampling area showing the start-of-set geographic position (black points) where shortfin mako sharks were caught and reported. Biological information was collected by on board observers in the Brazilian leased fleet from 2005 to 2011

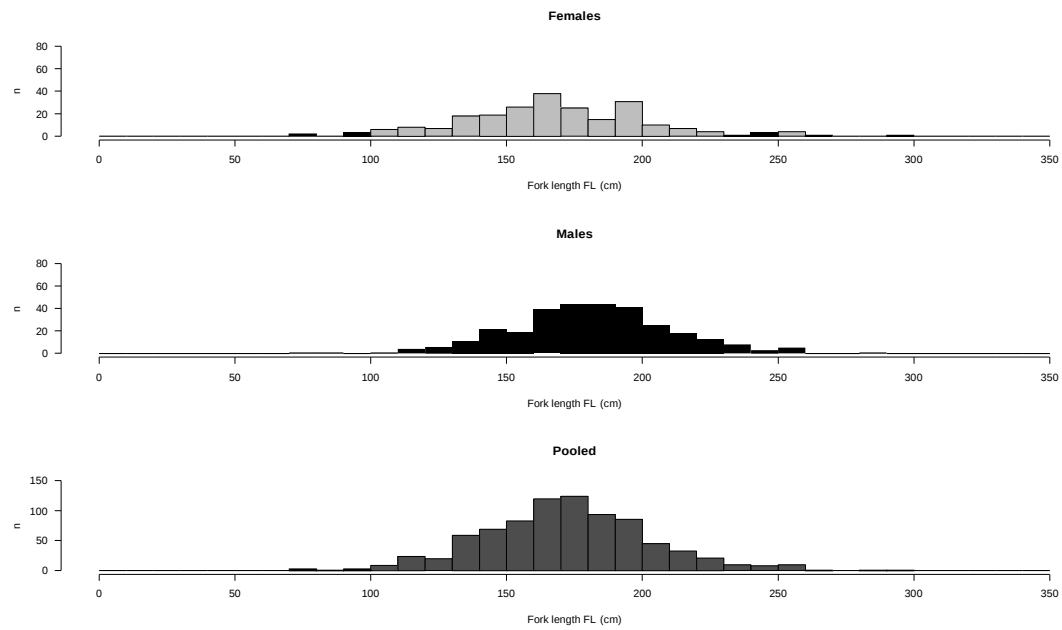


Fig. 2.Length frequency distribution for shortfin mako sharks sampled by on board observers on the Brazilian leased fleet from 2005 to 2011. Light grey bars: females; black bars: males; dark grey: combined sexes.

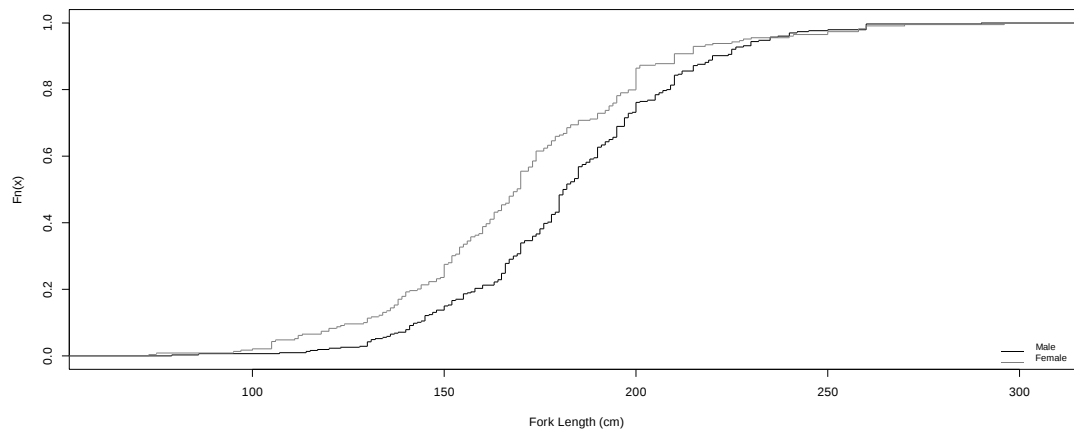


Fig. 3. Empirical cumulative distribution function (ECDF, FL cm) of shortfin mako sharks caught by the Brazilian chartered longline fleet (reported by onboard observers) from 2005 to 2011.

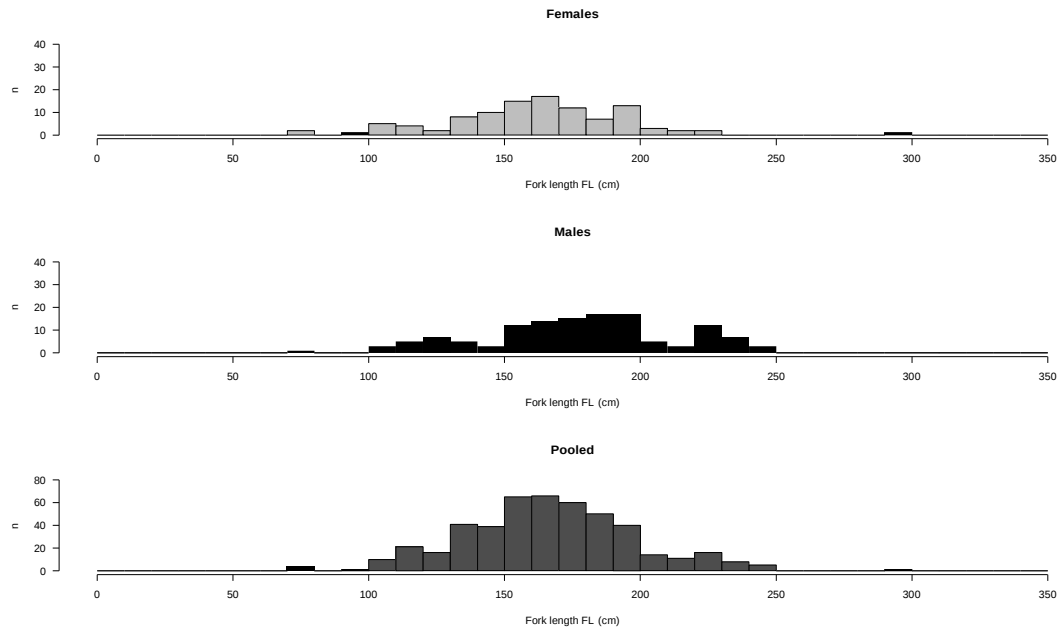


Fig. 4. Range of lengths (vertebrae sample) of shortfin mako sharks caught by the Brazilian chartered longline fleet, reported by onboard observers from 2005 to 2011.

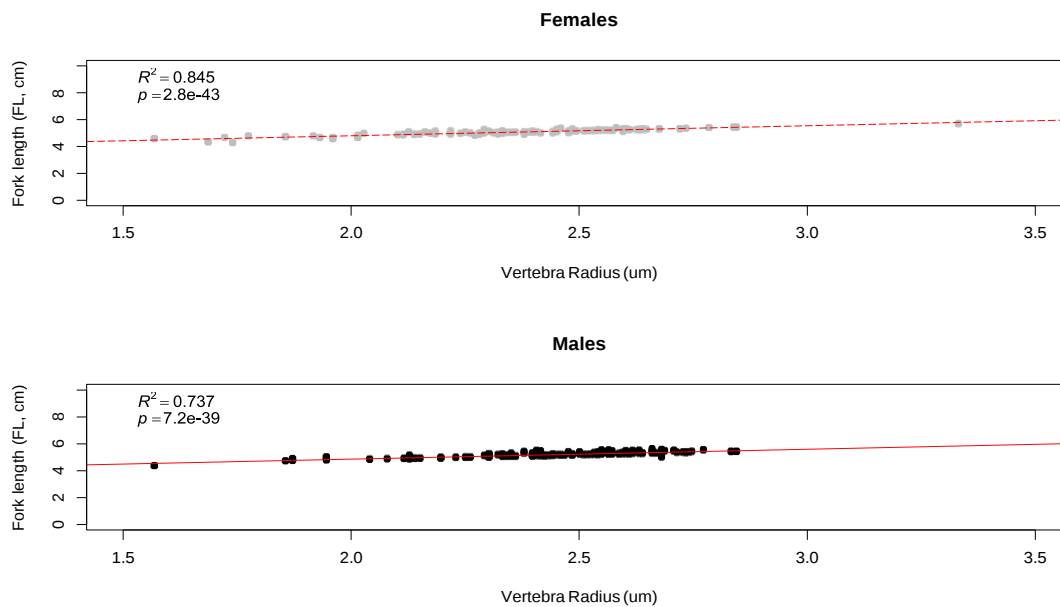


Fig. 5. Fork length vs. vertebra radius of shortfin mako sharks caught by the Brazilian chartered longline fleet, reported by onboard observers from 2005 to 2011.

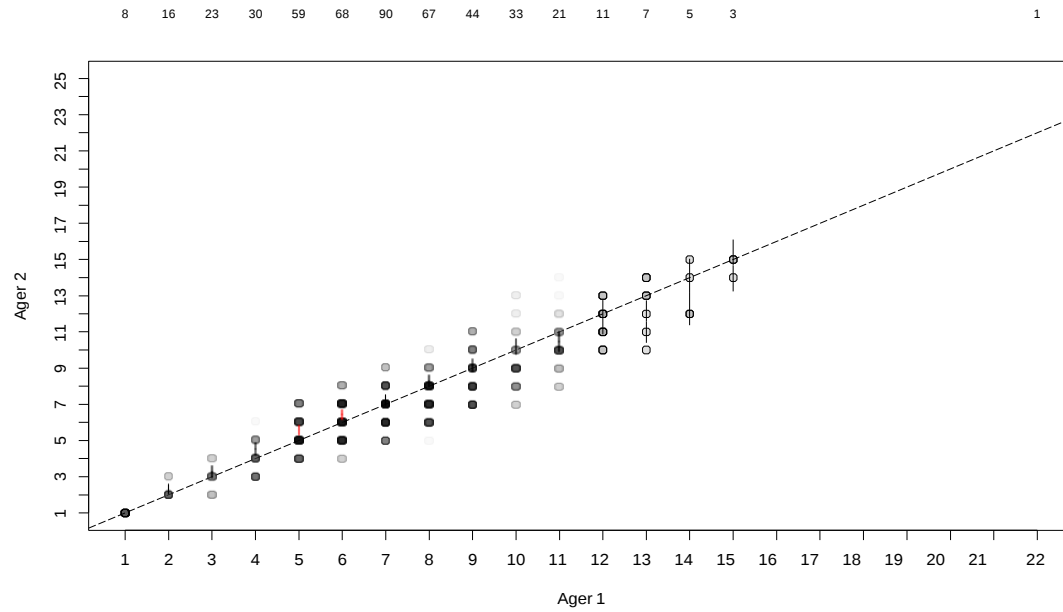


Fig. 6. Age bias plot of shortfin mako sharks caught by the Brazilian chartered longline fleet.

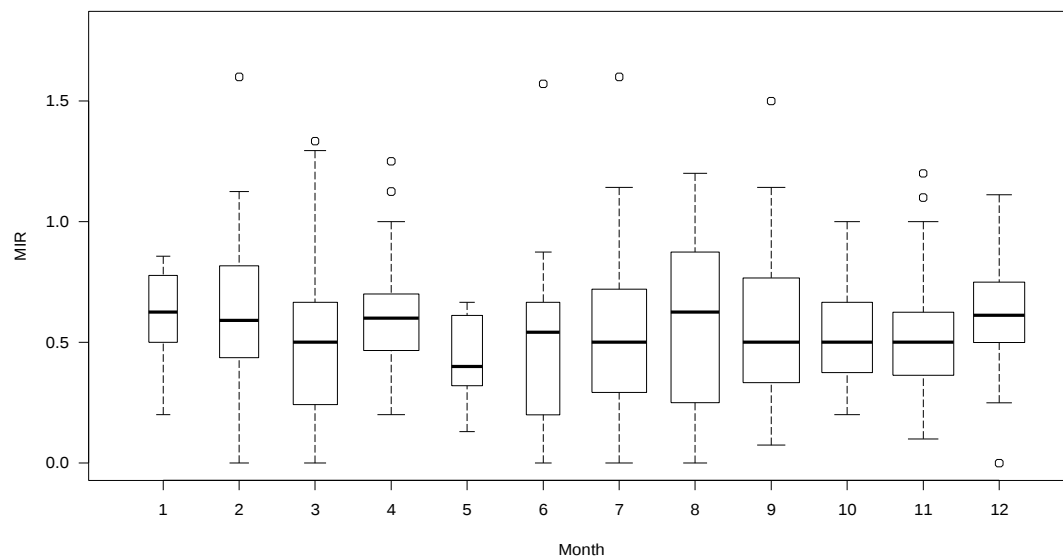


Fig. 7. Median vertebral marginal increment ratio (MIR) by month for shortfin mako sharks caught by the Brazilian chartered longline (n = 448).

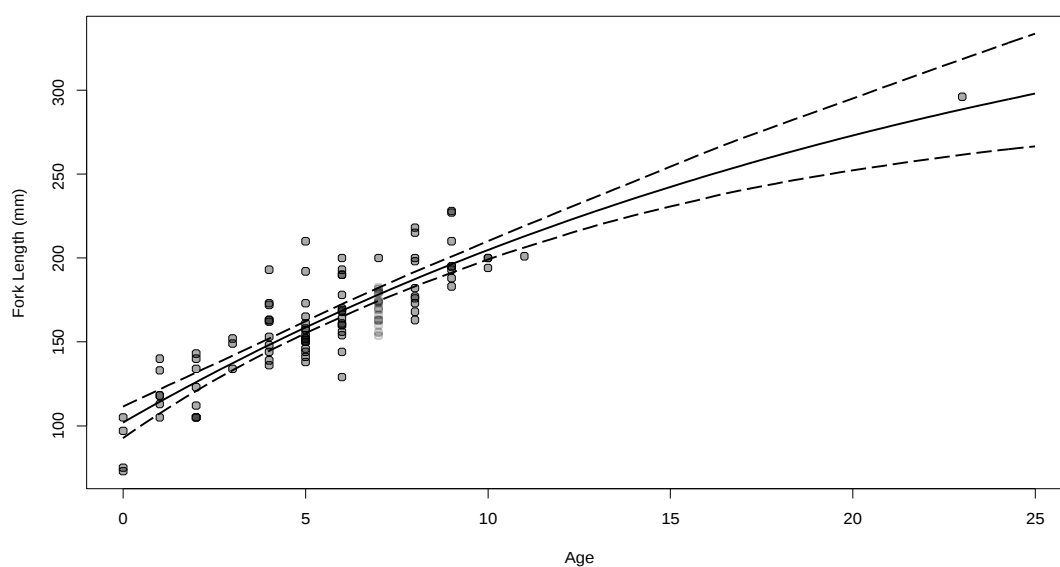


Fig. 8. Length versus age with superimposed best-fit von Bertalanffy growth model for female shortfin mako sharks. Dashed line means 95% confidence interval.

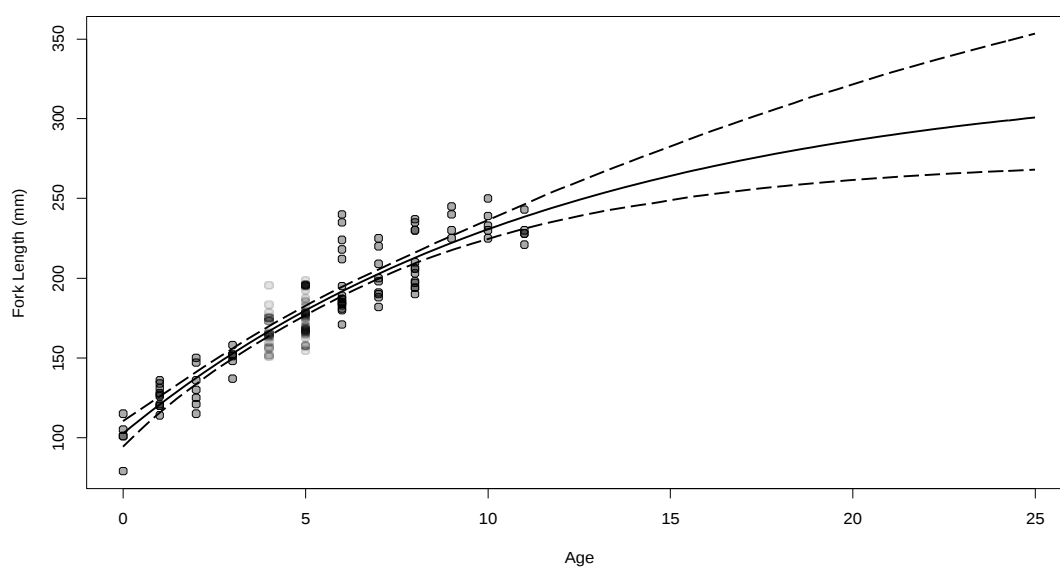


Fig. 9. Length versus age with superimposed best-fit von Bertalanffy growth model for male shortfin mako sharks. Dashed line means 95% confidence interval.

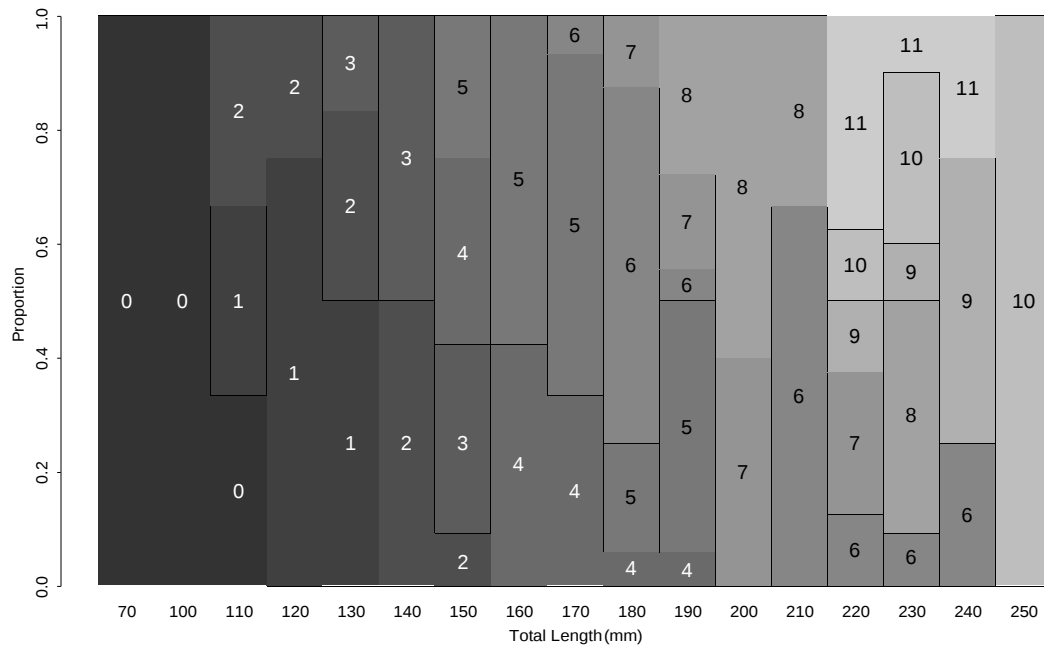


Fig. 12. Age-length key for male shortfin mako sharks caught by the Brazilian chartered longline fleet (reported by onboard observers) from 2005 to 2011.

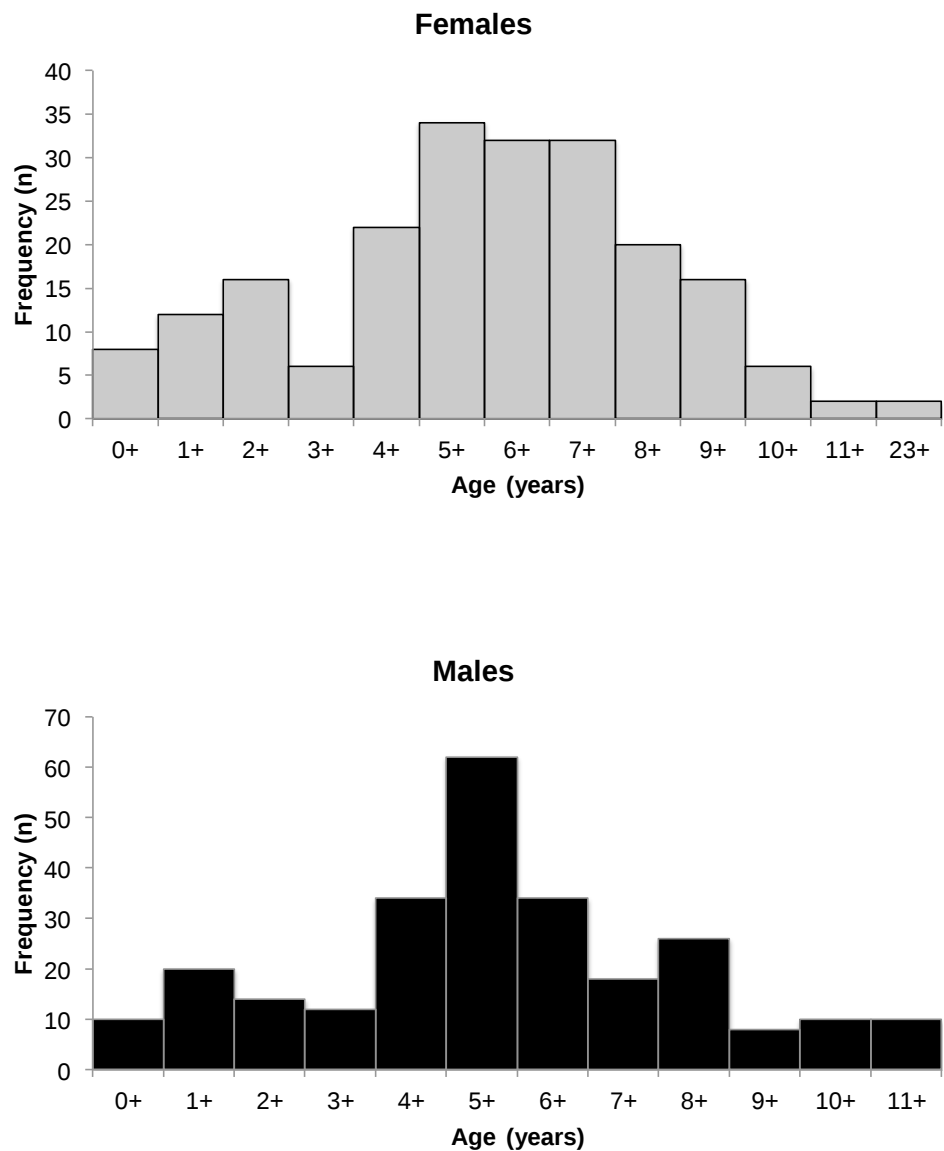


Fig. 13. Age composition (entire sample) of shortfin mako sharks caught by the Brazilian chartered longline fleet reported by onboard observers from 2005 to 2011.

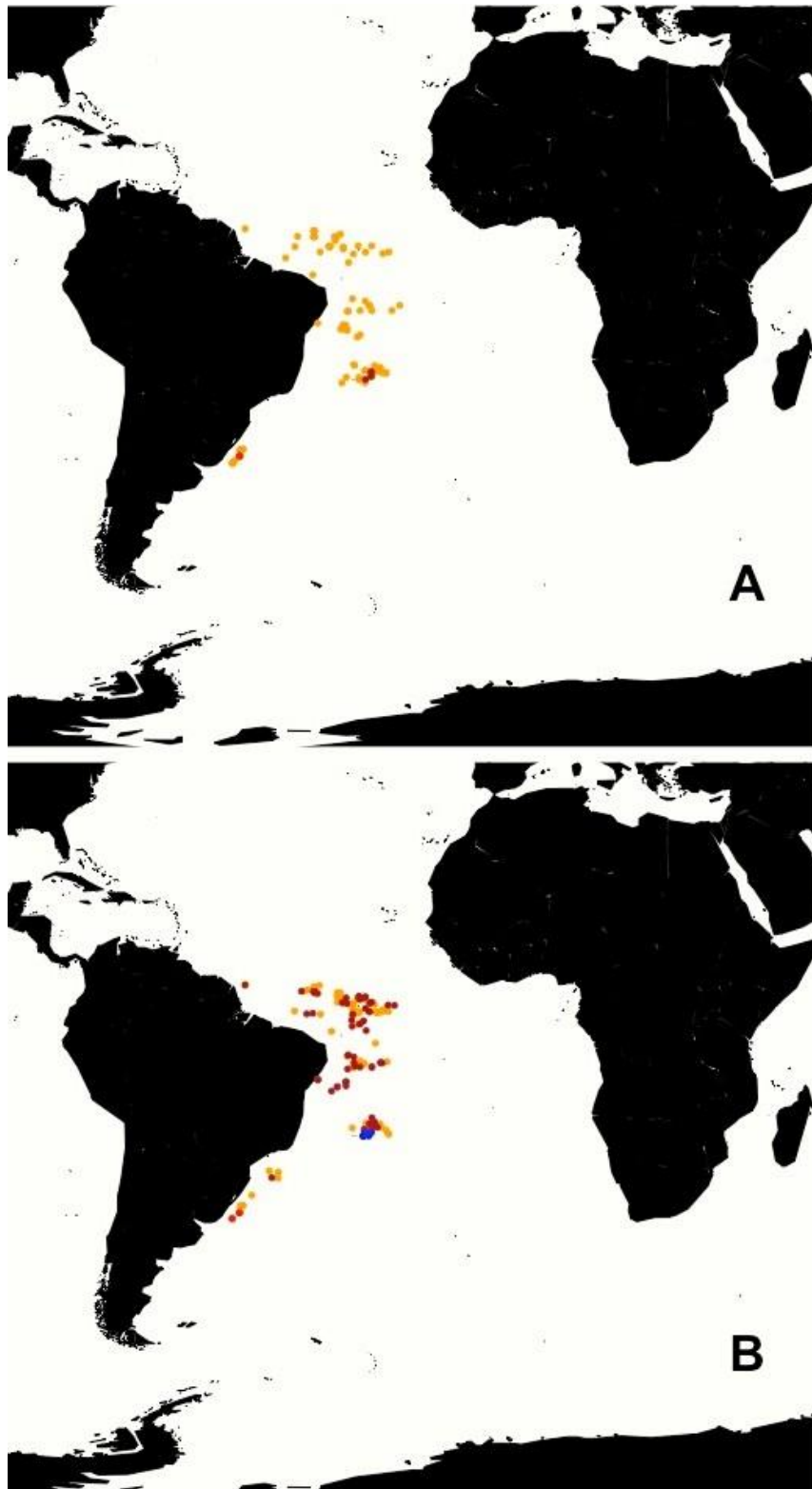


Fig. 14. Spatial distribution of shortfin mako sharks: (A) Females, (B) Males. Life stages: red = YOY; orange = juveniles; brown = sub-adults and blue = adults.

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3. Trends in the exploitation of South Atlantic shark populations

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Abstract

Approximately 25% of globally reported shark catches occur in the Atlantic pelagic longline fisheries. Strong declines in shark populations have been detected in the North Atlantic while in the South Atlantic the situation is less clear, although fishing effort has been increasing in this region over the last 50 years. Here we provide a synthesis of information on shark catch rates (based on 871,177 sharks caught on 86,492 longline sets) for the major species caught by multiple fleets in the South Atlantic between 1979 and 2011. Three distinct phases in these data are identified: a first phase, characterized by a few fleets mainly fishing for tunas; a second phase, where several fleets were fishing for tunas, swordfishes and sharks; and a third phase, where fleets were reduced but still fishing for multiple species and restrictive measures were being implemented. Generalized linear models were used to standardize

catch rates and identify trends in each of these phases. Shark catch rates increased in the first phase, when fishing effort was low, then decreased in the second phase, when fishing effort was rapidly expanding, and remained stable in the third phase, when fishing effort was again low. Our results indicate that most shark populations in the South Atlantic are currently depleted, but can recover where fishing effort is reduced accordingly. In this context, it is of concern that comprehensive data collection and management of these fisheries has ceased.

Keywords: Pelagic sharks, conservation, South Atlantic Ocean, recovery, threatened species

Introduction

According to the International Union for Conservation of Nature (IUCN), 29% of non-data deficient sharks are currently threatened with extinction (Dulvy et al. 2014). The cumulative extinction risk for these species is substantially higher than for most other marine vertebrates due to a vulnerable life history, including slow growth, late maturity, and low fecundity (Smith et al. 2008). These features make sharks and other elasmobranchs particularly vulnerable to increased mortality from fisheries (Smith et al. 2008).

Historically, however, most shark species have been a low priority for regional fisheries management organizations (RFMOs). Consequently, there is a paucity of data for many species and regions. Existing data are often sourced from fisheries logbooks (reported by fishers, landing reports, or onboard observers), and may be incomplete (Pauly et al. 1998, Clarke et al. 2006, Worm et al. 2013), have low taxonomic resolution, and be influenced by

technological changes in fishing gear and preferences for target species (Harley et al. 2001; Baum et al. 2003).

Surveying sharks over large ocean regions is both expensive and impractical (Baum et al. 2003, Jensen et al. 2012) and in many cases, fisheries-dependent data are often the only available source of information to estimate trends in relative abundance and spatial distribution of oceanic sharks. Although more problematic than survey data (Bishop, 2006; Jensen et al. 2012), fisheries-dependent data can in some cases be used to estimate abundance indices using standardization and appropriate statistical methods (Harley et al. 2001; Maunder & Punt 2004; Bishop, 2006).

Over the last few decades, population declines of oceanic sharks have largely been attributed to longline fishing in the North Atlantic, Pacific, and Indian oceans (Worm et al. 2013; Dulvy et al. 2014). High seas longlining in the Atlantic Ocean generates about 25% of reported global shark catches (Clarke et al. 2008). Fishing effort has been high on both sides of the Atlantic Ocean, and intensified around the same time (Meneses et al. 2000). However, most of the information on the effect of fishing on large pelagic sharks comes from the North Atlantic Ocean, while data analyses from the South Atlantic Ocean (SAO) are fragmented and pertain only to the most abundant species (Mourato et al. 2008; Carvalho et al. 2010; Tolotti et al. 2013; Carvalho et al. 2014).

There is international and national concern over the conservation status of pelagic sharks with respect to shark by-catch by international fishing fleets operating in the SAO. In fact, since 2008, ICCAT, the main RFMO managing pelagic fisheries in the Atlantic has made recommendations for no retention,

catch or commercialization of some of the species exploited by these fisheries including all sphyrnids (hammerhead sharks), bigeye thresher, oceanic white-tip and silky sharks (*Alopias superciliosus*, *Carcharhinus longimanus* and *C. falciformis* respectively) (Tolotti et al. 2015). The Brazilian Ministry of the Environment (MMA) has attempted to protect these species by including them in the national list of endangered species (ICMBio, 2014). However, there is strong opposition from the fishing industry and some ordinances guaranteeing protection to endangered species in the country are currently canceled (Dario et al. 2014).

The simultaneous exploitation of the SAO by several fishing fleets led to high levels of underreporting and the migratory patterns of the major species had also hampered analysis in the SAO (Hazin et al. 2008). Coastal nations from the SAO, such as Brazil, Uruguay, South Africa and Namibia, have historically allowed their ports to be used by international fleets from Asia and the European Union (Domingo et al. 2014) in order to build their own fleets and meet quotas established by RFMOs. These fleets changed their target strategies over time due to market demands, technological advancements and declines in abundance of commercial species (Hazin et al. 2008).

Here we have responded to this information gap by assembling a large database of longline catch and effort data on multiple species of large pelagic sharks recorded in logbooks of 21 fishing fleets operating in the southwestern Atlantic over a time span of 33 years. All species analyzed in this study are listed in Annex 1 of United Nations Convention on the Law of the Sea (UNCLOS) as highly migratory. Highly migratory species usually performs large-scale movements to accomplish stages of their life cycle and thus occur

both in open ocean and within EEZs (Maguire et al. 2006). Recent studies suggest that there is a single stock of blue sharks in each hemisphere of Atlantic and these sharks uses a wide tropical and subtropical area of these oceans across their life cycle (Carvalho et al. 2011; Vandeperre et al. 2014; Queiroz et al. 2012). Habitat use of shortfin mako, hammerhead, oceanic white-tip sharks, and other pelagic charcharhinids is expected to occur on similar scales (Maguire et al. 2006, Camhi et a. 2008). Yet, as there are no evidences of marked populations structure in SAO we assume that our results are indicative of broader populations trends across the SAO. Our data however, was provided exclusively by Brazilian institutions, and covers mainly the western and central part of the SAO (Fig. 1).

We extracted trends in standardized catch rates for the major species, and used these data together with information from the literature and other sources to infer changes in patterns of exploitation and their effect on the species population abundance. This work is timely, as Brazil will re-assess the conservation status of marine fauna in the next few years (MMA, 2014), despite the fact that onboard observer programs have been cancelled, and national data collection from fisheries has mostly ceased as of 2012 (Dario et al. 2015). This study may contribute a baseline population assessment for globally threatened oceanic shark species and inform further data collection, management and conservation decisions.

Methods

Data

Our database consists of a compilation of data from fishing logbooks, reported by fishing companies, fishers and onboard observers from different fleets. We obtained these data from different Brazilian institutions monitoring longline fisheries in the western and central SAO (Supporting information). We screened all sources of data to identify commonly logbook errors (i.e. set coordinates on land and typing errors), presence of duplicates, and created an identification code for each set (*setid*, identified by the combination of coordinates, boat name, flag name, institutional source and date). Suspect or repeated sets were then discarded.

Variables useful for catch rate standardization were retained from the source datasets (Supporting Information) and then merged in a new database. The resulting database included: counts of sharks caught in individual sets (*n*), numbers of hooks used on individual sets (*H*), the identity of the fishing fleet (*flag*), year of the set (*Y*), month of the set (*m*) and geographic coordinates (latitude = *lat*, longitude = *lon*). Additionally, we incorporate a variable that takes into account seasonal variations through the use of a sinusoidal function of month (*m*):

$$f(m) = \sin\left(\frac{2\pi}{12}m\right) + \cos\left(\frac{2\pi}{12}m\right).$$

Species

Problems with fishermen and observers incorrectly or inconsistently identifying sharks led us to combine some sharks into broad groups. Blue (*Prionace glauca*), shortfin mako (*Isurus oxyrinchus*), silky (*Carcharhinus*

falciformis), oceanic white-tip sharks (*C. longimanus*) and bigeye thresher (*Alopias superciliosus*) were analyzed at the species level since they tend to be identified reliably. The three species of hammerhead sharks were analyzed at the genus level (*Sphyrna* spp.). Sharks not properly identified were included in the group "other sharks" (most of them are part of the Carcharhinidae family), while together all *Carcharhinus* spp. (except the blue sharks) were also included in a group referred to as "grey sharks" (Table 1).

Dividing data into phases of exploitation

In the southwestern Atlantic, significant changes have occurred over time in total fishing effort, species targeting, and catch reporting. Changes in fishing methods were associated with the introduction of new technologies, management measures and market demands. Furthermore, even within single fishing fleets there were mixed fishing strategies (Supporting information).

We first performed an exploratory data analysis, initially using catches for all fleets referring to all shark species. The total number of sharks caught, hooks used, and nominal catch rates recorded (number of sharks divided by the number of hooks) each month were plotted over time (Fig. 2). Then we used the number of sets deployed by each fishing fleet each year to build a mosaic plot of fishing operations showing the relative proportion of fishing effort deployed by all fleets in the databank (Fig. 3). From these results and information from the literature, we identified three distinct exploitation phases (Fig. 2, Table 2, Supporting information).

Phase A (1979–1997) — Data reported by five fleets by approximately 40 vessels of whose 28 came from Japan and two operated under multiple flags (Figs. 1 and 3). During this phase, overall, fleets employed similar gears. Vessels operated during the early morning, using deep multifilament longlines (>200m) with J hooks and small fish as bait, targeting tunas (Hazin et al. 2008). Logbooks had a high incidence of zeros, but a lower portion of missing values (Table 2).

Phase B (1998–2007) — 20 fleets reported data during this phase, of which 18 recorded shark catches (Figs. 2 and 3). Out of these fleets, data from 10 were retained for modeling. Approximately 100 vessels reported data to the government of Brazil. At least 10 vessels changed flags during this phase, most of them Spanish. Fishing practices also changed during this phase, mainly because of the introduction of monofilament lines, and circle hooks to target swordfishes and sharks. Nominal catch rates of sharks during this period were considerably higher than for the other phases (Table 2). This was an effect of an increased use of swordfish longlines which have a greater shark by-catch than tuna longlines and an indication of more directed shark fishing, probably due to the increasing demand for shark fins from Asian markets in the 1990s (Clarke et al. 2006). We detected that while the proportion of zeros decreased, the proportion of missing values increased relative to phase A (Table 2).

Phase C (2008–2011) — Since 2005 Brazil has required that all foreign vessels host onboard observers but the program effectively started in late 2007. Concurrently, the landing port of the foreign fleets has shifted from Southern to Northeast Brazil. Thirty vessels of three fleets report shark data

during this phase (Spain, Brazil, and Honduras). Fishing fleets still had floating fishing strategies targeting tunas, swordfishes and sharks. Catch rates were in general considerably lower than in phase B, but slightly higher than in phase A. Because of onboard observers on foreign fleets, we expected this phase be the most reliable over the entire period. However, the proportion of missing values (Table 2) was considerable higher than for phases A and B.

Modeling trends in catch rates

Our database had a large portion of zeros and missing values (Table 2). Zeros could be missing values, unreported catches replaced by zeros in the logbooks, or true zero catches. Nonetheless, previous research has shown that logbooks seem fairly accurate for positive catches (Baum et al. 2003). Since it was impossible to distinguish between real zeros and missing values in our database, we removed them and used zero-truncated negative binomial distributions to model only the positive catches (Baum et al. 2003; Martin et al. 2005).

We fit zero truncated negative binomial generalized linear models to the data of each species for each fishing phase. All covariates were used to build an initial model. We included the logarithm of the number of hooks as offset terms in order to model catch rates while still retaining the probabilistic nature of the response variable. The basic model structure was: $\log(\mu) = X\beta + \log(H)$. Where X is the matrix of explanatory variables, β is the vector of parameters (explanatory variables, fixed effects), H is a vector of the number of hooks (treated as an offset), and μ is the expected catch (response variable).

We then refined our models by iteratively selecting the most appropriate combination of explanatory variables, according to their statistical significance (Table 3). Year (Y) was modeled both as continuous (to obtain overall estimates and percentage rate of change) and categorical explanatory variable (to detect annual variability). Annual mean catch rates were predicted by fixing the explanatory variables at their median or otherwise common (i.e., for factors) values, for 1000 hooks. Percent changes in catch rates were estimated by comparing the predicted catch rates of the initial and final year of the series.

Since fleets had fished in different ways, for different targets over time, we chose to keep fleets in the model (Supporting information), even if this process structure reduced the amount of data we could analyze (i.e. fleets with less than 2 years of data were excluded). To further justify this decision, we explored the outcome of using our models with the exclusion of this variable and also by creating fleets-specific models. When the fleets variable was excluded, catch rates fluctuated with the number of fleets fishing in any particular year.

Finally, to test the sensitivity of our models to the timing of the identified phases we refit the models to alternative phases (increasing and decreasing their cutoffs by 2 years) and also by excluding phases (Supporting information).

Results

Between 1979 and 2011, 871,177 sharks were reported on 86,492 longline sets performed by 339 vessels of 20 different fleets using a total of

142,450,304 hooks. We have identified the occurrence of 13 species; five of these were common enough to be analyzed at the species level. Our exploratory analyses revealed, however, that the structure of the dataset (n, effort and nominal catch rates) for all species changed considerably between 1997 and 1998 and again between 2007 and 2008 (Fig.2). These changes resulted from both changes in reporting as well as variation in catchability.

Phase A

During this phase, pelagic shark catch rates increased for most species, from 1.4-fold (blue sharks) to 10-fold (shortfin mako sharks). Only silky sharks showed a significant decrease (66%), from 0.67 sharks per thousand hooks in 1979 to 0.22 in 1997 (Fig. 4, Table 3). Bigeye thresher sharks showed no significant trend in catch rates (from 0.0157 sharks per thousand hooks in 1979 to 0.0109 in 1997), however, none of the explanatory variables had statistical significance to explain catch rates of this species (Table 2).

In general, fleets caught an average of 4.75 sharks per thousand hooks in 1979 and 9.22 in 1997. Yet there were statistically significant differences in catch rates among the three fleets we modeled, with Brazil generally reporting the highest catches (Table 3). Also, for most species we detected a significant seasonal difference of catch rates throughout the years (Table 3).

Phase B

In phase B, all sharks showed decreasing trends in catch rates except for oceanic white-tip and bigeye thresher. Changes ranged from a 94% decline of silky sharks and a 98% decline for all other requiem sharks (“other

sharks”) to an estimated 71%, 58%, and 20% decline of hammerhead, shortfin mako, and blue sharks respectively (Fig. 4). Except for blue sharks, catch rates of all species were considerably lower than in the previous phase (1.04 sharks per thousand hooks in 1998 and 0.11 in 2007). We also noted a sizeable discrepancy between the catch rates of shortfin makos and the ‘other sharks’ between phases A and B. Differences in shark catch rates among fleets were detected, with Uruguay and Honduras having generally higher catch rates than the others. In this phase we detected a significant spatial component of the catch rates (latitude and longitude were significant in all models), and a seasonal component for all sharks except oceanic white-tip and bigeye thresher sharks (Table 3).

Phase C

In phase C we detected an increase in catch rates for silky (7.7-fold) and bigeye thresher (6-fold); a 2.4-fold increase for blue; an almost 2-fold increase for the ‘other sharks’; and the decrease of oceanic white-tip (-14%) and hammerhead sharks (82%). Shortfin mako catch rates showed no trends during this phase. Except blue and oceanic white-tip sharks, all species had very low catch rates throughout the considered time period. Conversely, catch rates of blue sharks reached their highest level ever, with about 12 sharks per thousand hooks in 2011 (Fig. 4). All fleets differed significantly in their levels of catch rates, with Honduras generally recording higher shark catch rates than the others (Table 3). Yet seasonal and spatial terms (coordinates and month) were important for all species except oceanic white-tip and bigeye thresher sharks (Table 3).

Discussion

From a conservation point of view, our most important result is that most of the species analyzed have declined precipitously from considerable fishing pressure and absence of fishing control, particularly in phase B. These declines coincided with significant fishing effort expansion, and apparent directed fishing for sharks by some fleets. Considering the percentage rate of change (Supporting information) between the last year of phase A in relation to the last year of the phase B we detect that with exception of blue and bigeye thresher, all species have declined by more than 90%. According to IUCN red list guidelines, particularly regarding criteria B and C (continuing decline, IUCN, 2012), our results indicate that shortfin mako and silky sharks might be endangered. Note that both species were recently assessed as near threatened in Brazil (Table 1).

Methodologically, the analysis of different phases allowed us to model catch rate trajectories in generally homogeneous fishing regimes. This approach allowed us to cope with differences in catchability across phases, major gear modifications (e.g. introduction of monofilament longlines in phase B), and implementation of more restrictive monitoring (e.g. the introduction of onboard observers on foreign vessels). Yet, we could not completely control for temporal changes in target strategies within some fleets and phases. Moreover, quotas and fishing licenses can be traded among signatory members of ICCAT, masking the real identity of some fleets. China-Taipei, for example, fished using flags from St. Vincent and Grenadines and also Belize, targeting tunas, billfishes and sharks (ICCAT, 2013). We also report results of models including these two variables, both as fixed and random effects, and

found that estimates were improved by using boats and flags as nested random effects (Supporting information). However these models only converged for the most data-rich species, blue sharks.

During the initial period of industrial exploitation (phase A), fishing was relatively moderate, with only Japan and Brazil reporting data regularly. Target species were mainly albacore (*Thunnus alalunga*) and bigeye tunas (*T. obesus*) (Hazin et al. 2008). The Brazilian fleet followed Japanese tuna fishing methods until the beginning of the 1990s. Except for silky and bigeye thresher, our analyses indicate that standardized catch rates for this phase showed increasing trends for all species (Fig. 4, Table 3). We hypothesize that the increases detected for these species are partly an effect of changes in reporting. Sharks were not commercially important in South America until the late 1980s and ICCAT only made the consistent reporting of shark catches mandatory from the 1990s (Hazin et al. 2008). Thus, our results for phase A are probably influenced by a systematic increase in the recording of sharks in logbooks, and should be interpreted in this light.

After the 1990s, significant declines in some high-value North Atlantic target species such as bluefin tuna (*Thunnus thynnus*) and swordfish (*Xiphias gladius*) led ICCAT and other North American and European fisheries management bodies to impose quota restrictions and tighter fisheries regulations, which resulted in a significant displacement of fishing effort from the North to the South Atlantic Ocean (Hazin et al. 2008, Worm et al. 2009, Worm & Branch 2012). In the South Atlantic, this expansion of fishing effort coincided with the introduction of new technologies and the rise of a global fin

market, which incentivized directed fishing for sharks. All species showed decreasing trends in phase B, except for bigeye threshers (Table 3).

When fishing effort decreased again and restrictive measures were adopted (use of onboard observers in foreign vessels, phase C), catch rates were found to increase or stabilize. However, onboard observers covered only 45% of the vessels fishing in this phase (all from Spain and Honduras), and we cannot exclude the possibility that catch rates were influenced by changes in reporting. Particularly for shortfin mako and silky, increasing trends in catch rates were observed in phase C, however when absolute estimates were compared with those from phase B, the species had actually declined. During phase C, most species (except blue and white-tip) had standardized catch rates very close to 0, but generally increasing. This result suggests that the level of shark exploitation observed in phase B likely led to significant changes in the shark assemblage in the SAO. Yet, increased regulation and oversight in phase C, and lower fishing effort may have set the stage for species recovery, as has been noted for other species and regions (Lotze et al. 2011).

Blue shark is one of the few species of pelagic sharks species caught in the SAO for which there is a considerable amount of information (Carvalho et al. 2014). Stock assessment models fitted to blue shark catches recorded by the Brazilian longline fisheries concluded that the population of blue sharks in SAO was above stock biomass at which maximum sustainable yield is achieved (Carvalho et al. 2014). In contrast, Pons and Domingo (2008), analyzing data from Uruguay between 1992-1998, concluded that catch per unit effort (CPUE) dropped by 30%; a trend similar to our estimate for phase

B. In the North Atlantic, Aires-da-Silva et al. (2008) and Baum and Blanchard (2010) reported declines of 30% and 53% for the time series from 1957 to 2000 and 1992 to 2005 respectively.

According to recent ICCAT ecological risk assessment, shortfin makos are predicted to be more vulnerable than other species (Cortes et al. 2010). While contrasting trends were previously observed in the SAO (Hazin et al. 2007; Mourato et al. 2008), the species has experienced significant decreases in other areas. In the NAO, catch rates of makos declined about 35% between 1992 and 2005 (Baum & Blanchard 2010). The extremely low catch rates observed in our study in phases B and C suggest that the shortfin mako population in the SAO is depleted. Unlike other sharks, this species is targeted for its fins and its meat, especially from Spanish vessels (Hazin et al. 2008).

Many species of the genus *Carcharhinus* are often combined into generic groups because of challenges in species identification. This practice hampers the identification of trends in species-specific catch rates. For example, in the NAO, silky sharks were grouped with other sharks (night and dusky), which declined by more than 75% between 1992-2005 (Baum & Blanchard 2010). Here, we could estimate a species-specific trend for the species and found a significant decline, which was similar to trends for the 'other sharks' and 'grey sharks' groups (which mostly consisted of Carcharhinids, especially *C. falciformis* and *C. signatus*). Similar to other analyses (Tolotti et al. 2013), our results were inconclusive for white-tip sharks. In the NAO, this species declined by 50% from 1992 to 2005 (Baum & Blanchard, 2010), and now is currently protected by several RFMOs including

ICCAT (Tolotti et al. 2015). Nonetheless, our results are generally supported by independent assessment approaches, such as demographic and productivity/susceptibility analysis (PSA), which have indicated that some Carcharhinidae species are at risk in the SAO, specifically *Carcharhinus signatus*, *C. galapagensis*, *C. falciformis*, and *C. longimanus* (Santana et al. 2011; Luiz et al. 2011; Cortes et al. 2010).

Similar to our results, hammerhead sharks (*S. lewini*, *S. zygaena*, *S. mokarran*) have shown a remarkable decline of more than 75% in the NAO (Baum & Blanchard 2010). *S. lewini* composes most of the catches for this group, at least in southern Brazil (around 80%). However, not being able to extract species-specific trends for these three vulnerable species of large sharks (Gallagher et al. 2014) is concerning. *A. superciliosus* was the least frequent species among those analyzed, and was often discarded according to the data collected by onboard observers. Like mako sharks, thresher sharks also have a more vulnerable life history and are currently depleted in several locations (Cortes et al. 2010).

We conclude that the SAO has experienced significant levels of depletion during the mid-1990s/mid-2000s “gold rush” on sharks and other pelagic species, but that decreasing effort and increasing regulation is suggesting that species can recover from previous depletion. In light of our findings we are concerned about the cessation of systematic data collection from fleets fishing over Brazilian jurisdiction since 2012 (Chao et al. 2015), and the cancellation of onboard observer programs at the same time, which renders any further monitoring of SAO shark populations difficult or impossible. In addition, fisheries statistics in countries such as Uruguay,

South Africa, Namibia and Argentina are also limited (Hazin et al. 2008), and there are no integrated management plans between the SAO nations.

While countries such as Brazil, Uruguay and South Africa have been creating favorable conditions for many fishing fleets to expand in the area, proper monitoring of these fleets has been inconsistent. This situation severely impedes proper stock assessment and hence the evaluation of current conservation status for threatened species in SAO waters. In this respect it is significant that some shark species of South America (such as *Carcharhinus plumbeus*, *C. porosus*, *C. galapagensis*, *Sphyrna tudes*, *S. tiburo*, *S. lewini*, *S. media*, *S. tudes*, *S. zygaena*, *Isogomphodon oxyrinchus*, *Galeorhinus galeus*, *Mustelus fasciatus*) are at even greater risk from unregulated and unobserved fishing, and may be close to extinction in Brazilian waters (ICMBio, 2014). In accordance with the recently implemented Brazilian NPOA (ICMBio, 2014) we suggest that further actions should focus primarily in improving fisheries statistics, management and monitoring to inform proper recovery strategies for depleted shark populations in the South Atlantic Ocean.

Supporting information

Further information about phases and fleets (literature review, empirical and statistical rationale, Appendix S1), sensitivity of our models to the timing of the identified phases (Appendix S2), sensitivity of our models to the inclusion of the variable (Appendix S3), detailed information of the data sources (Appendix S4), percentage rate of change between phases (Appendix S5). The authors are solely responsible for the content and

functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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Table 1. Description of the data set, including family, species, common names, and total number of sharks reported by multiple fleets in the western and central SAO between 1979 and 2011; IUCN Red List categories (global and for Brazil). IUCN Red List categories: EX - Extinct, RE - Regionally Extinct, EW - Extinct in the Wild, CR - Critically Endangered, EN - Endangered, VU - Vulnerable, NT - Near Threatened, LC - Least Concern, DD - Data Deficient.

Family	Species	Code	Common name	n reported	IUCN	Brazil (MMA)
Isuridae	<i>Isurus oxvrinchus</i>	SMA	Shortfin mako	35411	VU	NT
	<i>Isurus paucus</i>	LMA	Longfin mako	3	VU	DD
Alopiidae	<i>Alopias superciliosus</i>	BTH	Bigeye thresher shark	5114	VU	VU
	<i>Alopias vulpinus</i>	ALV	Common thresher shark	1	VU	VU
Pseudocarchariidae	<i>Pseudocarcharias kamoharui</i>	PSK	Crocodile shark	30	NT	DD
Sphyrnidae	<i>Sphyrna lewini</i>	SPL	Scalloped hammerhead	50900	EN	CR
	<i>Sphyrna zygaena</i>	SPZ	Smooth hammerhead	1	VU	CR
	<i>Sphyrna mokarran</i>	SPK	Great hammerhead	1	EN	EN
	<i>Sphyrna spp.*</i>	SPX	Hammerhead sharks	63989	-	-
	<i>Prionace glauca</i>	BSH	Blue shark	445587	NT	NT
Carcharhinidae	<i>Carcharhinus falciformis</i>	FAL	Silky shark	26177	NT	NT
	<i>Carcharhinus longimanus</i>	OCS	Oceanic whitetip shark	3288	VU	VU
	<i>Carcharhinus signatus</i>	CCS	Night shark	132	VU	VU
	<i>Galeocerdo cuvier</i>	TIG	Tiger shark	15	NT	NT
	<i>Carcharhinus spp.**</i>	CAX	Grey sharks	135345	-	-
Not identified	Other sharks	OTHSARKS	Other sharks	105183	-	-
All sharks	-	-	-	871177	-	-

* Composed by all sharks identified as *Sphyrna* spp., plus *S. lewini*; ** Composed by all sharks identified as *Carcharhinus* spp., plus *C. falciformis*, *C. signatus* and *C. longimanus*

Table 2. Catch rates, zero observations, and missing values in the data set. Mean annual catch rate is calculated for each species or species group in each phase (**A** - 1978–1997; **B** - 1998–2007; **C** - 2008–2011). The proportion of zeros for each species in each phase, and the proportion of missing values (NA) is calculated for each species and phase.

Species	Phase	Annual Catch Rate	% of 0	% of NA
<i>P. glauca</i>	A	0.97	61.62	0.03
	B	26.06	44.27	0.48
	C	10.16	6.48	9.84
<i>I. oxyrinchus</i>	A	0.08	90.39	0.91
	B	2.07	76.97	5.18
	C	0.35	21.46	62.21
<i>Sphyrna spp.</i>	A	0.06	94.70	0.96
	B	4.31	83.58	7.12
	C	0.17	23.19	71.93
<i>A. superciliosus</i>	A	0.01	96.56	2.22
	B	0.35	88.10	8.63
	C	0.05	23.97	75.32
<i>C. longimanus</i>	A	0.02	97.95	1.22
	B	0.13	88.53	9.79
	C	0.19	19.81	71.47
<i>C. falciformis</i>	A	0.44	91.02	1.22
	B	0.55	88.22	9.76
	C	0.36	22.82	71.15
Other sharks	A	0.80	78.84	0.00
	B	5.24	79.68	0.02
	C	0.43	22.51	64.84

Table 3. Generalized linear model results. Models assumed a zero truncated negative binomial distribution in the three different phases of exploitation (**A** - 1979–1997; **B** - 1998–2007; **C** - 2008–2011). Model coefficients for each species and covariate are shown, as included in the final model. Values highlighted in bold indicate significant covariates ($p < 0.05$); (-) indicates covariates that were dropped from the final model.

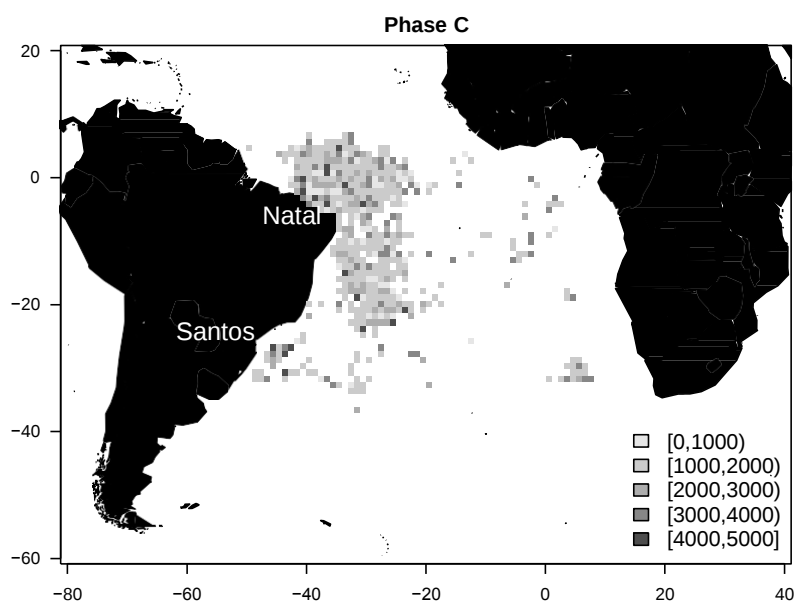
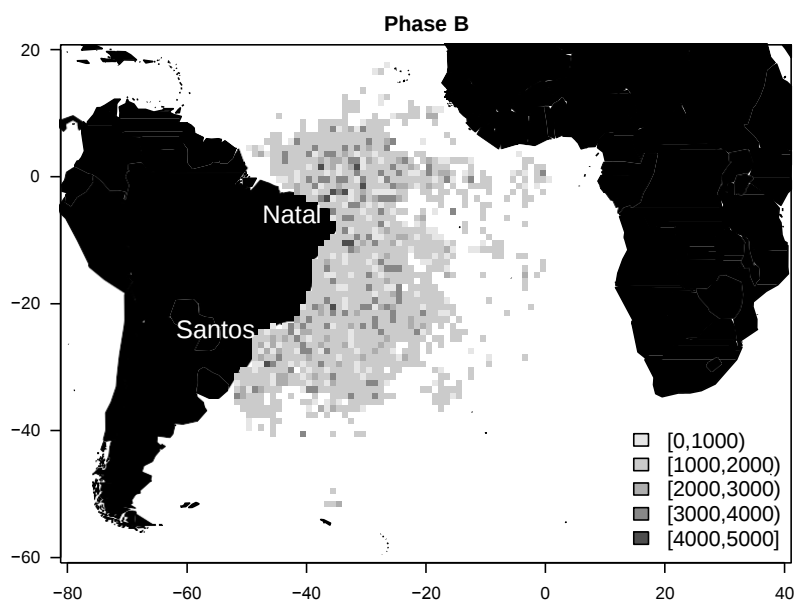
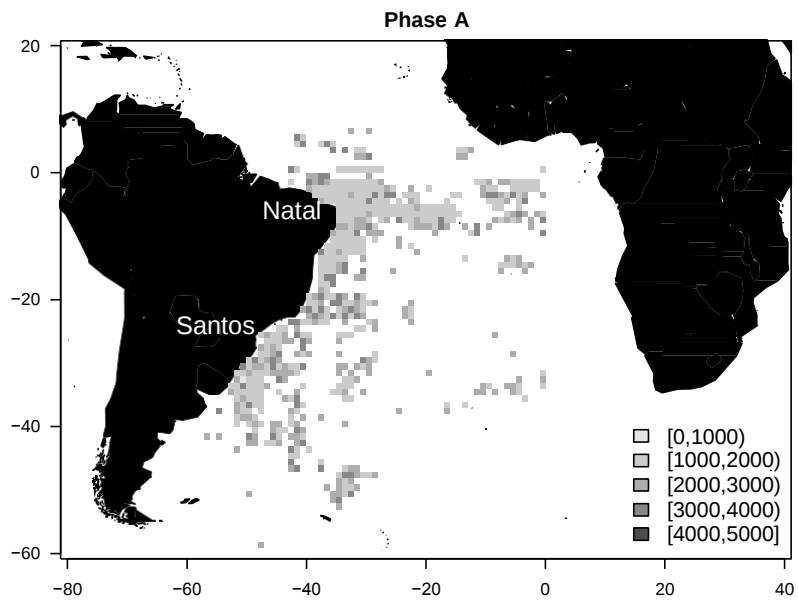
Phase	Covariates	<i>P. glauca</i>	<i>I. oxyrinchus</i>	<i>C. falciformis</i>	<i>C. longimanus</i>	<i>A. superciliosus</i>	<i>Sphyrna spp.</i>	Grey sharks	Other sharks
A	Intercept	,44.7511	,233.3007	95.8047	,215.0000	20.5884	,124.0212	,34.2307	,143.0000
	y	0.0197	0.1133	,0.0523	0.1050	,0.0205	0.0572	0.0139	0.0692
	flagBRAOBLZ	,0.4468	,0.0630	,2.3011			,1.5963	,2.6995	,2.9900
	flagBRAOJPN	,0.4671	,0.5226	,1.4733	,0.2420	12.7292	,0.0589	,1.6392	,1.5600
	flagBRAOKOR	,0.3356	,0.7170	,0.7169			,1.8919	,1.7086	,2.6800
	flagBRAOTAI	,0.6128	,0.3391	,1.1246		12.8006	,0.8673	,2.1920	,2.4600
	smonth	,0.0756	,0.1084	,0.5807	,0.1140	0.1567	,0.3521	,0.3946	,0.2360
	cmonth	0.0320	0.2089	,0.1159	,0.3850	,0.1292	0.1100	0.0762	0.1870
	lat				0.0105			,0.0107	,0.0052
	lon	0.0065			0.0035				
	Number of observations (> 0)	6007.0000	1419.0000	1275.0000	122.0000	159.0000	660.0000	4773.0000	3543.0000
	Negative binomial dispersion parameter	0.3929	0.1626	0.0067	2.0551	2.0538	0.0067	0.0436	0.1137
	std. err	0.0228	0.0660	0.0000	2.2712	0.1329	0.0000	0.0169	0.0226
	Loglikelihood	'16562.6000	'2428.0200	'3889.8000	'136.4510	'284.4600	'1314.5700	'13375.4000	'9439.8300
B	Intercept	54.0941	153.0000	607.8928	152.7711	,79.1000	201.4102	740.3746	752.2789
	y	,0.0293	,0.0820	,0.3090	,0.0827	0.0330	,0.1068	,0.3751	,0.3810
	flagBRAOBLZ	,0.2454	,0.4120					,2.4187	,2.4313
	flagBRAOBOL	,2.0221	3.2500					,1.5056	,1.5049
	flagBRAOCAN	,0.4854	,33.0000					,1.5856	,1.6610
	flagBRAOESP	,0.3999	0.6750	,0.3085	1.0601	,0.1420	0.9334	,0.1181	,0.0460
	flagBRAOHND	,0.0763	0.4100	0.6187	1.9565	0.6770	0.8614	,0.1167	,0.2972
	flagBRAOISL							,0.2072	,0.1987
	flagBRAOKIT	,1.9172	,1.6500					,1.3036	,0.9908
	flagBRAOMAR	,0.4385	,1.2800					,0.0042	0.0376
	flagBRAOPAN	,0.9188	,1.0900	1.0873	0.6019	1.1700	1.9692	,1.0219	,1.0445
	flagBRAOPRT	,0.2557	1.0300	,1.3169	1.1304	,1.3600	0.8028	,0.6405	,0.5806
	flagBRAOTAI	,1.3638	0.4590	,2.3518		,16.1000	,2.7085	,1.7648	,1.7382
	flagBRAOUK	,1.1813	,2.5100					,1.3438	,1.2537
	flagBRAOURY	0.3850	,0.1150	1.3129	3.4547	4.9800	3.0264	0.0322	,0.1499
	flagBRAOUSA	,0.0725	,0.5520	0.9240			,1.2035	1.0060	0.9665
	flagBRAOVCT	,0.7797	0.1530	0.1477		,1.7500	,1.6990	,1.2514	,1.3896
	flagBRAOVUT	,0.9522	,1.1500	,1.3390		,0.2470	,1.4995	,2.0145	,3.1735
	smonth	,0.0800	,0.0581	,0.3507	,0.1664	,0.0970	,0.2420	0.1319	0.1605
	cmonth	,0.2506	,0.0785	0.3110	,0.0151	,0.1270	0.5842	0.1248	0.1080
	lat	,0.0551	,0.0296	,0.0138	,0.0421	,0.1750	0.0080	,0.0066	,0.0079
	lon	0.0373		,0.0353	,0.0643	0.0062	,0.0684	,0.0363	,0.0332
	Number of observations (> 0)	35071.0000	11284.0000	1208.0000	1001.0000	1955.0000	5478.0000	15331.0000	13067.0000
	Negative binomial dispersion parameter	0.7051	0.0067	0.0067	0.0535	0.0067	0.0067	0.0067	0.0067
	std. err	0.0113	0.0000	0.0000	0.1117	0.0000	0.0000	0.0000	0.0000
	Loglikelihood	'103005.0000	'17556.2000	'2533.6700	'1084.7600	'2254.8200	'11232.8000	'33504.5000	'29303.8000
C	Intercept	,508.0000	,197.0000	,1190.0000	95.3279	,1204.8140	1006.3533	,471.0710	,749.5987
	y	0.2510	0.0922	0.5810	,0.0509	0.5960	,0.5104	0.2264	0.3650
	flagBRAOESP	0.1060	,0.3110	,1.5400	,2.3208	,3.9030	,1.6186	,0.6159	,0.3924
	flagBRAOHND	0.2900	1.4000	1.3800	,1.8535	,19.4610	,0.0340	,0.0527	,0.7462
	smonth	,0.6620	,0.6940	0.3630	,0.0116		0.8548	,0.0143	,0.0728
	cmonth	0.0190	0.1630	,0.8040	,0.5926		,1.2776	,0.3561	,0.3328
	lat	,0.0214	,0.0353	,0.1210	,0.0711		0.0574		
	lon	0.0443	0.0169	,0.2620	,0.0265		,0.2469	,0.1738	,0.1590
	Number of observations (> 0)	5270.0000	1030.0000	380.0000	550.0000	45.0000	308.0000	1606.0000	798.0000
	Negative binomial dispersion parameter	1.3490	0.0067	0.0067	0.9633	0.0067	0.0067	0.0067	0.0067
	std. err	0.0451	0.0000	0.0000	0.3746	0.0000	0.0000	0.0000	0.0000
	Loglikelihood	'15570.5000	'994.1840	'585.3090	'583.2840	'72.3580	'307.7280	'2404.6200	'1129.7700

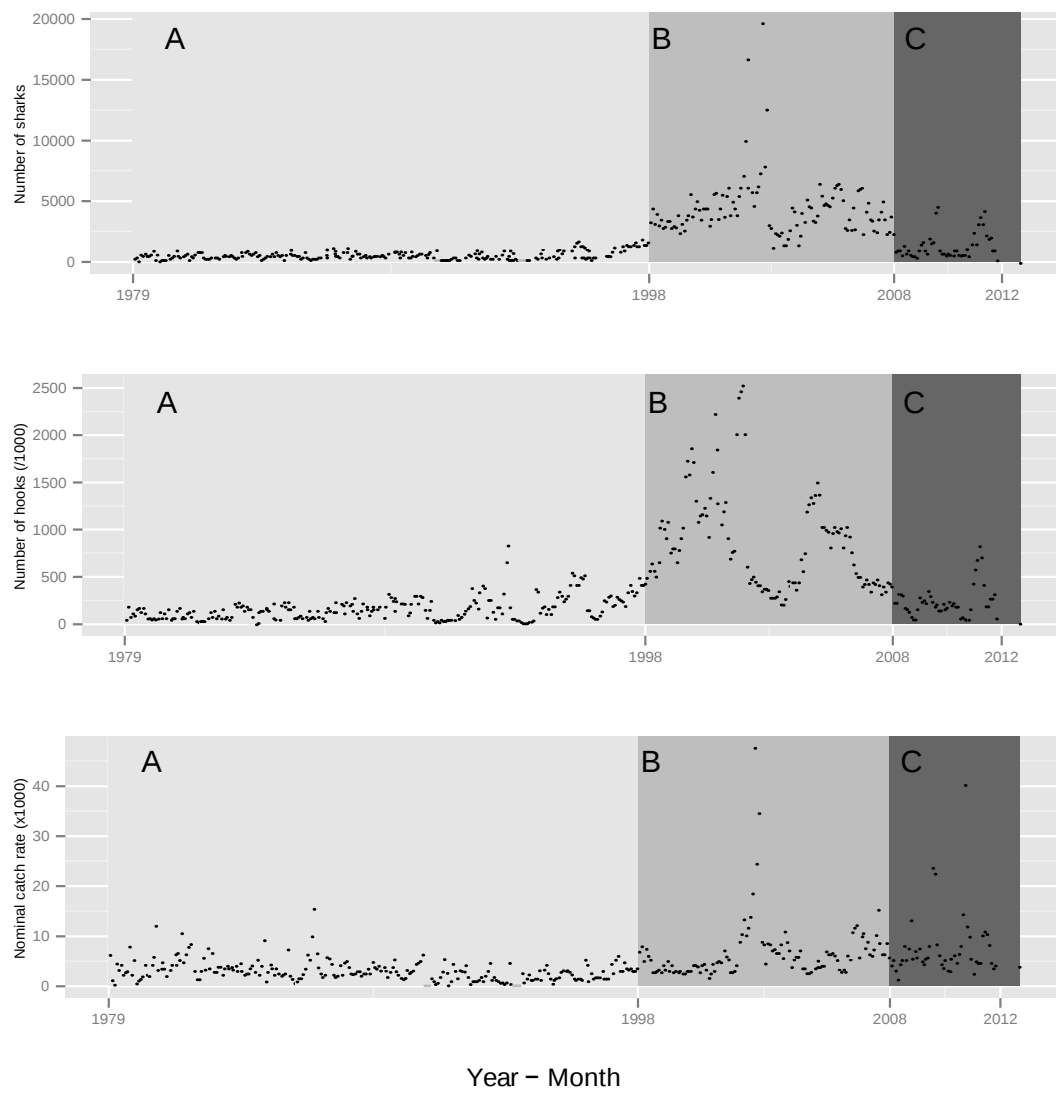
Figure 1. Spatial distribution of reported longline fishing effort in the three identified phases; **A** - 1979–1998; **B** - 1998–2007; **C** - 2008–2011. Only fleets that reported more than 2 years of fishing were included.

Figure 2. Total number of sharks (n), fishing effort (number of hooks), and nominal catch rates (total number of sharks divided by total number of hooks multiplied by 1000, by month) reported by the multiple fleets for Brazilian institutions between 1979 and 1997 (phase **A** - light grey), 1998–2007 (phase **B** - medium grey), 2008–2012 (phase **C** - dark grey). Positive observations are denoted by black dots, zero observations (months without reported shark catches) by grey dots.

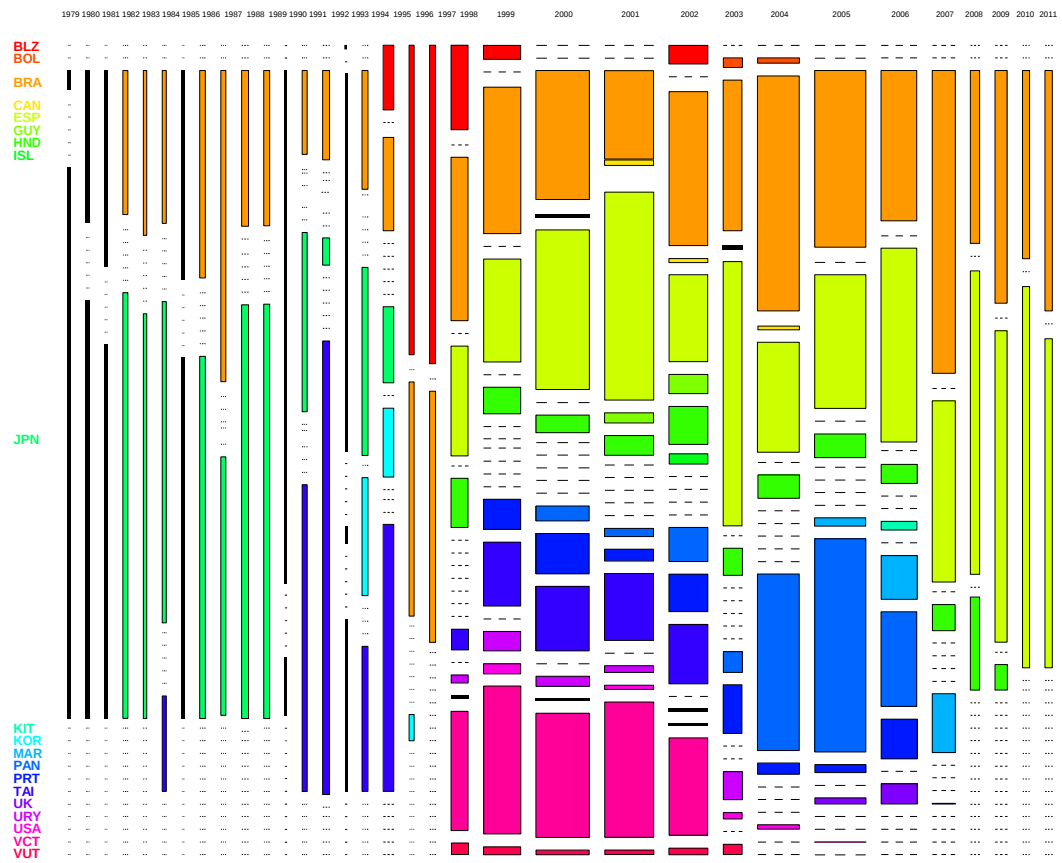
Figure 3. Mosaic plot of fishing effort (number of sets) by year from fleets that reported shark catches for Brazilian institutions between 1979 and 1997.

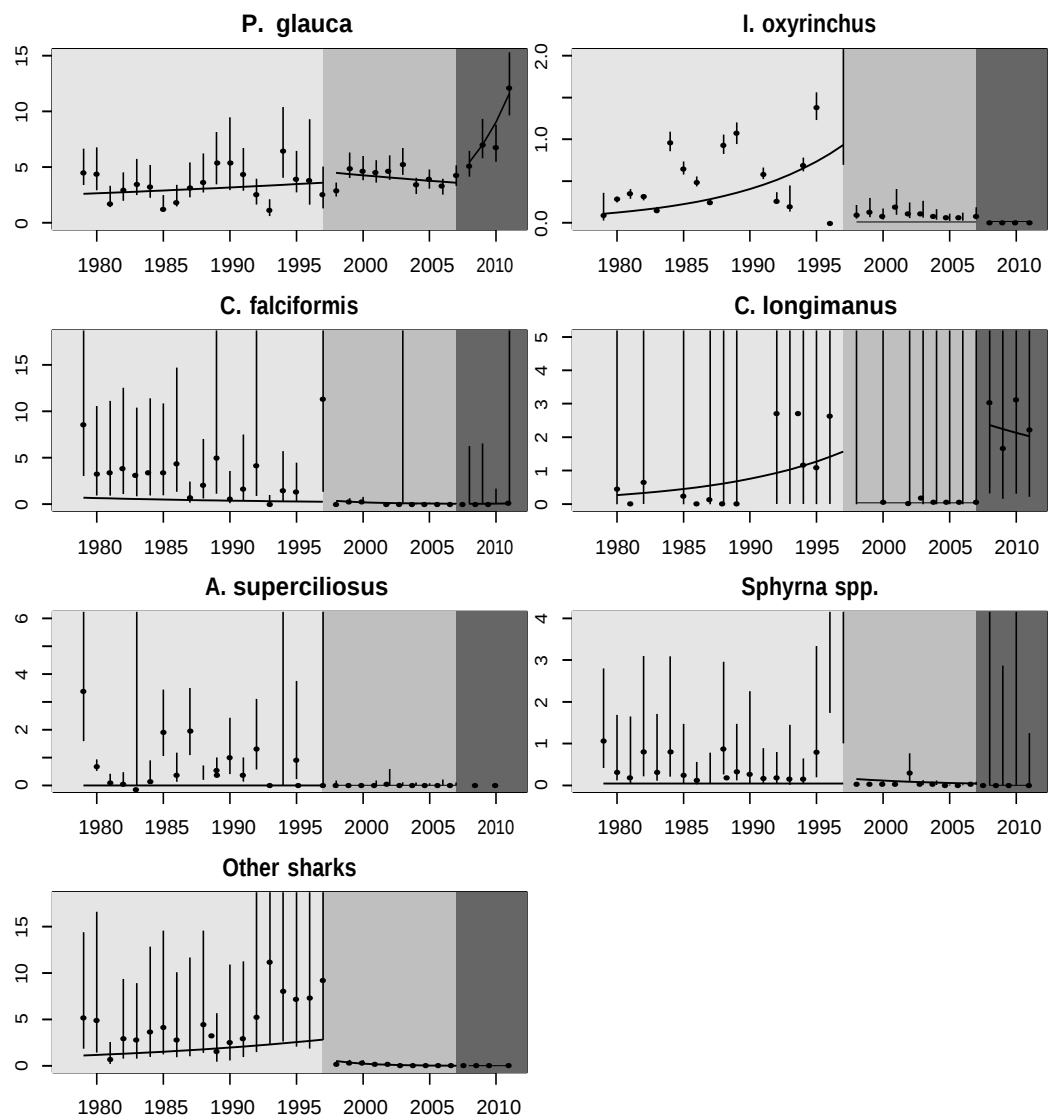
Figure 4. Trends in standardized catch rates (shark catches per 1000 hooks, estimated from generalized linear mixed models using a zero truncated negative binomial distribution) in the three identified fishing phases, plotted for each species. Solid lines show the overall trends (using year as continuous variable) and dots indicate the individual year estimates (using year as factor, ± 95 CI). Shaded zones indicate the different phases; 1979–1997 (**A** - light grey), 1998–2007 (**B** - medium grey), 2008–2011 (**C** - dark grey).





Fishing fleets in the Western South Atlantic





Appendix S1. Phases of exploitation in the southwestern Atlantic Ocean (Briefly history of longline fisheries in the SAO).

Brazil and Uruguay were the first coastal nations to develop pelagic longline fisheries in the South Atlantic Ocean. These countries began their fishing activities when distant fleets (from Japan, China, Korea, Spain, Taiwan) began, in the 1950s, to explore distant waters in order to meet the commercial demand for tunas. While in Brazil fishing began around 1955, with boats headquartered in Santos (Southeast Brazil) and Recife (Northeast Brazil), in Uruguay the activity started in 1969 from Montevideo and La Paloma (Hazin et al. 2008; Marin et al. 1998).

Brazil has been recording fishing data since 1979 (Table S1, Fig. S1). Until the mid-1990s, Brazil targeted tunas with Japanese-style multifilament deep longlines, with J hooks baited with small pelagic fishes like the Brazilian sardine (*Sardinella brasiliensis*) (Arfelli, 1996, Hazin et al. 1998). Then, gear configurations changed considerably to target swordfish. Surface monofilament longlines lines were included, and fishing operations were carried out at night using light sticks and squids as bait (Broadhurst and Hazin, 2001; Tolotti et al. 2013). Effort increased notably (Table S1, Fig. S1).

Similarly, the Uruguayan fleet went through significant changes in gear configuration in response to market demands (initially targeting tunas, then swordfishes and finally sharks). In our database, we had data on six Uruguayan chartered vessels that reported 729 sets between 1998 and 2003 (Table S1, Fig. S1).

South Africa and Namibia are the latest coastal nations to engage in longline fishing in the SAO. These countries were influenced by distant fleets

which fished in the region since the 1960s but it was from the end of the 1990s that national vessels from South Africa and Namibia began to operate in the SAO (Kroese, 2000). Data from these fleets are limited but preliminary assessments revealed that the predominant species are the same as from the southwestern Atlantic (Penney and Griffiths, 1999; Kroese, 2000; Hazin et al. 2008).

As Brazil, Uruguay and South Africa occupy a wide area of the SAO, these countries historically (since the 1950s) have opened their ports to distant fleets (Domingo et al. 2014). These nations have adopted a leasing strategy with foreign fishing companies (mainly from Asia and the European Union) in order to develop their own fleets and skilled labor. The leasing is characterized by an agreement between a foreign fishing company (lessor) and a national fishing company (lessee). The fishing company is responsible for technological and operational resources (Ministry of Fishing and Aquaculture - Brazil).

Stimulated by this leasing strategy, numerous foreign fleets were attracted to the SAO waters over the last decades (data from Western SAO are available in Table S1 and Fig. S1). Traditionally longline fleets that operate very far from home need to rely on local ports to process landings. The traditional fleets like Japan, Spain and Chinese-Taipei have changed their landing ports in the SAO over time due to changes in fishing grounds, commercial demands for target species and also to international management measures dictated mainly by ICCAT. For example, in three years (1969–1971), China-Taipei used 15 different ports in the Atlantic: St. Maarten, Abidjan, Cape Town, Las Palmas, São Vicente, Monrovia, Tema, Dakar,

Santa Cruz, Walvis Bay, Buenos Aires, Recife, Montevideo, Paranaguá and Tenerife (Yang & Yuan, 1973; Domingo et al. 2014).

Japan was the first foreign fleet operating in the SAO, initially targeting albacore (*Thunnus alalunga*) and yellowfin tuna (*T. albacares*), before shifting effort toward bigeye tuna (*T. obesus*). It began fishing in the area in the 1950s and continued until the beginning of the 1990s (Tables S1 and S2, Fig. S1), but data collection on fishing operations started only in 1978. Approximately 15–30% of the boats report catching mainly species of commercial interest (Hazin et al, 2008). Yet, the number of sets reported by Japan to the Brazilian government considerably exceeds those reported by the Brazilian fleet, except for 1987, 1990, 1992 and 1993 (Tables S1 and S2). Gear configuration of this fleet is described in Hazin et al. (1990).

China-Taipei (which in our database indicates vessels from the Republic of China and also Taiwan) came to the SAO in 1984 to catch mainly tunas and developed fisheries only during this year when they migrated to other regions of the Atlantic. This fleet returned to the SAO between the years 1991 and 1995 and particularly for these years, a large number of sets were reported by approximately 30 vessels (Tables S1 and S2). Gear configurations of this fleet are similar to the ones used by Japan (Tolotti et al. 2013). However, after the 1990s, China-Taipei started using different gear configurations to target sharks and swordfishes (Hazin et al. 2008). Additionally, China-Taipei purchased additional licenses to fish in the SAO on behalf of small countries of Central America such as Belize and Saint Vincent and the Grenadines (ICCAT, 2013).

Belize and Korea started fishing in the SAO in 1993 and 1994 respectively. Korea fished in the area until 1996. Belize continued fishing until 2002 (Tables S1 and S2). Since 1995, the effort carried out by these fleets in relation to other fleets has been considerably higher (Table S1 and Fig. S1).

Spain began fishing in the SAO in the early 1990s (prior to 1988, vessels were restricted to the North Atlantic Ocean, Hazin et al., 2008). However only in 1997 did data start to be reported (for Brazilian institutions). The Spanish fleet traditionally targeted swordfish but recently extended their target to blue and mako sharks (Mejuto et al., 1985, Castro et al., 2000; Hazin et al. 2008). Just like Japan, the Spanish fleet had a profound influence on longlining development in the SAO, being responsible for almost all fish (and other oceanic species) entering Europe. This fleet, together with China-Taipei and Japan, have been using flags of other countries, particularly those not having the fishing capacity to meet the quotas allocated by ICCAT. Although fishing for a shorter period of time than Brazil and Japan, Spain ranked second for the amount of sets deployed in the area according to our database (Table S1 and Fig. S1). Likewise, considerable levels of unreporting should be considered (Hazin et al. 2008).

Between the end of the 1990s and mid-2000s, at least 15 different fleets simultaneously exploited the SAO over the Brazilian leasing strategy using different fishing strategies (Tables S1 and S2, Fig. S1). According to Tolotti et al. (2013) gear configuration of these fleets can be divided into two major fishing strategies: Japanese and Spanish. Japan mainly targets tuna species, while Spain fishes for swordfishes and sharks. While Panama and Portugal are fleets that had been fishing using the Japanese strategy, Morocco,

Honduras and the United Kingdom had fished in the Spanish way. All these fleets reported a significant number of fishing sets, particularly between the years 1998 and 2007.

In the last 10 to 15 years sharks started to receive international attention from regional and international fisheries organizations (FAO [Food and Agriculture Organization], ICCAT, etc.), and international conservation bodies and agreements (i.e IUCN and CITES). In 1999, FAO introduced the IPOA-Sharks (International Plan of Action for the Conservation and Management of Sharks) aiming to set effective actions with regard to sustainable use of sharks caught directly or indirectly by global fisheries (FAO, 2000). The document contextualized within that plan provides a series of actions required for effective conservation group, supplemented by the management techniques for elasmobranch fisheries (APEC, Musick & Bonfil, 2005).

From the late 1990s ICCAT had been encouraging signatory members to report sharks catches in particular, and in 2002 the organization approved an amendment banning the practice of finning in the Atlantic. Since 2008 (based on data-poor assessments), ICCAT has recommended no retention, catch or commercialization of all species of the *Sphyrna* genera (2008), *Alopias superciliosus*, *C. longimanus* and *C. falciformis* (2009, 2010 and 2011 respectively). Yet, numerous fleets continue to not report catches and to fish illegally (Oceana, 2009, 2011, 2013). Figure S3 shows a dynamic representation of the evolution of the international fishery in the area (<http://baseline.stanford.edu/evolutionFishery.gif>)

Table S1. Fleet composition, gear type (DL – deep longline, SL – surface longline), target species, number and origin of the boats, amount sets reported and main literature sources.

Fleet	Gear	Target	Year	Boats	Boat orig.	Sets reported	Literature source of operational information
Saint Kit and Nevis	DL/SL	Swordfish, tunas, sharks	2006	1	China	97	http://www.fao.org/docrep/005/y4260e/y4260e0e.htm , Hazin et al. 2008; Carvalho et al. 2011
Iceland	SL	Sharks, tunas, swordfish	2002	1	Iceland	121	http://firms.fao.org/firms/resource/7/en ; Hazin et al. 2008
Bolivia	DL/SL	Swordfish, sharks, tunas	2003-2004	2	Spain, Japan	122	Hazin et al. 2008
Canada	SL	Sharks, tunas, swordfish	2000-2004	1	Canada	246	http://firms.fao.org/firms/resource/7/en ; Hazin et al. 2008
United Kingdom	SL	Swordfish, sharks, tunas	2005-2006	3	Unite Kingdom	322	Oceana, 2009
St. Vincent and Grenadines	DL/SL	Tuna, sharks	1998-2005	38	Spain, China	341	http://www.fao.org/docrep/005/y4260e/y4260e0e.htm , Hazin et al. 2008; Carvalho et al. 2011
Eq. Guinea	DL	Tuna, sharks	2001-2002	4		383	http://firms.fao.org/firms/resource/7/en ; Hazin et al. 2008; Carvalho et al. 2011
United States	SL	Sharks, tunas, swordfish	1998-2004	1	USA	436	http://www.nmfs.noaa.gov/sfa/hms/related_topics/bycatch/documents/fse_is_final_section_6.pdf
Korea	DL	Tunas, swordfish, sharks	1994-1996	2	Korea	473	Anderson et al. 2011
Uruguay	DL/SL	Tunas, swordfish, sharks	1998-2003	6	Spain, Japan, China	729	Domingo et al. 2008, Anderson et al. 2011; Carvalho et al. 2011
Morocco	SL	Swordfish, sharks, tunas	2005-2007	3	Spain	1030	http://firms.fao.org/firms/resource/7/en ; Hazin et al. 2008
	DL/SL	Tunas, sharks	1995-2002	7	China	2119	http://firms.fao.org/firms/resource/7/en ; Carvalho et al. 2011
Portugal	SL	Swordfish, sharks	1999-2006	5	Portugal	2618	Dos Santos et al. 2002; Oceana, 2009; Hazin et al. 2008; Carvalho et al. 2011
Honduras	DL/SL	Sharks, tunas, swordfish	1998-2009	5	Spain, Japan, China	3184	Clarke et al.2008; Tolotti et al. 2013; Hazin et al. 2008; Carvalho et al. 2011
China-Taipei	DL/SL	Tuna, sharks	1984-2002	34	China	6438	Yeh et al. 2011, Anderson et al. 2011; Liu et al. 2005; Xiao-jie & Zhan-qing, 2000; Carvalho et al. 2011
Japan	DL	Tunas, swordfish, sharks	1979-1995	28	Japan	6663	Anderson et al. 2011; Hazin et al. 2008; Carvalho et al. 2011
Vanuatu	SL	Swordfish, sharks, tunas	1998-2003	1		7563	http://firms.fao.org/firms/resource/7/en ; Hazin et al. 2008
Panama	SL	Tunas, sharks, swordfish	2000-2006	34	Spain, China	7579	Clarke et al. 2008; Tolotti et al. 2013; Hazin et al. 2008; Carvalho et al. 2011
Spain	SL	Swordfish, sharks, tunas	1998-2011	34	Spain	20660	Anderson et al. 2011; Tolotti et al. 2013; Hazin et al. 2008; Carvalho et al. 2011
Brazil	DL/SL	Tunas, sharks, swordfish	1979-2011	152	Brazil, Japan, Spain, China	25368	Amorim et al. 1998; Hazin et al. 2008; Anderson et al. 2011; Carvalho et al. 2011

Table S2. Number of sets reported by each fleet.

FLAG	Year																																			
	Total	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011		
BRA	16	155	160	226	179	192	173	361	506	334	281	260	135	193	204	210	304	364	473	861	1677	2115	1334	1846	885	3019	2784	1656	2135	487	873	408	562	25368		
BRA.BLZ																																			2119	
BRA.BOL															2			209	481	599	445	160		223		55	67							122		
BRA.CAN																									48	82	48	19	49						246	
BRA.ESP																					579	1175	2618	3139	1039	1550	1414	2110	2138	1278	855	1170	826	769	20660	
BRA.GUY																									153	230									383	
BRA.HND																					260	301	291	295	455	159	300	372	208	184	263	96			3184	
BRA.ISL																									121										121	
BRA.JPN	465	427	305	668	440	404	298	631	421	887	751	29	289	58	9	333	248																		6663	
BRA.KIT																														97					97	
BRA.KOR																																				473
BRA.MAR																	209	223	41									131	483	416					1030	
BRA.PAN																																				7579
BRA.PRT																																				2618
BRA.TAI						120								495	980	93	257	867			109	730	1061	1011	715										6438	
BRA.UK																													95	221	6					322
BRA.URY																					43	223	169	101	27	166									729	
BRA.USA																					16	116	28	65	20	39	57								341	
BRA.VCT																					628	1688	2035	2045	1165			2							7563	
BRA.VUT																					61	92	75	68	79	61									436	

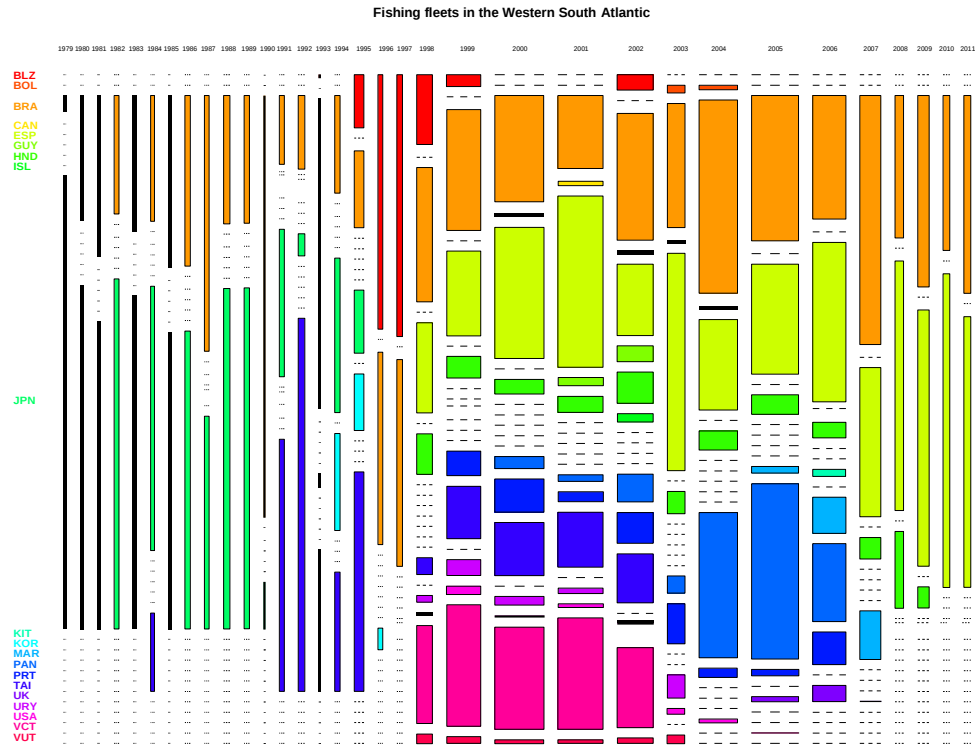


Figure S1. Fishing fleets and sample size of long-line sets reported to Brazilian agencies in the SAO between the year of 1979 and 2011.

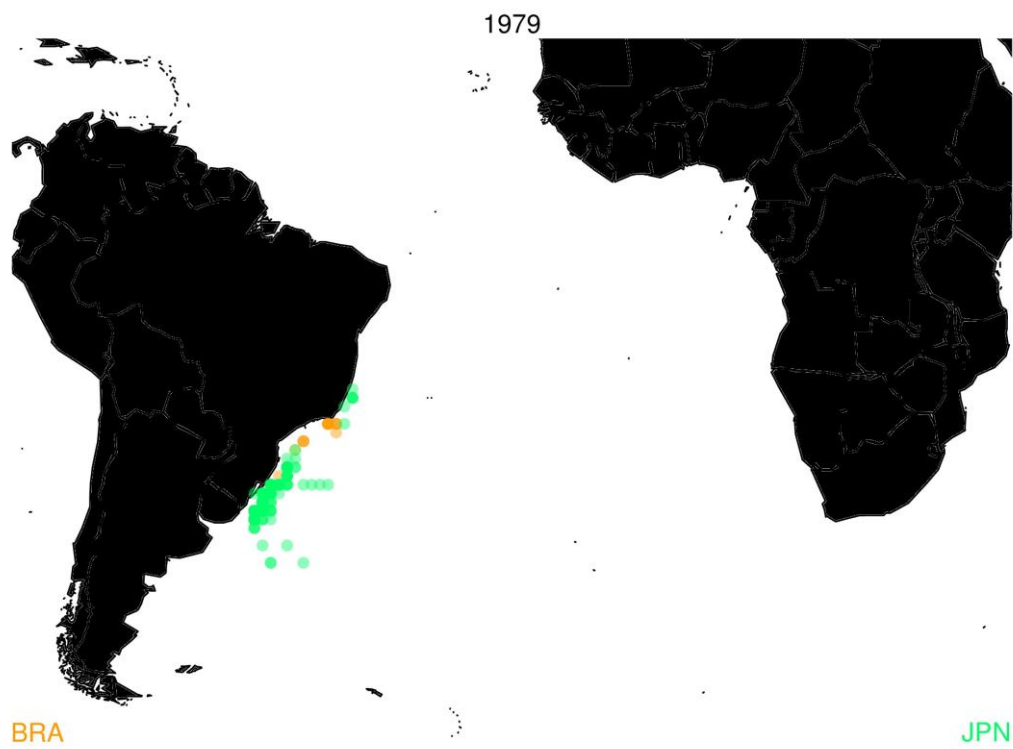


Figure S2. Dynamic representation of the evolution of the international fishery in the area [the animated plot(<http://baseline.stanford.edu/evolutionFishery.gif>)

Appendix S2. Phases of exploitation in the southwestern Atlantic Ocean (Empirical and statistical rationale)

Between 1978 and the the mid-1990s the southwestern Atlantic Ocean has been exploited by a small number of fleets (Brazil, Japan, China-Taipei and Korea) using monofilament longlines (Japanese style) using J hooks and small fishes as bait in order to catch tuna species (Fig. S1). Because all these fleets use similar fishing gears and were based at Santos city (Brazil) we consider this period as a distinct fishing phase (phase A, Fig. 1). During this phase, reporting, particularly with respect to zero, was different than the other periods. Table S2 shows the total number of sets per fleet by year, Tables S3 and S4 indicate the number of zeros and NAs (respectively) reported for blue sharks during the same period. NAs were not present in the database, and zeros were very common. This pattern is even more evident when looking at less abundant species such as silky sharks (Tables S5 and S6) and oceanic white-tip sharks (Tables S7 and S8).

After the mid-90s, fisheries management regimes have undergone profound changes in order to capture species other than tunas (swordfish and sharks). Because some fleets continued to catch tunas while other fleets targeted swordfishes and sharks, a mixed fishing phase is considered (Fig. S1). Methods for reporting data also changed. This aspect became evident when looking at the data from the Brazilian fleet. NAs began to appear more often (Tables S3 to S8).

Finally, a third stage (phase C, Fig. 1) is identified, from 2008 to 2011 (Fig. S1). Although some fleets (particularly Spain and Brazil) continued fishing, the amount of sets reported was considerably lower (Fig. S1, Table

S2). Zeros were no longer reported in the database while NAs appeared more frequently, and this was also true for data of non-Brazilian fishing fleets (Tables S3 to S8). This third stage is characterized by the implementation of an onboard observer program in foreign boats that landed in Brazil and by the displacement of the major landing port from southern to northern Brazil. Also, it was during this period that both ICCAT and the Brazilian Ministry of the Environment (MMA) adopted more restrictive management measures such as banning finning and retention for some species.

Table S3. Number of zero catches reported by the fleets (blue sharks).

Flag	Year																																	
	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	
BRA	9	80	93	126	93	112	72	193	365	247	208	225	135	175	180	171	270	351	449	234	715	1175	360	328	228	1060	630	416	654	258	1	0	0	
BRA.BLZ	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	151	329	302	220	102	0	1	0	0	0	0	0	0	0	0	0	
BRA.BOL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	44	56	0	0	0	0	0	0	0	
BRA.CAN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	34	72	37	7	24	0	0	0	0	0	0	0	0	
BRA.ESP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	85	991	2299	735	581	745	472	466	449	230	91	1	0	0	
BRA.GUY	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	153	230	0	0	0	0	0	0	0	0	0	0	
BRA.HND	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	17	221	276	152	57	42	43	32	5	48	57	1	0	0	
BRA.ISL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	121	0	0	0	0	0	0	0	0	0	0	
BRA.JPN	293	183	103	269	195	146	122	279	250	445	550	23	164	44	8	282	169	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.KIT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	53	0	0	0	0	0	
BRA.KOR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	102	120	15	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.MAR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	36	187	105	0	0	0	0	
BRA.PAN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	218	93	67	12	624	1439	285	0	0	0	0	0	0
BRA.PRT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	268	650	18	250	286	13	5	40	0	0	0	0	0	0
BRA.TAI	0	0	0	0	84	0	0	0	0	0	0	0	0	0	0	0	0	0	0	62	441	479	846	715	0	0	0	0	0	0	0	0	0	0
BRA.UK	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	38	78	1	0	0	0	0	
BRA.URY	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	27	146	153	87	36	0	0	0	0	0	0	0	0	0	0
BRA.USA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	11	114	12	45	20	39	50	0	0	0	0	0	0	0	0
BRA.VCT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	251	1176	1279	2045	1165	0	0	0	0	0	0	0	0	0	0
BRA.VUT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table S4. Number of missing values identified in the logbooks reported by the fleets (blue sharks).

Flag	Year																																	
	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	
BRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	49	57	60	17	10	16	15	3	2	1	2	148	126	125		
BRA.BLZ	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.BOL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.CAN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.ESP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	3	3	1	1	3	90	87	20	0	
BRA.GUY	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.HND	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	33	5	0	0	0	0	0	0	0	0	0	20	0	0	0	
BRA.ISL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.JPN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.KIT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.KOR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.MAR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.PAN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	
BRA.PRT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.TAI	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.UK	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.URY	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.USA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.VCT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.VUT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	1	0	0	2	0	0	0	0	0	0	0	0	0	0	0

Table S5. Number of zero catches reported by the fleets (silky sharks).

Flag	Year																																			
	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011			
BRA	9	118	90	167	153	153	111	313	484	325	273	252	135	188	204	202	304	364	467	288	719	1184	671	1415	510	2515	2306	1339	2024	465	0	0	0			
BRA.BLZ	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	191	458	585	445	156	0	0	223	0	0	0	0	0	0	0	0	0	0		
BRA.BOL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	55	67	0	0	0	0	0	0	0	0	0		
BRA.CAN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	47	82	48	19	49	0	0	0	0	0	0	0	0	0		
BRA.ESP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	544	1106	2455	3139	976	1534	1397	2020	2065	1257	745	0	0	0	0	0		
BRA.GUY	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	153	230	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.HND	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	34	253	291	295	417	156	300	369	207	166	230	0	0	0	0	0	0	0	
BRA.ISL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	121	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.JPN	390	3461	281	589	400	365	249	527	395	807	741	29	278	58	326	234	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.KIT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	96	0	0	0	0	0	0		
BRA.KOR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	192	154	38	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	
BRA.MAR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	131	458	400	0	0	0	0	0	0	0	0	
BRA.PAN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	227	126	409	122	2262	3327	1042	0	0	0	0	0	0	0	0	0	
BRA.PRT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	288	658	178	448	286	143	125	378	0	0	0	0	0	0	0	0	0	
BRA.TAI	0	0	0	0	120	0	0	0	0	0	0	476	950	89	255	827	0	109	695	999	1011	715	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.UK	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	95	221	6	0	0	0	0	0	0	0	
BRA.URY	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	37	206	140	101	27	115	0	0	0	0	0	0	0	0	0	0	0	0
BRA.USA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	16	116	25	65	20	30	5	20	30	5	2	0	0	0	0	0	0	0	0	0	0	
BRA.VCT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	607	1670	2001	2045	1165	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.VUT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Table S6. Number of missing values identified in the logbooks reported by the fleets (silky sharks).

[illegible]

Table S7. Number of zero catches reported by the fleets (white-tip sharks).

Flag	Year																																			
	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011			
BRA	16	152	147	222	179	192	168	359	501	333	280	260	135	189	191	273	355	473	297	725	1201	671	1416	1150	2520	2286	1431	2086	487	0	0	0	0			
BRA.BLZ	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	209	481	599	445	160	0	0	233	0	0	0	0	0	0	0	0	0	0		
BRA.BOL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	55	67	0	0	0	0	0	0	0	0	0		
BRA.CAN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	48	82	48	19	49	0	0	0	0	0	0	0	0		
BRA.ESP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	579	1175	2601	3139	1031	1536	1382	1952	1939	1143	563	0	0	0	0	0		
BRA.GUY	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	153	230	0	0	0	0	0	0	0	0	0	0	0	0		
BRA.HND	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	43	273	291	295	455	159	300	327	208	166	200	0	0	0	0	0		
BRA.ISL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	121	0	0	0	0	0	0	0	0	0	0	0		
BRA.JPN	465	423	305	667	440	404	298	628	417	887	751	29	289	58	0	333	248	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
BRA.KIT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	79	0	0	0	0	0	0	0		
BRA.KOR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	223	41	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0		
BRA.MAR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	120	412	379	0	0	0	0	0	0		
BRA.PAN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	246	126	409	122	2197	3186	960	0	0	0	0	0	0	0	0	
BRA.PRT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	344	658	178	448	286	143	124	404	0	0	0	0	0	0	0	0	
BRA.TAI	0	0	0	0	120	0	0	0	0	0	0	0	495	980	93	257	867	0	0	109	730	1061	1011	715	0	0	0	0	0	0	0	0	0	0	0	
BRA.UAK	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	95	199	6	0	0	0	0	0		
BRA.UKY	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	41	223	168	101	27	115	0	0	0	0	0	0	0	0	0	0	0	
BRA.USA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	16	116	28	65	0	0	116	28	65	20	39	57	0	0	0	0	0	0	0	0	0	
BRA.VCT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	628	1688	2035	2045	1165	0	0	2	0	0	0	0	0	0	0	0	0	0
BRA.VUT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	29	0	0	0	0	0	0	0	0	0	0	0	

Table S8. Number of missing values identified in the logbooks reported by the fleets (white-tip sharks)

Flag	Year																																
	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011
BRA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	564	952	914	663	428	374	499	421	212	49	0	865	407	552
BRA.BLZ	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.BOL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.CAN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.ESP	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8	14	0	48	0	0	45	1107	747	699
BRA.GUY	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.HND	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	217	28	0	0	0	0	0	0	0	0	0	87	0	0
BRA.ISL	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.JPN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.KIT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.KOR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.MAR	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.PAN	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.PRT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.TAI	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.UK	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.URY	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	51	0	0	0	0	0	0	0	0
BRA.USA	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.VCT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
BRA.VUT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	61	92	75	68	50	61	0	0	0	0	0	0	0	0

Appendix S3. Sensitivity analysis (Phases and fleets)

We tested the robustness of our results by changing the timing of the identified fishing phases. We moved the cutoff (by 2 years in both directions) and also modeled data without phases (Fig. S4, BC is the base case). Changing the cutoff of the phase B to 2 years ahead resulted in creating a phase C of just one year (2011). Hence, in this case we chose to also include 2011 in phase B. In all situations we kept the same modelling framework, using the same error structure and link.

We also tested the effect of including fleets within the set of explanatory variables. For testing the inclusion of fleets, we used generalized additive mixed models (GAMMs). Excluding the variable *flag* from the analyses, more information could be included in the modeling process (all data from fleets that caught species for less than two years, previously excluded). GAMMs are often used in catch rates standardization because of their flexibility in allowing complex functional forms between predictors and the response. In this way we also tested the effect of the variables *flag* and *boat* as random terms, however we only could extract results for blue sharks (the most abundant species). Model results for the timing of the phases are available in Table S9 and Fig. S4.

Phase A

We identified through the base case that the species had a significant increase over the years in catch rates (0.019, Tables 3 and S9, Figs. 4 and S4). Changing the final year of this phase for 2006 the year effect (coefficients) on catch rates were similar to the base case (0.02, Table S9 and

Fig. S4). We observed the opposite trend when we moved the cutoff ahead. The year effect doubled in relation to the base case (Table S9 and Fig. S4). In all scenarios, year effect estimates were significantly positive in catch rates, an effect that was observed for all species (Table S9). The silky shark presented results that were opposite to those observed for the blue. While the base case and the third scenario showed similar results, when the second scenario was used the observed decrease was relatively higher (Table S9). This was the only species that presented negative rates (and statistically significant results) in all situations.

For mako and thresher sharks, catch rates decreased in relation to the base case in both scenarios. The variation observed between the base case and the second scenario (-2 years) was lower than the +2 years scenario (Table S9). As observed for blue sharks, in all situations an increase in catch rates was observed, but these increases were statistically significant only for makos (Table S9).

For the white-tip and the "other sharks" group we observed exactly the same year effect between the base case and the third scenario (+2 years). They were both positive and statistically significant (Table S9). For both species this increase was considerably smaller (and not significant) when the second scenario was applied (-2 years). For the group consisting of the carcharhinids species, catch rates were very similar in all scenarios (Table S9).

Contrasting results for this phase were observed for hammerhead sharks. While in the base case, we observed a non-significant increase in the catch rates, in both alternative scenarios estimates were considerably lower.

In the second scenario (-2 years) we detected a significant decrease in catch rates, the opposite from both of the other scenarios.

Phase B

All species (except threshers and white-tip) declined significantly at this phase (Fig. 4). By using the second scenario blue sharks showed a negative trend in catch rate (Table S9, Fig. S4). Whereas when the cutoff was moved forward by two years we observed a significant increase in the catch rates (Table S9, Fig. S4).

For mako, silky and white-tip sharks, "other sharks" and "grey sharks", we observed significant declines in catch rates in both scenarios (except white-tip, Table S9, Fig. S4). Only thresher sharks showed increasing catch rates in all scenarios but only in the third scenario was this increase significant (Table S9).

The most contrasting results for Phase B were observed for hammerhead sharks. As in phase A, when the second scenario was applied, trends were reversed in relation to the base case, at this time changing from significantly negative to non-significantly positive (Table S9, Fig. S4).

Phase C

Despite minor variations, the results do not change considerably for blue and silky sharks and the "other sharks" group. For makos and the group composed of *Carcharhinus* species, when the second scenario was applied the trends that were not significant in the base case became significant. Interestingly for the group of grey sharks, the year effect was considerably lower than that in the base case (Table S9, Fig. S4). This same trend was

observed for hammerhead sharks, which in turn are the only species with negative trends at this stage in both scenarios tested. *C. longimanus* is the only species that showed a reversal in the trends observed in the original phase. However, in both situations year estimates were not statistically significant.

No phases

Aside from blue, thresher and white-tip sharks, all species showed significant declines in catch rates when modeled without phases. Regarding the species that showed an increasing trend in phase C, the estimates were not statistically significant only for thresher sharks.

Flag

Catches recorded in our database were recorded by fishing fleets fishing with varying fishing strategies across fleets and even within the same fleet across vessels and years (Figs. S5-S8). This process may have been captured by the inclusion of both vessel and fleet as nested random effects or by the inclusion of other covariates identifying the set fishing strategy (e.g. hook type, fishing gear, or target fish). Unfortunately our dataset does not have enough information to properly characterize the fishing process at the set level, and an attempt to include vessels and boats as random effect worked only for the most abundant species, the blue sharks. However, modelling blue sharks with such a random structure of the data was illustrative to explain the nature of the fluctuations detected in the original simpler models with flag as a fixed effect (Fig. S9). The fluctuations observed in the original

GLMs were due to the different number and kind of fleets fishing each year. It is in fact evident that the trajectories of catch rates gradually become smoother, going from a simpler model including flag as fixed effect to flag and boat included as nested random effects (Fig. S9).

Table S9. Results and (scenarios) from the sensitivity analysis (timing of phases). Values in bold had statistical significance at $p = 0.05$.

Species	Scenarios	Year effect		
		A	B	C
Blue sharks	Base case	0.0197	.0.0358	0.2510
	2 years (-)	0.0205	.0.0288	0.2320
	2 years (+)	0.0412	0.0101	!
	No phases		0.0261	
Mako sharks	Base case	0.1144	.0.0820	0.0922
	2 years (-)	0.0935	.0.0999	0.1100
	2 years (+)	0.0707	.0.1014	!
	No phases		.0.0498	
Silky sharks	Base case	.0.0523	.0.2518	0.5810
	2 years (-)	.0.0801	.0.1594	0.3560
	2 years (+)	.0.0454	.0.1659	!
	No phases		.0.0798	
White.tip sharks	Base case	0.1050	!0.0984	!0.0509
	2 years (-)	0.0862	!0.0815	0.0233
	2 years (+)	0.1050	!0.0056	!
	No phases		0.0028	
Thresher sharks	Base case	!0.0562	0.0201	0.5960
	2 years (-)	!0.0413	0.0169	0.1030
	2 years (+)	!0.0208	0.0649	!
	No phases		0.0321	
Hammerhead sharks	Base case	0.0426	.0.1068	.0.5104
	2 years (-)	.0.0840	0.0233	!0.0746
	2 years (+)	0.0168	.0.2313	!
	No phases		.0.05937	
Other sharks	Base case	0.0139	.0.3760	0.2264
	2 years (-)	0.0016	.0.4003	0.1050
	2 years (+)	0.0139	.0.2120	!
	No phases		.0.1234	
Grey sharks	Base case	0.0718	.0.3816	0.3650
	2 years (-)	0.0647	.0.4181	0.0903
	2 years (+)	0.0650	.0.2123	!
	No phases		.0.1280	

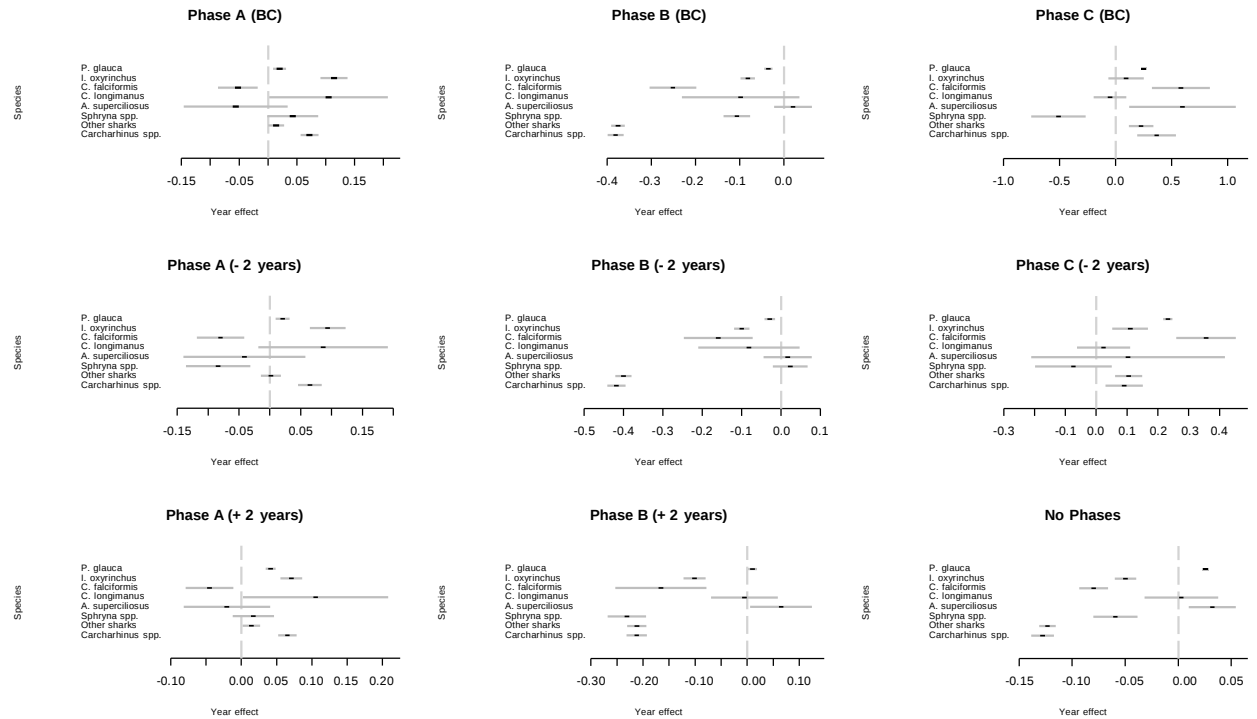


Figure S4. Results (year effect) of the sensitivity analysis performed by changing the cutoff levels of phases. BC is base case.

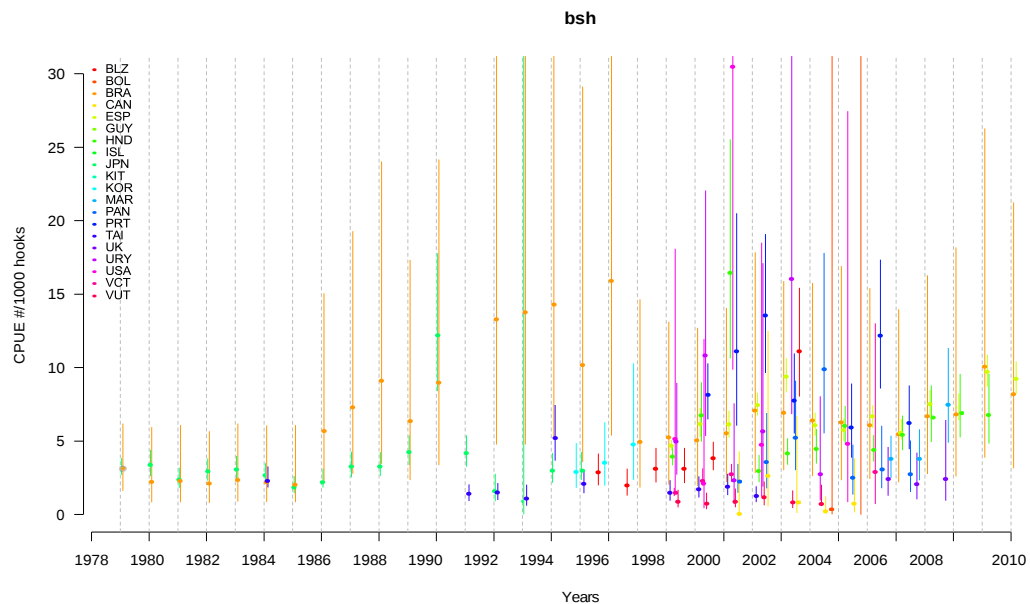


Figure S5. Standardized catch rates by fleets, for blue sharks reported to Brazilian agencies in the SAO by using GLMs and a zero-truncated negative binomial distribution for the positive catches.

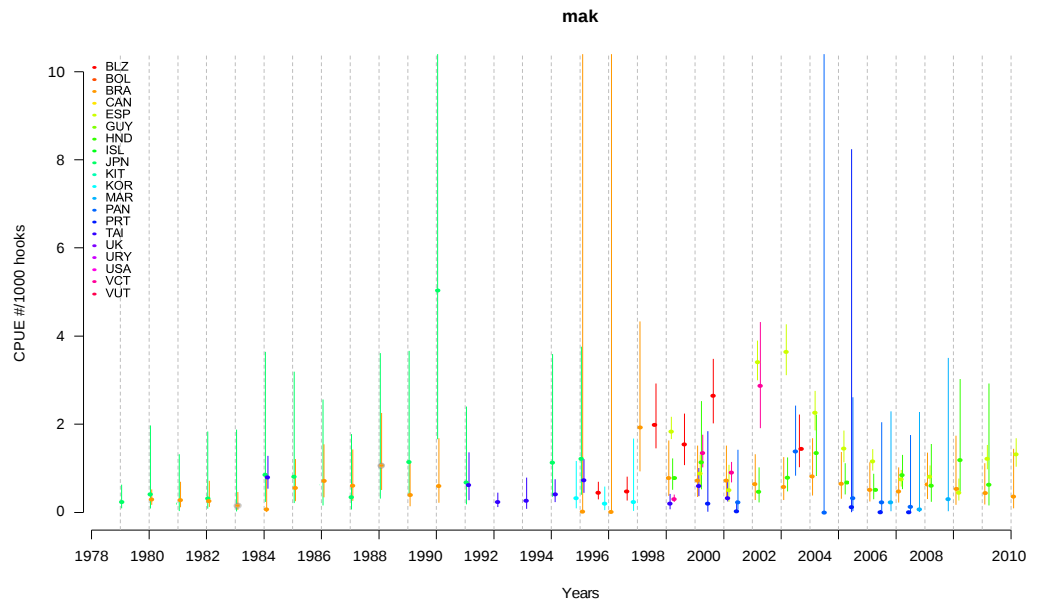


Figure S6. Standardized catch rates by fleets, for shortfin mako sharks reported to Brazilian agencies in the SAO by using GLMs and a zero-truncated negative binomial distribution for the positive catches.

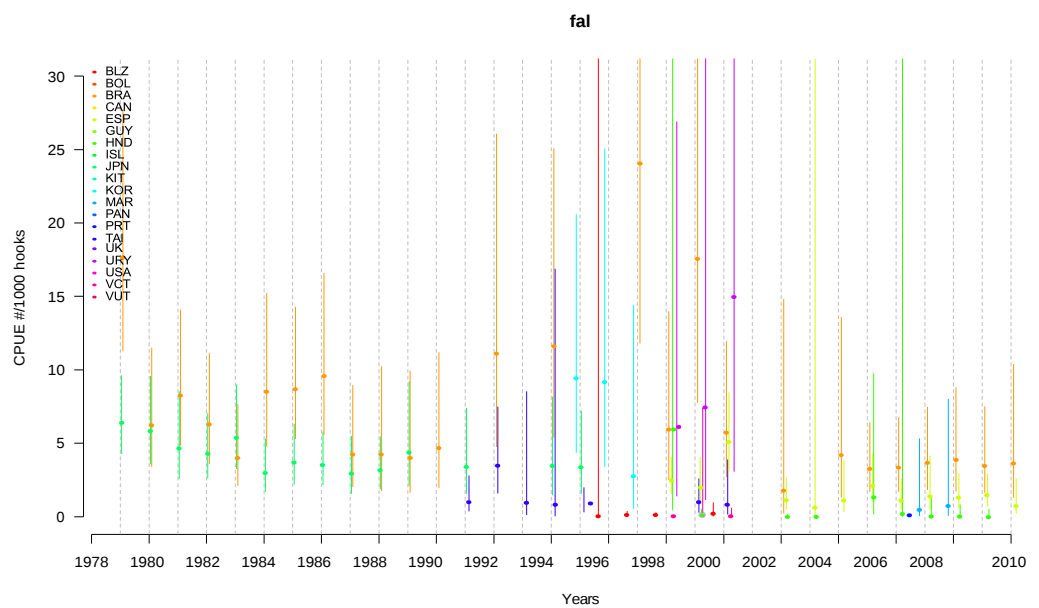


Figure S7. Standardized catch rates by fleets, for silky sharks reported to Brazilian agencies in the SAO by using GLMs and a zero-truncated negative binomial distribution for the positive catches.

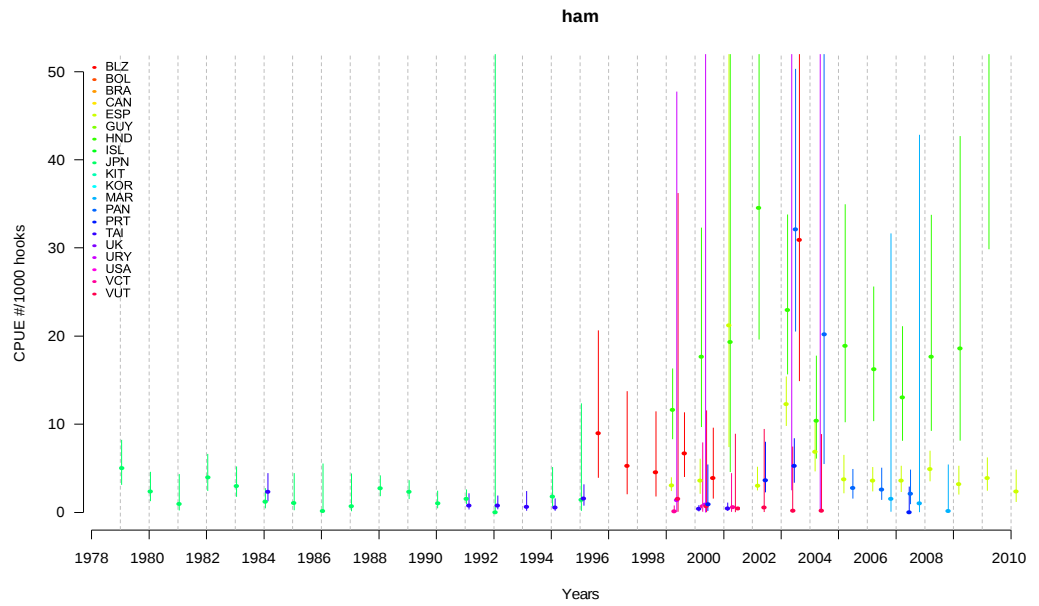


Figure S8. Standardized catch rates by fleets, for Sphyrnidae sharks reported to Brazilian agencies in the SAO by using GLMs and a zero-truncated negative binomial distribution for the positive catches.

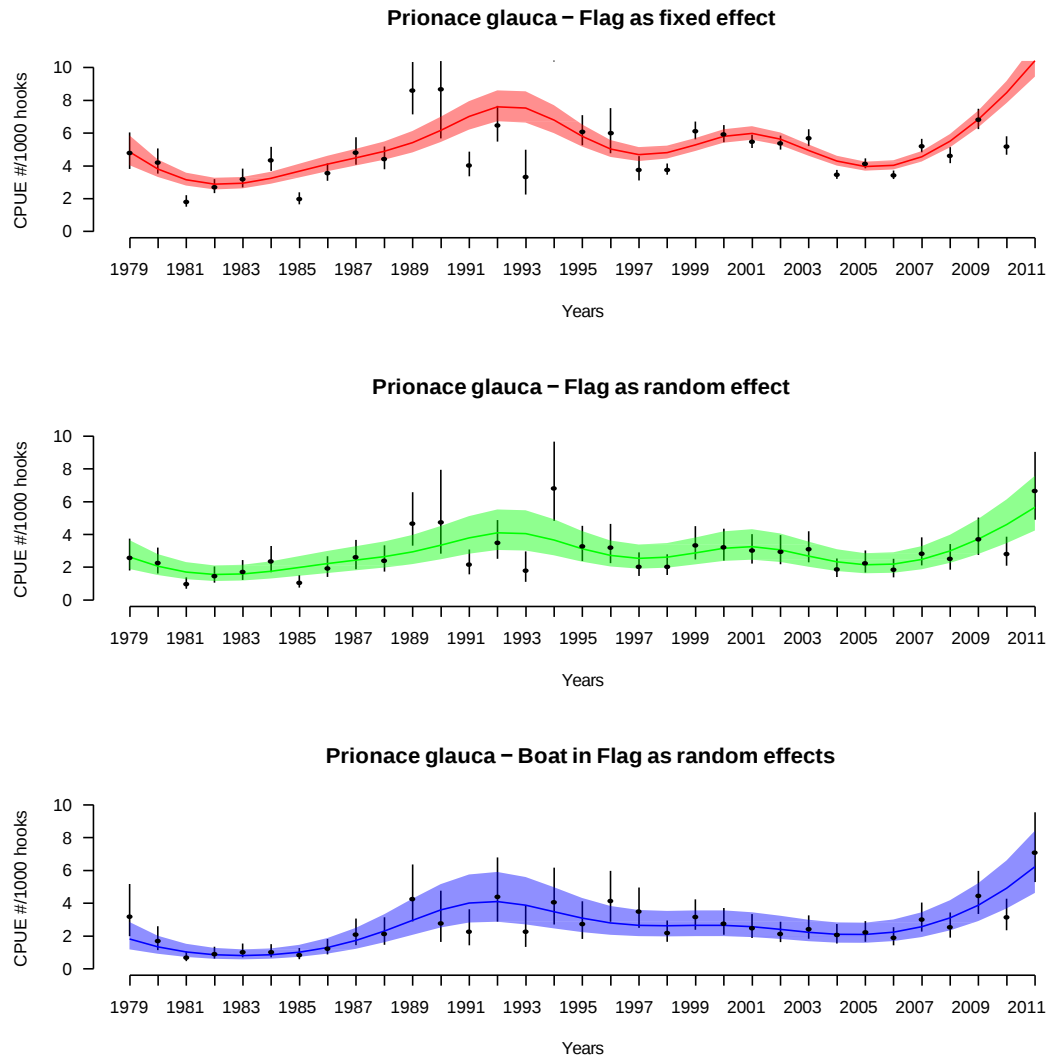


Figure S9. Sensitivity analysis of our models to the covariate flag (fleet) both as random and fixed effect. Catch rates were standardized by using generalized additive mixed models GAMM, with a zero truncated negative binomial distribution for the positive catches of blue sharks reported to Brazilian agencies in the SAO between 1979 to 2011.

Appendix S4. Data sources used in this study.

Source of data	Description
BNDA (Banco Nacional de Dados da Pesca de Atuns e Afins, Ministry of fishing and aquaculture - MPA/Brazil)	Approximately 74% of the database (66,786 sets). A compilation of <i>logbooks</i> reported by Brazilian and foreign boats operating under the Brazilian leasing strategy to the Ministry of Fisheries and Aquaculture of Brazil between the years 1978 and 2011.
IP - SP (Instituto de Pesca, Sao-Paulo, Brazil)	Approximately 7% of the database (6,188 sets). Includes information about fishing trips reported by Brazilian and foreign boats operating under the Brazilian leasing strategy based in Southeastern Brazil during the period from 1997 to 2008.
PROBORDO (National Observers Program, Ministry of fishing and aquaculture - MPA/Brazil)	Approximately 19% of the database (16,284 sets). Information reported by on-board observers, only for foreign boats operating under the Brazilian leasing strategy based in Northeast Brazil during the period from 2005 to 2011.

Appendix S5. Percentage rate of change estimated between phases.

Specie	AB	BC
<i>P. glauca</i>	26.73	> 100
<i>I. oxyrinchus</i>	< -100	16.97
<i>C. falciformis</i>	<- 100	21.93
<i>C. longimanus</i>	<- 100	>100
<i>A. superciliosus</i>	34.10	>100
<i>Sphyrna spp.</i>	-48.38	-98.31
<i>Other sharks</i>	<- 100	70.99
<i>Grey sharks</i>	<- 100	56.98

4. Vulnerability of South Atlantic large pelagic sharks.

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Abstract

Sharks play an important role in structuring marine ecosystems and are characterized by low productivity rates that implies in small litters, slow growth rates, late sexual maturity and long inter-birth interval. Modern fisheries have explored a wide range of sharks during the last decades both caught as target (for the global shark fin trade) and another uses (meat). Longline fishing is one of the major industrial modes of exploitation in open ocean areas and linked to declines of several species of oceanic sharks. We present the first demographic study performed with multiple species (blue, shortfin mako, silky, night, oceanic white-tip and *Sphyrna* spp.) in order to infer about conservation status and vulnerability of the shark assemblage in the South Atlantic. During the years 2005 to 2011 onboard observers reported catches of 24,772 sharks, distributed in 4 families and 11 species. Blue shark was the most reported species followed (in order) by hammerhead, shortfin mako, white-tip, silky, night, porbeagle and bigeye thresher sharks. A subsample of aged individuals was taken from researchers that had studied the main species in the area in order to estimate the age structure for the onboard observers sample.

Demographic analysis were performed using different methodologies in order to identify broader populations parameters by using the identified age structure and life history parameters extracted from the literature. Highest natural mortality estimates were found for blue sharks and carcharhinids in general, while the lowest for lamniformes (shortfin mako and bigeye threshers) and hammerhead sharks. Blue and night sharks were the sharks that showed the highest fisheries mortality necessary to drive a species to extinction, while shortfin makos and bigeye thresher showed the lowest. Total mortality estimated from Chapman and Robson catch curves range from $Z = 0.984$ for blue sharks to $Z = 0.346$ for white-tip. All shark species analyzed in this study showed negative intrinsic rates of increase (r) and population growth (λ) above 1, a clearly indication that populations are decreasing in size due to the fishing pressure and absence of fishing control. Lamniformes sharks, hammerheads and 2 carcharhinids were predicted to be the most vulnerable species in the area.

Keywords: Sharks, South Atlantic, Longline fisheries, Vulnerability

Introduction

Many species of sharks play an important role in structuring marine ecosystems and likely have influenced the diversification and distribution of prey and competitor species over evolutionary time (Holden 1974; Lindberg & Pyenson, 2006; Ferretti et al. 2010). Sharks are characterized by low productivity rates that imply small litters, slow growth rates, late sexual maturity and long inter-birth interval (Cortes et al. 2002; Garcia et al. 2008).

These features lead to extremely low resilience for most species, giving to populations a limited capacity to recovery in events of overfishing and even in moderate increases of natural mortality rates, for example habitat losses/degradation (Smith et al., 1998; Stobutzki et al, 2002, Ferretti et al. 2013). According to the International Union for Conservation of Nature (IUCN), 29% of non-data deficient sharks are currently threatened with extinction (Dulvy et al. 2014).

A wide range of sharks has been exploited by modern fisheries during the last decades both caught as target, for the global shark fin trade and another uses (Clarke et al. 2015; Worm et al. 2013), and as bycatch, in longline pelagic fisheries for tunas and billfishes for example (Amorim et al.1998; Fredou et al. 2015). Overfishing associated with bycatch (or incidental catch) represents the major threat to sharks globally and affected populations may quickly decline (Casey & Myers, 1998; Baum & Myers, 2004, Ferretti et al. 2013).

There are well-documented cases of shark population collapses due to fisheries such as *Lamna nasus* in the North Atlantic; *Galeorhinus galeus* in California and Australia; *Cetorhinus maximus* in England; *Squalus acanthias* in the North Sea and British Columbia; *Pristis pectinata* in Florida and Louisiana, and large coastal sharks off the west coast of the United States, highlighting *Carcharias taurus* and *Carcharhinus obscurus* (IUCN, 2013). South Atlantic fisheries (particularly those reported for Brazil) also reached critical levels for species such as *Carcharhinus porosus*, *Sphyrna tudes*, *S. tiburo* and *Isogomphodon oxyrinchus* in North Brazi (Lessa et al. 2006ab; Mycock et al. 2006); *Carcharhinus plumbeus*, *C. galapagensis*; *Sphyrna*

lewini; *S. zygaena* in the western and central part of SAO; *Rhinobatos horkelli*, *Squatina guggenheim*, *S. occulta*; *Galeorhinus galeus*, *Mustelus fasciatus*; *M. schmitti* *Carcharias taurus* in the South Brazil (Vooren, 1997; Lessa et al., 1999; ICMBio, 2015).

Longline fishing is one of the major industrial modes of exploitation in open ocean areas and one of the least selective. It basically consists of using a main line where there are regular intervals with outputs of secondary lines employing baited hooks (FAO, 2012). Notice that levels of shark bycatch in these fisheries can be higher than that recorded for targeted species (Gallagher et al. 2014, Oliver et al. 2015). Over the last few decades, population declines of oceanic sharks have largely been attributed to longline fishing in the North Atlantic, Pacific, and Indian oceans (Worm et al. 2013; Dulvy et al. 2014). High seas longlining in the Atlantic Ocean generates about 25% of reported global shark catches, whereas fishing effort has been high in both sides of this ocean (Clarke et al. 2008; Meneses et al. 2000; Hazin et al. 2008). However, most of the information on the effect of fishing on large pelagic sharks comes from the North Atlantic Ocean, while data from the South Atlantic Ocean (SAO) are fragmented and problematic, mainly due to underreport and large scale movements of the species (Hazin et al. 2008). Also, the simultaneous exploitation of the South Atlantic by several coastal and international fishing fleets, under leasing agreements with SAO major nations such as Brazil, Uruguay, South Africa and Namibia likely had hampered stock assessments of most species in the area (Hazin et al. 2008).

Faced with the problem related to absence and poor quality of fishery dependent data, and with the difficulties in performing scientific surveys in

areas so heterogeneous as the open sea, the use of demographic techniques (which depends only on life history information) have been growing among fisheries scientists (Cortes et al. 2002; Simpfendorfer et al. 2006). Life histories are modeled by the energy available in the habitats for allocation of different biological processes and also by the interactions that influence allocation decisions (Garcia et al. 2008). Consequently the extinction risk of a species is strongly correlated to its life-history traits (Hutchings 2002, Frisk et al. 2005, Garcia et al. 2008). Demographic methods allow identifying, for example population rates of increase, replacement potential and at what point of the life cycle the species are more vulnerable to changes in vital rates, such as mortality from fisheries.

Here we present the first demographic study performed with multiple species in order to infer the conservation status and vulnerability of large pelagic sharks caught by longliners in the SAO. This work is timely, as Brazil will re-assess the conservation status of marine fauna in the next few years (MMA, 2014). As there are no evidences of marked populations structure in SAO we assume that our results are indicative of broader populations status, our source of data however, was exclusively Brazilian. We estimate demographic parameters by using different methodologies to assess vulnerability for 5 species of sharks (blue, shortfin mako, silky, night, oceanic-white tip) and one group of species (*Sphyrna* spp.).

Methods

Fishery data

Our database consists of information collected by onboard observers through logbooks from the Brazilian pelagic longline chartered fleet (Spain, Panamá, Honduras, Morocco, Portugal, United Kingdom) based in Northeast Brazil during the period from 2005 to 2011 (Fig. 1). Information retained from logbooks used in this study were: specie (Tables 1 and 2), year of the set in whose individuals were caught (*Y*), month of the set in whose individuals were caught (*m*), geographical coordinates of the set (*lat and lon*), sex and fork length (*FL* in cm) from the individuals. Whenever the *FL* was not available, the *TL* (Total length) and the *ID* (interdorsal length) were converted to *FL* using species-specific conversion equations.

Although the literature reports that at least 18 species of pelagic sharks occur (or have occurred) in the SAO, problems with fishermen and observers incorrectly or inconsistently identifying sharks lead to the combination of some sharks into general groups like “other sharks” and “hammerhead sharks” (Tables 1 and 2). Species that tend to be reliably identified include blue, shortfin mako, silky (some concern about this specie), night, bigeye thresher and oceanic whitetip sharks (Table 2). Histograms by length classes were used to visualize size structure by specie and sex and the sex bias was analyzed through the use of χ^2 -tests.

A subsample of aged individuals was taken from researchers which had been studied age and growth of the main species in the same study area in order to build the age-length keys (Lessa et al., 1999; Santana et al., 2000; Lessa et al., 2004; Santander 2009 and Barreto unpublished). Because

estimates of age and growth of all fish collected from fisheries are impractical age–length keys or *ALKs* are used to assign ages for individuals in whose length was taken (Coggins et al., 2013, Ogle et al. 2015). Length classes for shark sizes were fixed at 10 cm of length interval and contingency tables were build with the frequency (and proportion) of each species in each length class interval with the specific age combinations (Ogle, 2015). To assign the individual ages of the entire sample and summarize the results, we used the method described by Isermann and Knight (2005) and (Ogle, 2015). Age frequency distributions were plotted for each species and year..

Life history

Life-history data were obtained from a detailed literature review, including the most recent regional studies, thesis and grey literature, while prioritizing peer-reviewed publications. We constructed a database including information length, age, growth and reproduction (Table 4). T_{max} was set as the maximum observed age in age and growth studies. Age at first maturity or T_{mat} was derived by using *ALKs* applied to specie-specific lengths at first maturity extracted from the literature. Annual reproductive rate (b) was corrected by embryonic sex ratio and also length of reproductive cycles reported in the literature to identify the average number of female pups per adult female. Maximum size (L_{max}) was defined as the maximum observed length for the species from our sample.

Demography

Natural mortality (M) of the species was estimated using empirical relationships based on life history theory using the methods in Then et al. (2014) and Kenchington, (2014). Methods selected to estimate M (and their details) are available in Table 3. Given that none of the methods have been specifically developed for sharks, we also estimated mortality by using the Brander's method (Brander, 1981). Brander's method is based on the fact that for a population to remain at a constant level and not to decrease or increase in size (i.e. equilibrium), the total rate of mortality of adults must be equal to the net rate of recruitment of mature individuals into the population (Simpfendorfer et al. 2006). Unlike other methods, this method allows estimates of threshold levels of mortality beyond which the population will collapse, considering that the actual number of young of the year (YOY, i.e. newborns produced per year) is known (Simpfendorfer et al. 2006): $Z_i = be^{-(M_i - M_{thr})}$; where: b is the annual reproductive rate corrected by embryonic sex ratio (Table 3), Z_i is the total mortality of young's (assumed to be constant across immature ages) (Simpfendorfer et al. 2006). We then used the mortality rates estimated with this method as a reference scenario for our demographic analysis.

Total mortality (Z) was estimated through the use of catch curves (Ricker, 1975; Ogle, 2015) by the method adapted by Chapman and Robson (Chapman & Robson, 1960; Robson & Chapman, 1961). This method is based on the assumption that catches-at-age on the descending limb of the catch curves follows a geometric probability distribution. This probability distribution can be used to derive a maximum likelihood estimator for the

survival parameter of the distribution (Ogle, 2015). According Dunn et al. (2002) and Smith et al. (2012) this method to estimate Z performs better than other common methods (weighted regression for example) because of its smaller variance (Ogle, 2015). Application of this method requires “re-coding” ages so that the first fully recruited age class of the catch curve descending limb is set to age 0 (Ogle, 2015). Age classes with less than 5 individuals were dropped (Smith et al. 2012; Ogle et al. 2015). The Chapman-Robson equation used to estimate the annual survival rate was: $S = \frac{n}{T - T_{rc}} = \frac{n}{\sum_{i=1}^n (a_i - a_{rc})}$;

where n is the total number of sharks observed on the descending limb of the catch curve, T is the sum of the re-coded ages of fish on the descending limb of the catch curve (i.e., the sum of catch multiplied by re-coded age), and T_{rc} the mean re-coded age of sharks on the descending limb of the catch curve (Miranda & Bettoli 2007, Ogle 2015). Standard errors for S estimates were

extracted following Miranda and Bettoli (2007): $SE_i = \sqrt{S \left(S - \frac{1}{n} \right)}$.

Unbiased estimates of Z plus standard errors were then estimated according Hoenig et al. (1983) and Smith et al. (2012) equations: $Z = -\log S$ ()

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$\frac{(1 - S)(1 - S)}{(1 - S)(1 - S)} ; SE = \frac{1 - S}{\sqrt{1 - S}} \sqrt{c}$; where c is the “usual chi-square goodness-of-fit

test statistic divided by the square root of the df.” Total mortality rates for the species derived from this methodology were used as our second demographic scenario.

Common demographic parameters for species using the two scenarios mentioned above were then estimated through the use of age structured Leslie matrices, assuming density independence (Caswell, 2001; Simpfendorfer et al. 2006). We used a first scenario to infer species demographic parameters

in a hypothetical situation of equilibrium (no fisheries), while the second may reflect the real situation of populations exposed to longline fisheries in our study area (total mortality). Parameters estimated from Leslie matrices were: λ = population finite growth rate, as the dominant vector of the projection matrix; w = stable age-distribution vector, as the dominant right eigenvector; v = age-specific reproductive vector, as the dominant left eigenvector and E_{ij} = the elasticities of matrix elements, being: $E_{ij} = \frac{v_i m_{ij} w_j}{\langle v, w \rangle}$ in which: m_{ij} is the matrix element from row i and column j , v is the reproductive value vector value from row i ; w_j is the stable age distribution vector from column j and $\langle w, v \rangle$ is the scalar product from vectors w and v (Caswell, 2001). Given the differences detected in longevity between species, elasticities were presented using three pre-defined common stages for all species, being: YOY (newborns), immature and adults (matures). The sum of elasticity values in which life stage defines the proportional contribution of m_{ij} to the overall population λ (Caswell, 2001).

In another demographic approach, the intrinsic rate of increase (r_z , also know as rebound potential) was also estimated using a variant of the traditional Euler-Lotka's equation, adapted by Au and Smith, 1997 and Smith et al. 1998. Unlike other r estimators this method incorporates concepts of sustainability (MSY) and population compensation to deal with problems associated to survival schedule and other data limitations (Smith et al. 2008). Also, the method allows the estimation of population growth rates for exploited populations, in particular, populations that are hypothetically exposed to a maximum sustainable level of harvest (assumed to be at about half of the virgin population size) (Smith et al. 2008). Stable-aged condition is assumed

and used as reference point (average situation) and the equilibrium mortality estimated in this study was used to solve r_z (Smith et al. 1998). The equation used was: $\sum_{i=1}^{T_{mat}} l_i e^{-r_z i} m_i = 1$; where: T_{mat} is the age at maturity, w is the maximum reproductive age (interpreted as T_{max}), M is the adult instantaneous natural mortality M , l_x is survival to age x , m_x is fecundity at age x in annual newborn females per adult female (Arr, Table 2), and r is intrinsic rate of increase (Smith et al. 1998). The value of r_z was then determined by removing fishing mortality allowing the population to rebound in an unfished state with juvenile survival remaining at the “enhanced” compensatory level (Smith et al. 2008). More details about r_z estimation are available in Au and Smith 1997, Smith et al. 1998, Smith et al. 2008 and Au et al. 2015.

Ultimately, we estimate the fishing mortality necessary to drive a species to extinction ($F_{extinct}$) following Myers and Mertz (1998) and Garcia et al. (2008). $F_{extinct}$ was calculated iteratively from the following equation: $b = \exp(F_{extinct} T_{mat} (-T_{50} + 1) (1 - \exp(-M + T_{50})))$; where: b , T_{mat} , T_{50} are parameters described in the life history section (Table 2) and M is the natural mortality (Table 3). Survival rates (S) for all mortality rates used in this study were estimated according the formula established by Ricker (1980) where: $S = e^{-M}$.

Results

Species

During the years 2005 to 2011 onboard observers reported catches of 24,772 sharks, distributed in 4 families and 11 species (Table 1, *Sphyrna* sp. it's a complex with at least 3 species: *S. lewini*, *S. zygaena*, *S. mokarran*). Blue shark was the most reported species followed (in order) by hammerhead, shortfin mako, white-tip, silky, night, porbeagle and bigeye thresher sharks (Table 2). Sex bias was identified for the three carcharhinids (silky, white-tip and night) and the shortfin mako sharks, being makos and night sharks with less females than males in general (Table 3). Size structure reported by onboard observers ranges for all species from 50 to 350 cm of FL (Fig. 1 SOM) being the smallest individuals reported for carcharhinid species, particularly *C. falciformis* and *C. longimanus* (Fig. 1 SOM). For the sample of aged individuals (blue, white-tip, mako, silky and night sharks) size structure for the species was similar to those reported by onboard observers, except for blue and night sharks whereas smaller individuals were less frequent in the studies performed by Lessa et al. (2005) for blue sharks and Santana et al. (1999) for night sharks (Fig. 2 SOM). We have identified a considerable variability of ages in relation to the length classes for all species through the ALKs, however, in the case of sharks that reach large sizes and have high longevity it is understood that these variations are acceptable (Figs. 3-7 SOM). Because both samples came from the same source (commercial longline chartered fleet based in Natal, BR), we assume that the age structure identified through the ALKs for the entire sample (Figs. 3-7 SOM) should reflect current size and age patterns of main pelagic shark species caught by

longliners in the study area. In this context we identified that all species with exception of blue and night sharks are caught from age 0 in our study area (Fig. 2). We also detect that adult shortfin mako sharks, particularly females were rare (Fig. 2, Table 3).

Table 4 shows the life history parameters rose from literature and used in this study. Shortfin mako was the only specie in which we identified the existence of more than one study regarding growth aspects while the white-tip shark, was the only species having more than one reproduction study. There are no regional reproduction studies for shortfin mako, bigeye thresher and hammerhead sharks in the SAO. In general, lamniformes (shortfin mako and bigeye thresher) and hammerhead sharks showed lower growth rates (k) and higher age of maturity (T_{mat}) and longevity (T_{max}) than carcharhinids, except *C. falciformis* (Table 5). The annual reproductive rate (b - average number of female pups per adult female) was lower for lamniformes (mako and thresher), white-tip and hammerhead sharks than ... (Table 4).

Regarding the empirical methods used to estimate natural mortality (M), we found that those using only maximum age (T_{max}) as predictor (Fig. 6 SOM) were more homogeneous than the others. They showed similar estimates for all species (Fig. 7 SOM). Conversely, methods using growth coefficient (k) and multiple other predictors showed some variation (Figs. 8-11 SOM). The two-parameter k equation from Jensen (2001) and the k equation in Then et al. (2014) (Table 1 SOM) showed the highest M estimates among k methods ($M = 0.441$ and 0.341 respectively). The “Alverson and Carney (1975) equation” (equation 10 of Zhang and Megrey, 2006) and the “Rikhter and Efanov (1976) (equation 1 of Kenchington, 2014), among the empirical

methods that uses multi life history information, showed highest M estimates, resulting in inconsistent survivorship curves for species (Fig. S10). We consider that methods that used T_{max} for estimating M (Fig. S6) were more reliable in relation to the longevity of species (Table 3, survivorship in Fig. S6). These methods are also recommended in Then et al. (2014) and Kenchington et al. (2014). Thus, we estimated M by using these methods (average, Fig. 7 SOM).

Highest M estimates were identified for blue and all carcharhinid sharks, while the lowest M were estimated for lamnids and hammerhead sharks (Fig. 4). Regarding the Brander's equilibrium mortality (Z_m), estimates were similar to those from M for all species, except for mako and thresher sharks, which showed equilibrium mortality, lower than M , and hammerhead sharks, which showed natural mortality lower than Z_m (Fig. 4). Blue and night sharks were showed the highest $F_{extinct}$ while shortfin makos and bigeye threshers showed the lowest $F_{extinct}$ (Fig. 4). Total mortality estimated from Chapman and Robson catch curves ranged from 0.984 of blue sharks to 0.346 of white-tip sharks (highest and lower, Figs. 3 and 4). We recognize that this methodology was inconsistent for silky and white-tip sharks (Fig. 3), probably due the heterogeneity of the age distribution detected by using ALKs for these species (Fig. 2). Concerning, fishing mortalities (F_m and F_{zm}) that were estimated by simply subtracting M and Z_m from Z respectively (Fig. 4) showed that almost all species (except blue and night sharks) had F estimates close to $F_{extinct}$, with mako sharks having values of Z almost two times bigger the level of mortality it can sustain ($F_{extinct}$ and Z_m).

Demographic analyses (here we will concentrate our results only in the second scenario, using the total mortality Z , giving the first scenario (using Z_m , should not reflect fisheries and population growth are in hypothetical equilibrium) showed negative intrinsic rates of population increase (r) for all species (Table 5). Similarly, population growth rates were lower than 1 for all species suggesting that all these species are exposed to level of exploitation higher than what they can support. Expected number of replacements (R_0) was lower for mako and silky sharks than..... These species also showed the highest values of mean age of parents of offspring of a cohort (u_1) and generation time (T , time necessary to increase R_0), (Table 5). Elasticities showed that changes in population growth for all species might be strongly associated with changes in vital rates of the immature part of the stocks, particularly young and sub-adults (E_2 , Fig. 5). Finally, our last demographic approach (rebound potential), pointed out that the most vulnerable sharks analyzed were in order: shortfin mako, hammerhead, bigeye thresher, silky and night sharks (Fig. 6).

Discussion

All shark species analyzed in this study showed negative intrinsic rates of increase (r) and population growth (λ) above 1, a clearly indication that populations are decreasing due to the fishing pressure and absence of fishing control (Fig. 5). Even the blue shark, a species considered one of the most resilient among sharks (Aires da Silva & Gallucci, 2006), r was Furthermore, elasticity analysis for all species showed a strong influence of juvenile survival on population growth rates, and juvenile stages were the

portion of the population most exploited by fishing in the SAO (Figs. 2 and 4). Shortfin mako, bigeye thresher and the hammerhead sharks were the most vulnerable to changes in vital rates. These species showed more sensitive life history parameters, particularly those associated with life span and rebound potential (Table 3). These species were recently protected by ICCAT (Tollotti et al. 2015) and were ranked as vulnerable in the western SAO (ICMBio, 2014). The only exception was shortfin makos that were assessed as near threatened and have currently no fisheries restrictions from ICCAT.

The different methods used for estimating natural mortality indicated that some methodology might not be appropriate for long-lived species such as sharks (particularly those using k or multi-parameters to estimate M). Empirical methods that use only t_{max} for estimating M were more plausible. Mortality estimates from these methods were close to those estimated in a hypothetical scenario of equilibrium, indicating as expected for low-productive species such as sharks, considerable sensitivity to any change in their vital rates (also confirmed by the proximity of the estimates with the $F_{extinct}$). When total mortality (Z) was used to estimate population parameters all species most likely find themselves in conditions of overfishing. We recognize however that the chosen method for estimating Z was particularly flawed for two species: the silky and the oceanic white-tip sharks. These findings suggest that since they are oceanic-coastal species, exploitation affects all stages of their life cycle (Fig. 2).

The blue shark is one of the few species of pelagic sharks caught in the SAO for which there is a considerable amount of information (Carvalho et al. 2014). Stock assessment models fitted to blue sharks caught by the Brazilian

longline fisheries concluded that the blue shark population in SAO was above stock biomass at which maximum sustainable yield is achieved (Carvalho et al. 2014). In contrast, recent ICCAT evaluation (ICCAT 2015) showed that the species might be moderately overexploited, confirming results presented in this study (Table 5). In a demographic analysis conducted by Aires da Silva and Gallucci (2006) a positive population growth was observed. However, authors conclude that the status blue sharks in the NAO is ambiguous at a time when catch appears to be on the rise and new trade and exploitation patterns are emerging. The elasticity found for the species in the NAO by these authors was similar of our results. Longline fisheries in the western and central part of the Atlantic currently directly target this species and the Brazilian government has plans to expand these activities in the coming years (MPA, weblink).

Shortfin makos and bigeye thresher sharks were recently predicted to be the most vulnerable oceanic sharks in the Atlantic (Cortes et al. 2010). Lamnid sharks show special reproductive strategies, such as extremely late maturity (the largest of oceanic sharks) and very few pups per female (lower among oceanic sharks). While bigeye thresher sharks are often discarded by fishing fleets (ref), shortfin makos have high market value for fins and meat and thus are directly targeted from some fleets in the Atlantic. In north pacific and north Atlantic, demographic analyses showed that shortfin makos were declining under current fishing conditions (Takeuchi et al. 2005; Tsai et al. 2014). CPUE analyses also showed that the species has experienced significant decreases in abundance in other areas as in the NAO (- 35%

between 1992 and 2005; Baum & Blanchard 2010) and SAO (more than 100 % of decline between 1979 and 1998; Barreto *et al.* in press).

Many species of the genus *Carcharhinus* are often combined into generic groups because of problems in species identification. This practice hampers the identification of stock assessments and conservation status of species. Nonetheless, our results are generally supported by independent assessment approaches, such as demographic and productivity/susceptibility analysis (PSA), which have indicated that some Carcharhinidae species are at risk of extinction in the SAO, specifically *Carcharhinus signatus*, *C. galapagensis*, *C. falciformis*, and *C. longimanus* (Santana *et al.* 2011; Luiz *et al.* 2011; Cortes *et al.* 2010). Barreto *et al.* (in press) also indicated that this group is declining considerably in abundance.

Hammerhead sharks also have identification problems worldwide. However, life history parameters used in this study were exclusively from *S. lewini* which is the most frequently caught species in southern Brazil (around 80%; Kotas *et al.* 2011). This group of sharks have showed remarkable declines in many other ocean sectors and are among the most threatened group of sharks according IUCN and CITES. Baum and Blanchard (2010) showed a decline of more than 75% in the NAO. While Barreto *et al.* (in press) report declines of more than 95% in the SAO.

Catch rate analyses suggest that most large pelagic species are currently depleted in the SAO, and these results are supported by the present study. These declines coincided with significant fishing effort expansion, and apparent directed fishing for sharks by some fleets (Barreto, unpub.). We conclude that the SAO has experienced significant levels of depletion during

the mid-1990s/mid-2000s “gold rush” on sharks and other pelagic species. Furthermore, while countries such as Brazil, Uruguay and South Africa have been creating favorable conditions for many fishing fleets to expand in the area, proper monitoring of these fleets has been inconsistent and there is a general lack of catch and life history data. In this respect it is significant that some shark species of South America (such as *Carcharhinus plumbeus*, *C. porosus*, *C. galapagensis*, *Sphyrna tudes*, *S. tiburo*, *S. lewini*, *S. media*, *S. tudes*, *S. zygaena*, *Isogomphodon oxyrhynchus*, *Galeorhinus galeus*, *Mustelus fasciatus*) are close to extinction in Brazilian waters (ICMBio, 2014). In accordance with Smith et al. (2008), we suggest that information such as innate plasticity of growth and regeneration rates, extent of geographic range, abundance, extent of stock mixing, vulnerability to fishing on pupping, nursery and feeding grounds should be improved.

Table 1. Species and conservation status according the International Union for Conservation of Nature (IUCN, global and for Brazil, ICMBio, 2015).

Family	Species	Common name	IUCN (Global)	IUCN (Brazil)
Lamnidae	<i>Isurus oxyrinchus</i>	Shortfin mako	VU	NT
	<i>Isurus paucus</i>	Longfin mako	VU	DD
	<i>Lamna nasus</i>	Porbeagle shark	VU	DD
Alopiidae	<i>Alopias superciliosus</i>	Bigeye thresher shark	VU	VU
	<i>Alopias vulpinus</i>	Common thresher shark	VU	VU
Pseudocarchariidae	<i>Pseudocarcharias kamohar</i>	Crocodile shark	NT	DD
Sphyrnidae	<i>Sphyrna lewini</i>	Scalloped hammerhead	EN	CR
	<i>Sphyrna zygaena</i>	Smooth hammerhead	VU	CR
	<i>Sphyrna mokarran</i>	Great hammerhead	EN	EN
Carcharhinidae	<i>Prionace glauca</i>	Blue shark	NT	NT
	<i>Carcharhinus falciformis</i>	Silky shark	NT	NT
	<i>Carcharhinus longimanus</i>	Oceanic whitetip shark	VU	VU
	<i>Carcharhinus signatus</i>	Night shark	VU	VU
	<i>Galeocerdo cuvier</i>	Tiger shark	NT	NT
	<i>Carcharhinus spp.</i>	Grey sharks	-	-
Not identified	Other sharks	Mislabeled sharks	-	-

Table 2. Main species and number of individuals reported by onboard observers from Brazilian longline chartered fleet between 2005 and 2011.

Family	Specie	Code (ICCAT)	2005	2006	2007	2008	2009	2010	2011	NA	Total
Alopiidae	<i>A. vulpinus</i>	ALV	-	-	6	-	-	-	2	-	8
	<i>A. superciliosus</i>	BTH	-	-	36	8	-	13	5	-	62
Carcharhinidae	<i>P. glauca</i>	BSH	321	350	5408	1151	109	1871	10145	222	19577
	<i>C. signatus</i>	CCS	-	-	31	-	-	1	23	225	280
	<i>C. falciformis</i>	FAL	-	-	157	47	-	26	85	106	421
	<i>C. longimanus</i>	OCS	17	29	405	250	8	92	117	110	1028
Lamnidae	<i>I. paucus</i>	LMA	-	-	1	-	-	3	4	-	8
	<i>I. oxyrinchus</i>	SMA	22	27	444	134	130	206	314	48	1325
	<i>L. nasus</i>	POR	-	-	3	1	1	3	102	-	110
Sphyrnidae	<i>Sphyrna spp.</i>	SPX	111	165	721	75	4	146	472	-	1694
Other sharks		OTHRSHARKS	-	10	74	12	-	7	156	-	259

Table 3. Sex bias results of the χ^2 -tests. Values in bold were significant differently from 1:1 using a confidence level of 95% ($\alpha = 0.05$).

Specie	Prop. (F to M)	X2	p-value	Sample estimates (p)
<i>A. superciliosus</i>	1.4	1.02	0.3123	0.58
<i>P. glauca</i>	0.98	0.38	0.5338	0.49
<i>C. signatus</i>	0.63	11.29	0.000809	0.39
<i>C. falciformis</i>	1.46	9.95	0.001607	0.59
<i>C. longimanus</i>	1.16	4.53	0.03326	0.53
<i>I. oxyrinchus</i>	0.77	14.2	0.000164	0.43
<i>L. nasus</i>	0.88	0.24	0.6205	0.47
<i>Sphyrna spp.</i>	0.89	2.9	0.08834	0.47

Table 4. Updated life history (and sources) used in this study from main species of sharks caught by longliners in western and central South Atlantic.

Species	Reference sources	Linf*	k	t0	Lmat*	Lmax*	Tmat	Tmax	litter	Rep. cycle	b
<i>Alopias superciliosus</i>	Mancini et al., 2004	269.84	0.06	-5.04	183.49	271.49	12	19	3	2	0.75
<i>Carcharhinus falciformis</i>	Satander et al., 2011; Lana et al., 2012	232.28	0.09	-3.47	167.5	242.2	12	21	16	2	4
<i>Carcharhinus longimanus</i>	Lessa et al., 1999ab; Tambourgui et al., 2013	237.73	0.09	-3.39	147.52	207.72	6	13	6	2	1.5
<i>Carcharhinus signatus</i>	Santana et al., 2004; Hazin et al. 2000	224.60	0.11	-2.71	170.65	201.36	10	15	11	1	5.5
<i>Prionace glauca</i>	Lessa et al., 2004	293.63	0.15	-1.01	188.14	258.69	6	12	30	1	15
<i>Isurus oxyrinchus</i>	Barreto unpub.; Natanson et al., 2006	407.66	0.04	-7.01	275	296	20	23	9	2	2.25
<i>Isurus oxyrinchus</i>	Dono et al., 2013; Natanson et al., 2006	416.00	0.03	-6.18	275	330	18	28	9	2	2.25
<i>Sphyrna lewini</i> **	Kotas et al., 2010; Hazin et al. 2001	230.69	0.05	-	156.77	246.86	15	31.5	14	1	7.15

* TL converted to FL using specie-specific equations (Kohler et al., 1996); ** VBGF two parameter estimates

Table 5. Demographic parameters (Z)

Parameters	Species				
	Blue	Mako	Silky	White-tip	Night
<i>I</i>	0.84	0.77	0.84	0.94	0.87
<i>r</i>	-0.18	-0.26	-0.18	-0.07	-0.14
<i>Ro</i>	0.26	0.01	0.07	0.56	0.18
<i>T</i>	7.66	20.37	15.12	8.84	11.95
<i>μ1</i>	7.58	20.16	14.74	8.74	11.86
E1 (YOY)	0.13	0.05	0.06	0.11	0.08
E2 (Youngs)	0.77	0.87	0.77	0.67	0.83
E3 (Adults)	0.10	0.08	0.16	0.22	0.09

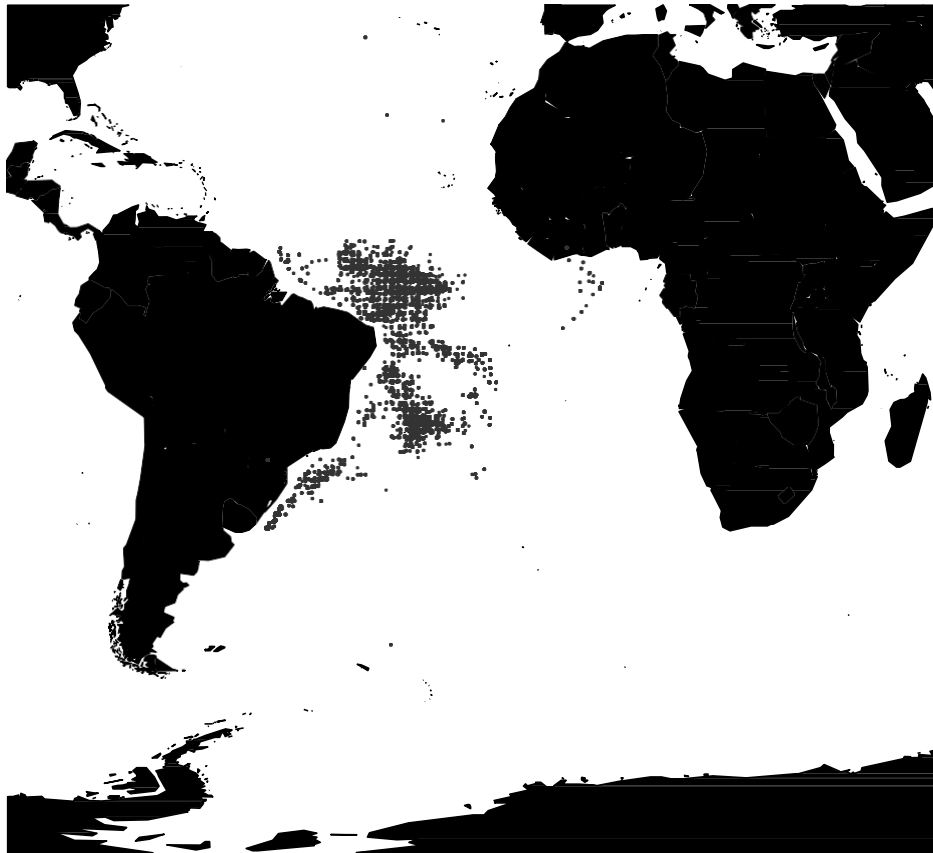


Figure 1. Spatial distribution of longline sets in which sharks were reported by onboard observers from the Brazilian chartered longline fleet between 2005 to 2011.

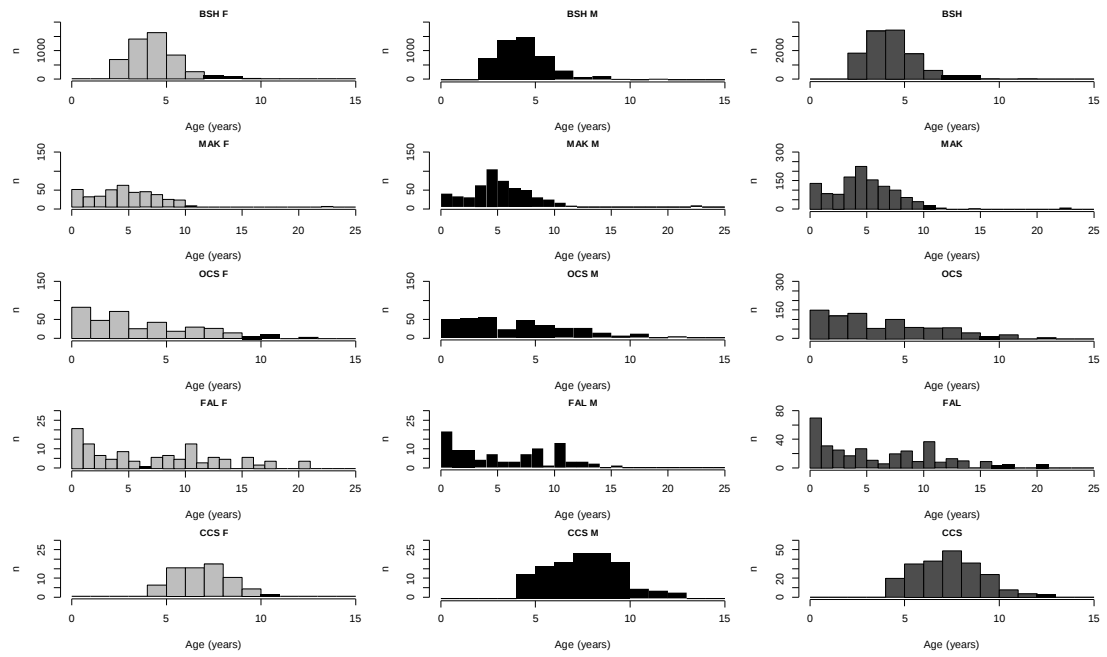


Figure 2. Age distribution (converted to total sample from ALKs) from South Atlantic large pelagic sharks caught by longliners. Unaged individuals smaller or bigger than aged individuals were excluded (ALKs).

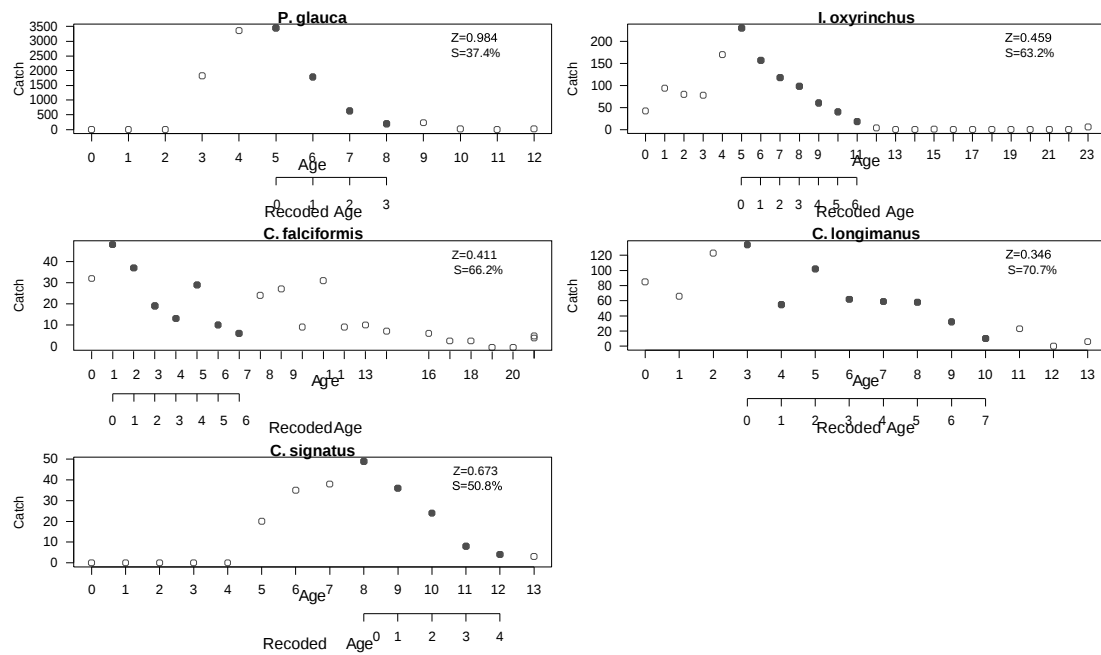


Figure 3. Total mortality (Z) estimated for large pelagic sharks caught by longliners in the South Atlantic Ocean using the Chapman and Robson method (Chapman and Robson, 1960; Robson and Chapman, 1961).

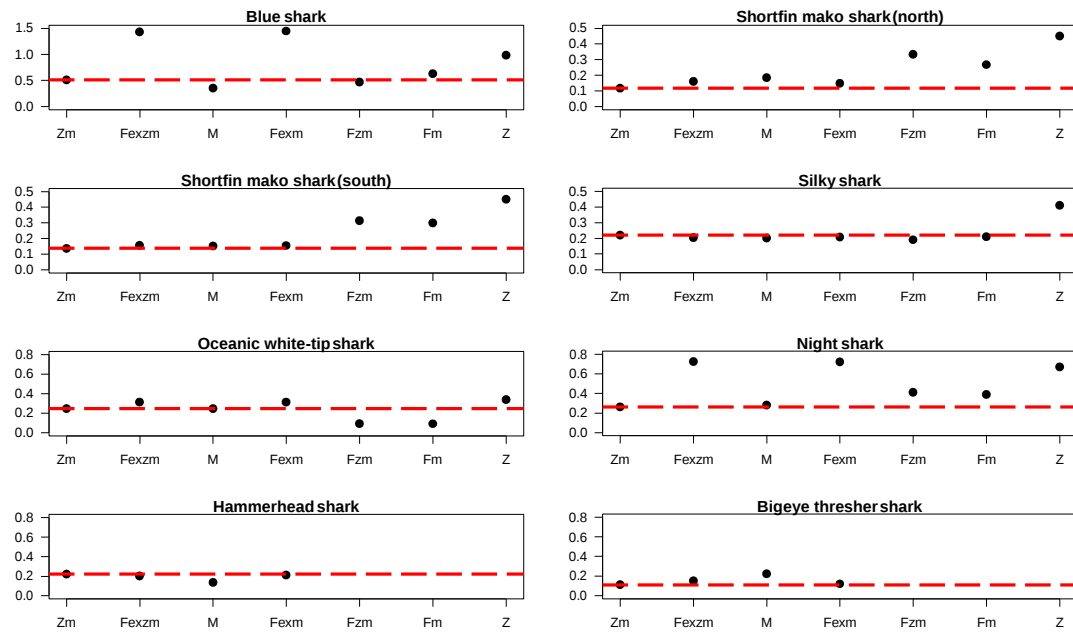


Figure 4. Brander's equilibrium mortality (Z_m); Fishing mortality necessary to drive species to extinction using Z_m as natural mortality (F_{exzm}); Mean natural mortality estimated from methods that uses t_{max} (M); Fishing mortality necessary to drive species to extinction using M as natural mortality (F_{exm}) and total mortality estimated using Chapman and Robson method (Z). Fisheries mortalities (F_{zm} and F_m) were estimated by simply subtracting M from $Z = F_m$ and Z_m from $Z = F_{zm}$. Dashed red line means the mortality level beyond populations will collapse.

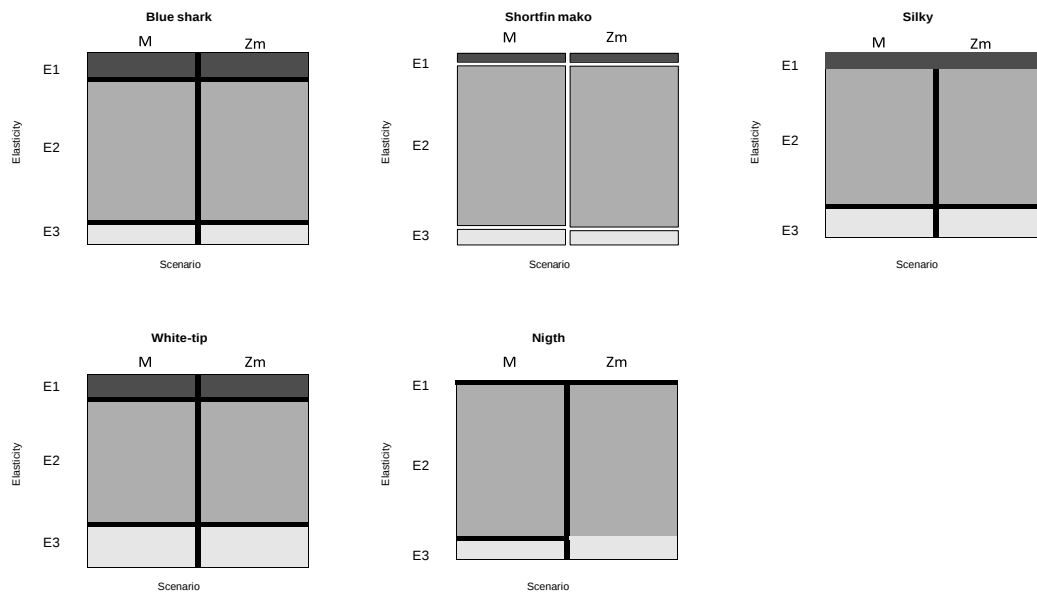


Figure 5. Proportional changes (elasticity) in vital rates on shark populations growth rates (E1 = YOY, E2 = young's and E3 = adults) using demographic scenarios 1 and 2 (M and Zm respectively).

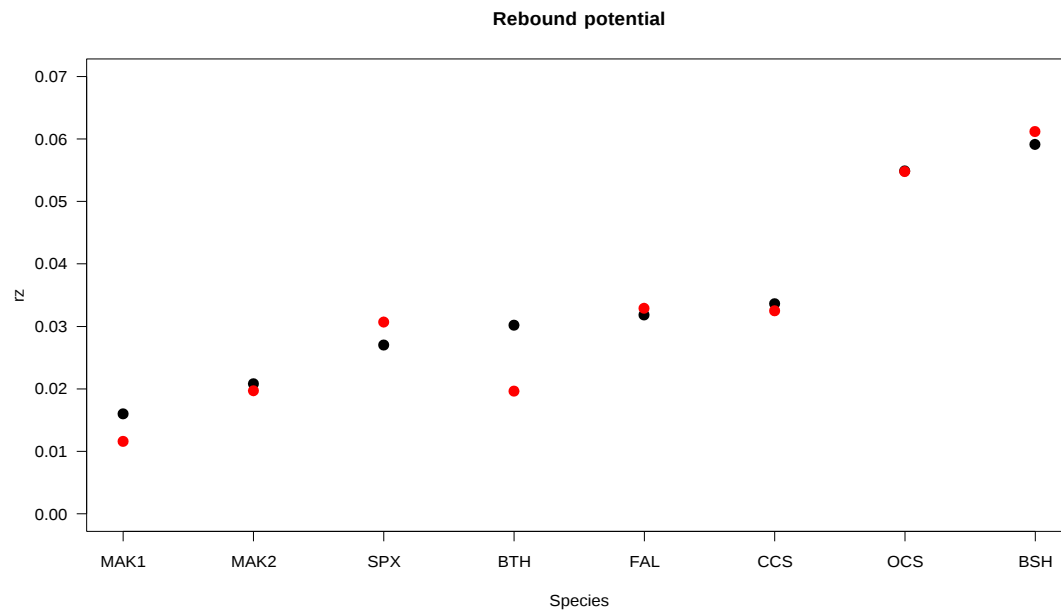


Figure 6. Rebound potential (rzmsy) estimated for large pelagic sharks caught by longliners in the South Atlantic Ocean following Smith et al. (1998). Black dots means that natural mortality estimated by empirical methods was used in the Smith's formulation (M) and red dots means (Zm).

Supporting Online Information

Table SOM 1. Methods used to estimate the natural mortality of shark species. Adapted from Ogle, (2015).

Method	Description	Input parameters
K1	The “one-parameter K equation” from the fourth line of Table 3 in Then et al. (2015)	K
K2	The “two-parameter K equation” from the fifth line of Table 3 in Then et al. (2015)	
JensenK1	The “Jensen (1996) one-parameter K equation”	
JensenK2	The “Jensen (2001) two-parameter K equation”	
tmax1	The “one-parameter tmax equation” from the first line of Table 3 in Then et al. (2015)	Tmax
HoeingO	The original “Hoenig (1983) composite” fitted through ordinary least squares (OLS)	
HoenigOF	The original “Hoenig (1983) composite” for fishes (OLS)	
HoenigOC	The original “Hoenig (1983) composite” for cetaceans (OLS)	
HoenigO2	The original “Hoenig (1983) composite” fitted with geometric mean regression (GMR)	
HoenigO2F	The original “Hoenig (1983) composite” for fishes (GMR)	
HoenigO2C	The original “Hoenig (1983) composite” for cetaceans (GMR)	
HoenigLM	The “modified Hoenig equation derived with a linear model” as described in Then et al. (2015)	
HoenigNLS	The “modified Hoenig equation derived with a non-linear model” as described in Then et al. (2015)	
HewittHoenig	The “Hewitt and Hoenig (2005) equation” from their equation 8	
PaulyLNoT	The “modified Pauly length equation” as described on the	K and Linf
Gislason	The “Gislason et al. (2010) equation” from their equation 2	K, Linf and L
Charnov	The “Charnov et al. (2013) equation” as given in the second	
PaulyL	The “Pauly (1980) equation using fish lengths” from his equation 11.	K, Linf and Temp.
RikhterEfanov1	The “Rikhter and Efanov (1976) equation (#2)” as given in the second column of pge 541 of Kenchington (2014) and in Table 6.4 of Miranda and Bettoli (2007)	T50
RikhterEfanov2	The “Rikhter and Efanov (1976) equation (#1)” as given in the first column of pge 541 of Kenchington (2014)	T50, K, t0, and b
AlversonCarney	The “Alverson and Carney (1975) equation” as given in equation 10 of Zhang and Megrey (2006)	Tmax and K
ZhangMegreyP	The “Zhang and Megrey (2006) equation” as given in their equation 8 but modified for demersal or pelagic fish	Tmax, K, t0, T50 and b

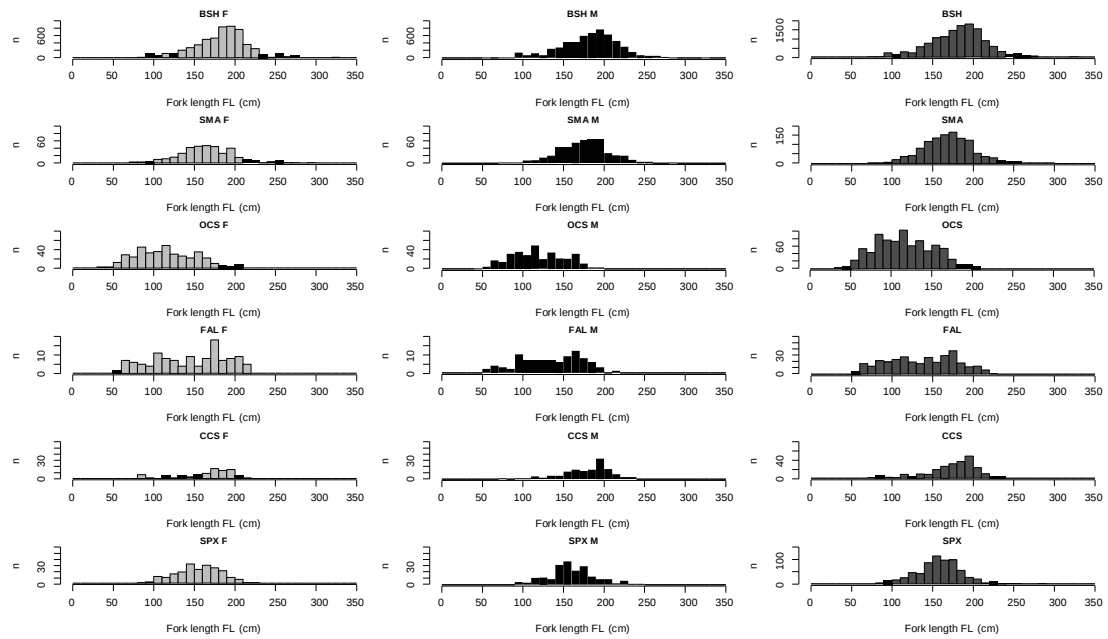


Figure SOM 1. Frequency of shark lengths (FL in cm, by sex and pooled), reported by onboard observers from the Brazilian chartered longline fleet between 2005 and 2011.

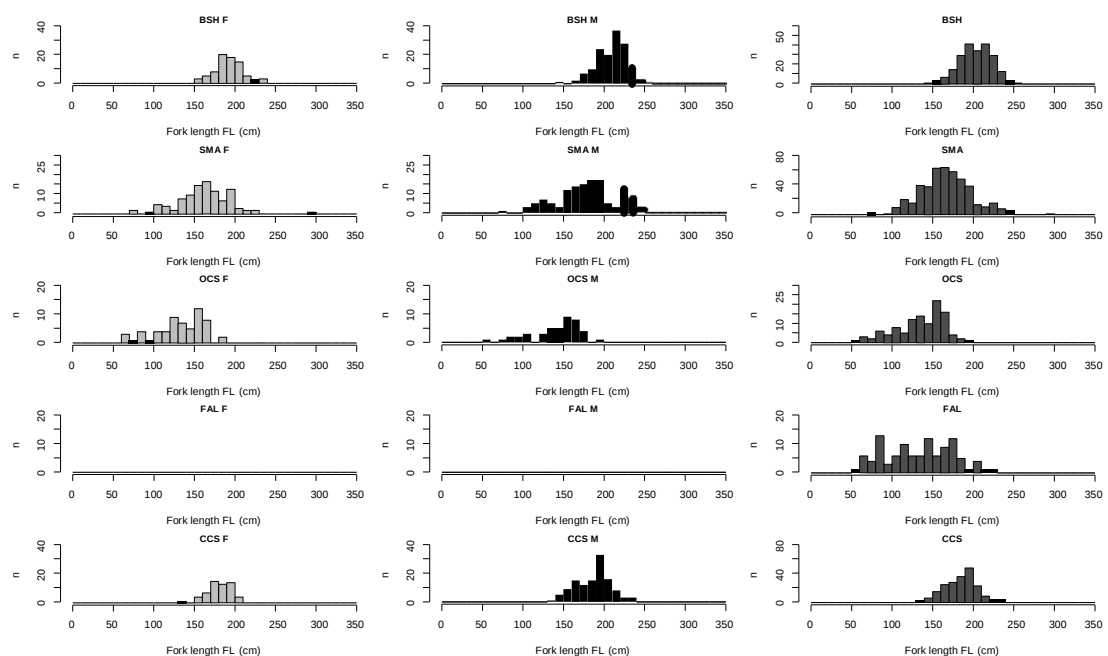


Figure SOM 2. Frequency of lengths of aged individuals sourced from authors who have studied age and growth of the main species in the same study area.

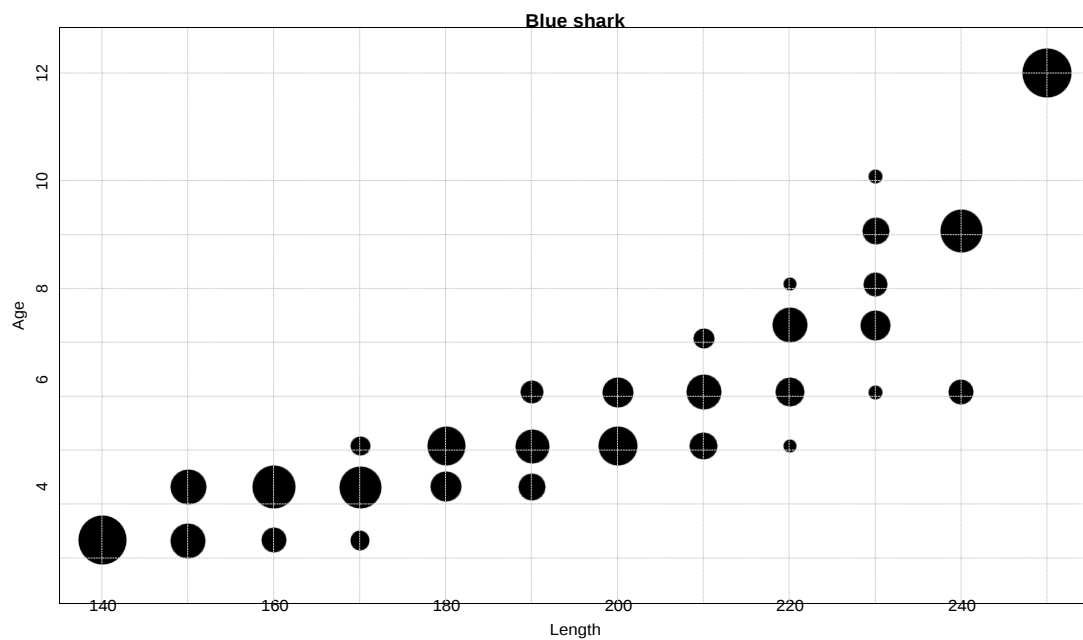


Figure SOM 3. Age length Key (ALK) for blue sharks reported by onboard observers from the Brazilian chartered longline fleet between 2005 and 2011. Unaged individuals smaller or bigger than aged individuals were excluded.

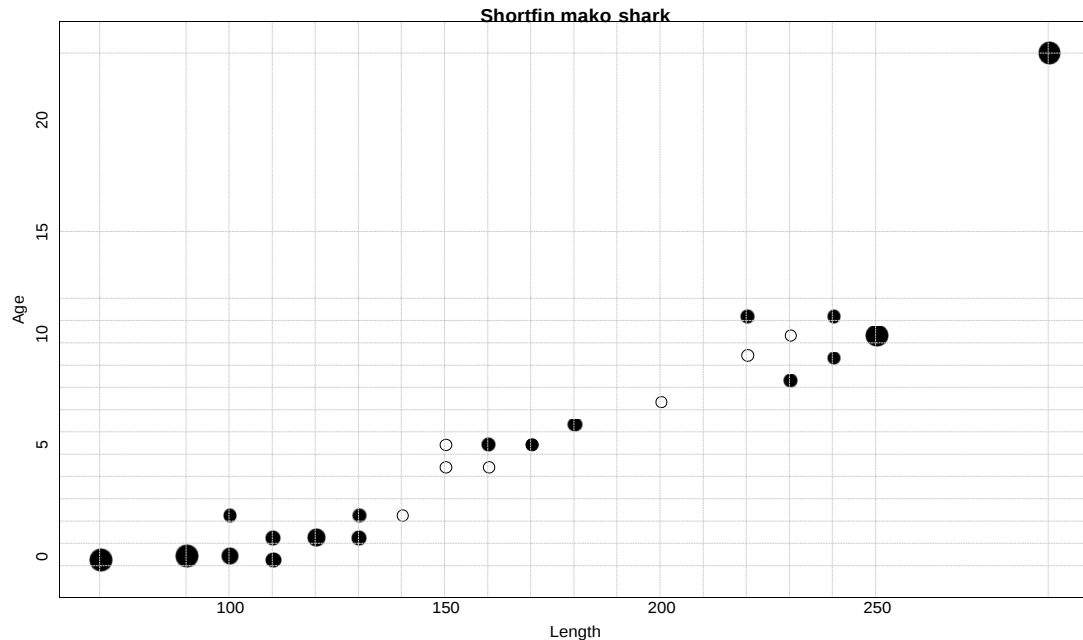


Figure SOM 4. Age length Key (ALK) for mako sharks reported by onboard observers from the Brazilian chartered longline fleet between 2005 and 2011. Unaged individuals smaller or bigger than aged individuals were excluded.

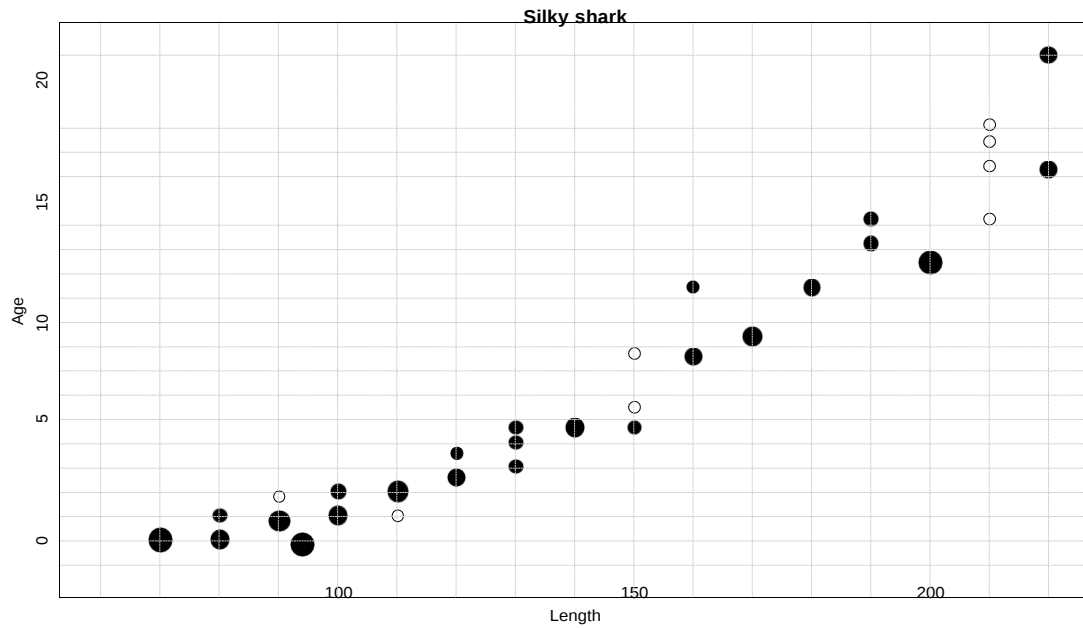


Figure SOM 5. Age length Key (ALK) for silky sharks reported by onboard observers from the Brazilian chartered longline fleet between 2005 and 2011. Unaged individuals smaller or bigger than aged individuals were excluded.

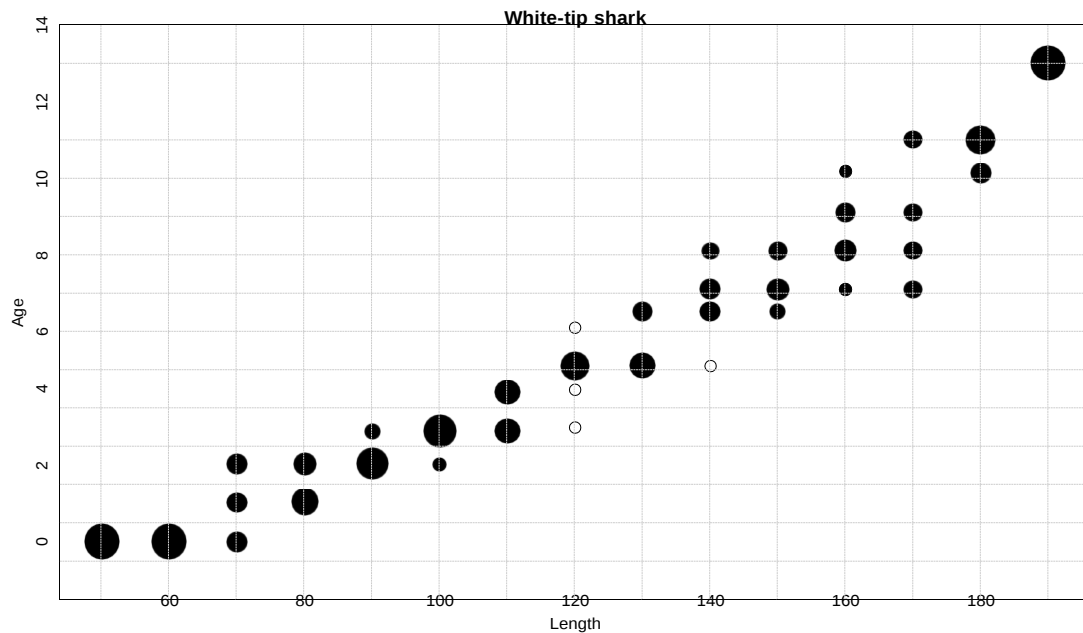


Figure SOM 6. Age length Key (ALK) for white-tip sharks reported by onboard observers from the Brazilian chartered longline fleet between 2005

and 2011. Unaged individuals smaller or bigger than aged individuals were excluded.

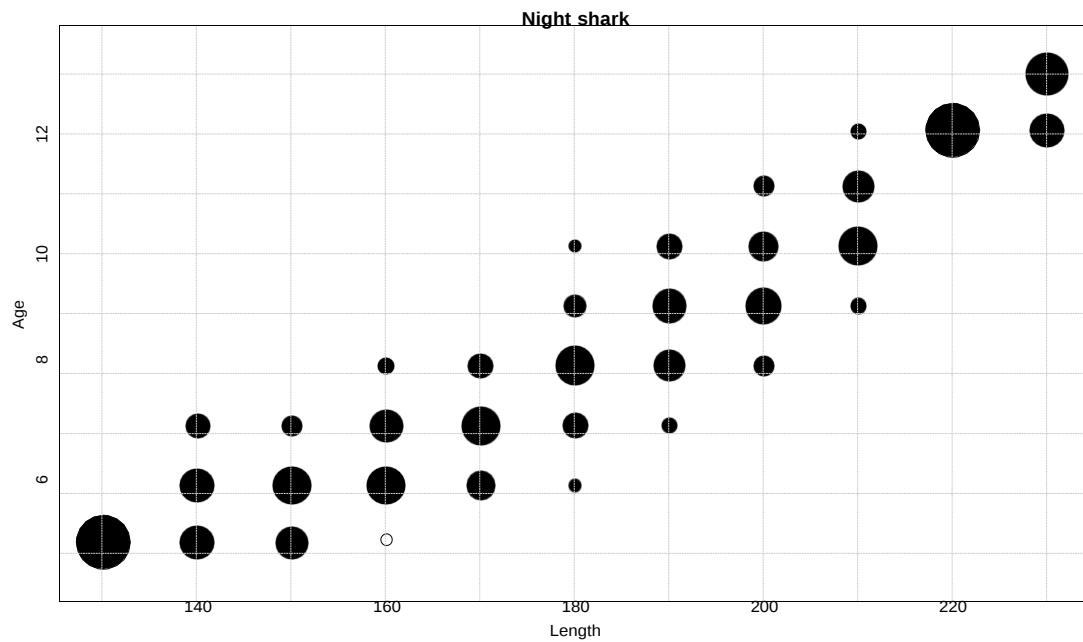


Figure SOM 7. Age length Key (ALK) for night sharks reported by onboard observers from the Brazilian chartered longline fleet between 2005 and 2011. Unaged individuals smaller or bigger than aged individuals were excluded.

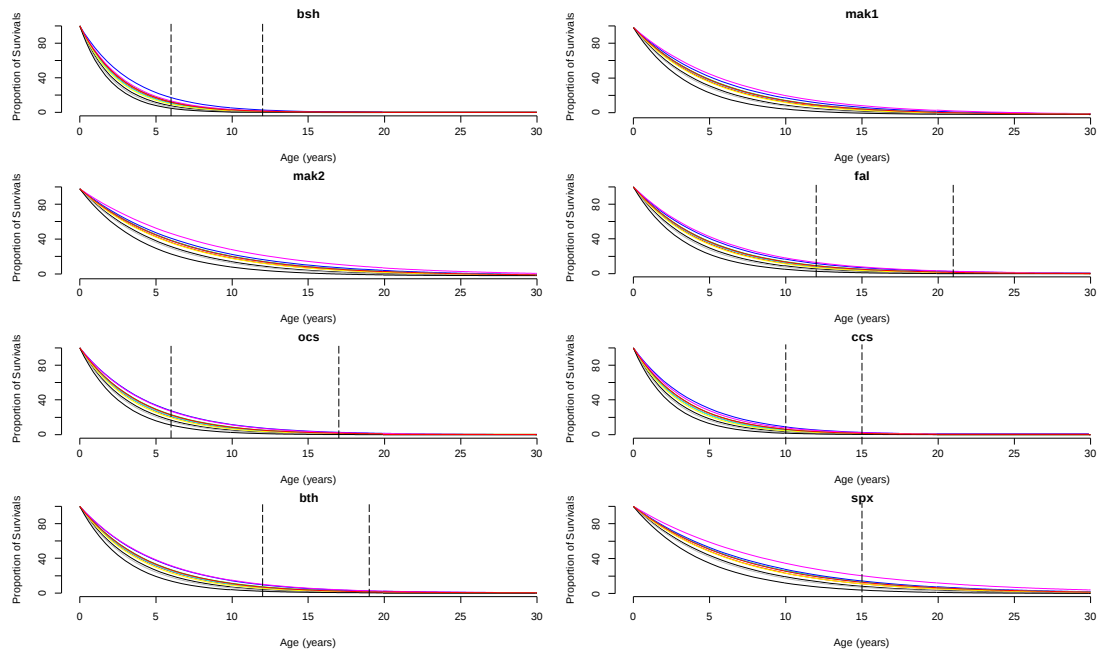


Figure SOM 6. Survivorships for natural mortality (M) estimated from empirical methods using Tmax (Table 3).

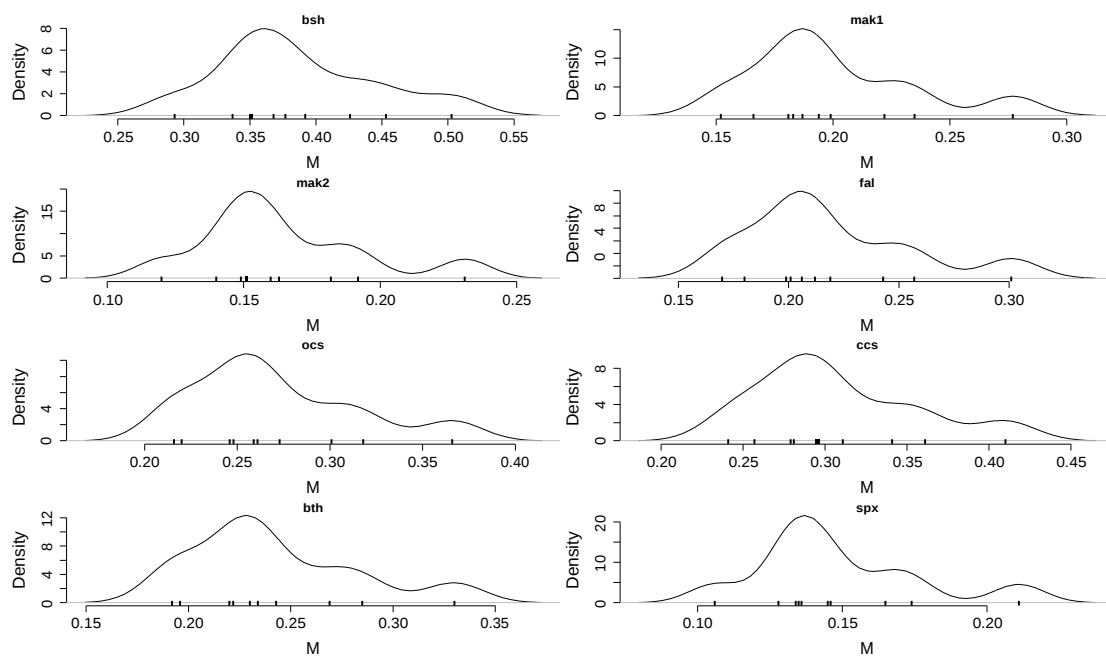


Figure SOM 7. Density plots natural mortality (M) estimated from empirical methods using Tmax (Table 3).

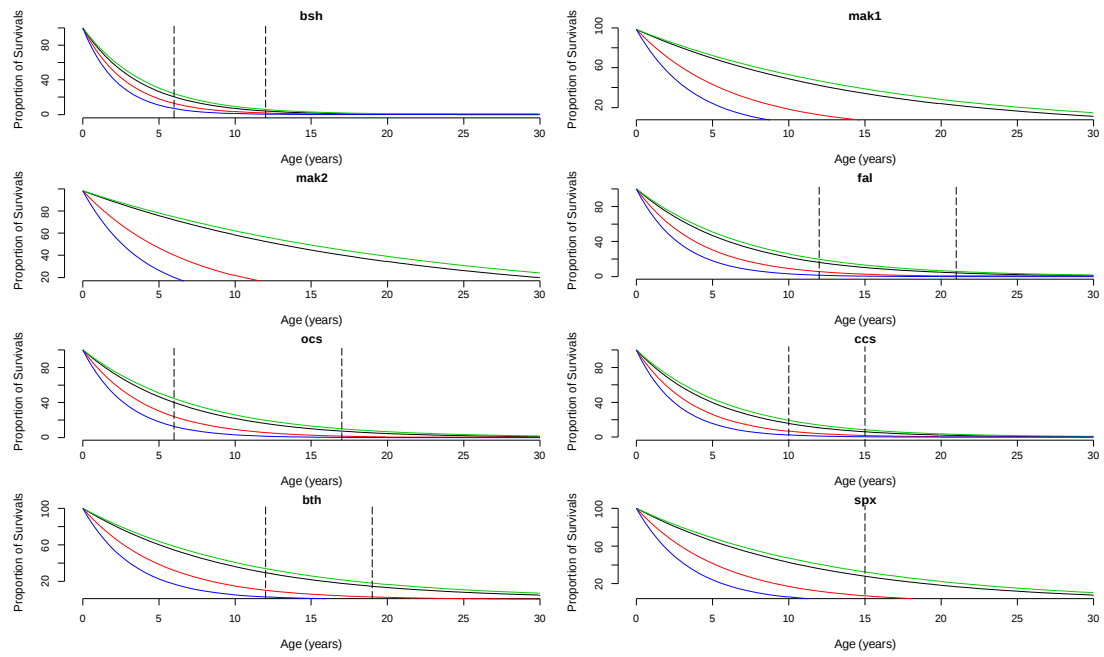


Figure SOM 8. Survivorships for natural mortality (M) estimated from empirical methods using k (Table 3).

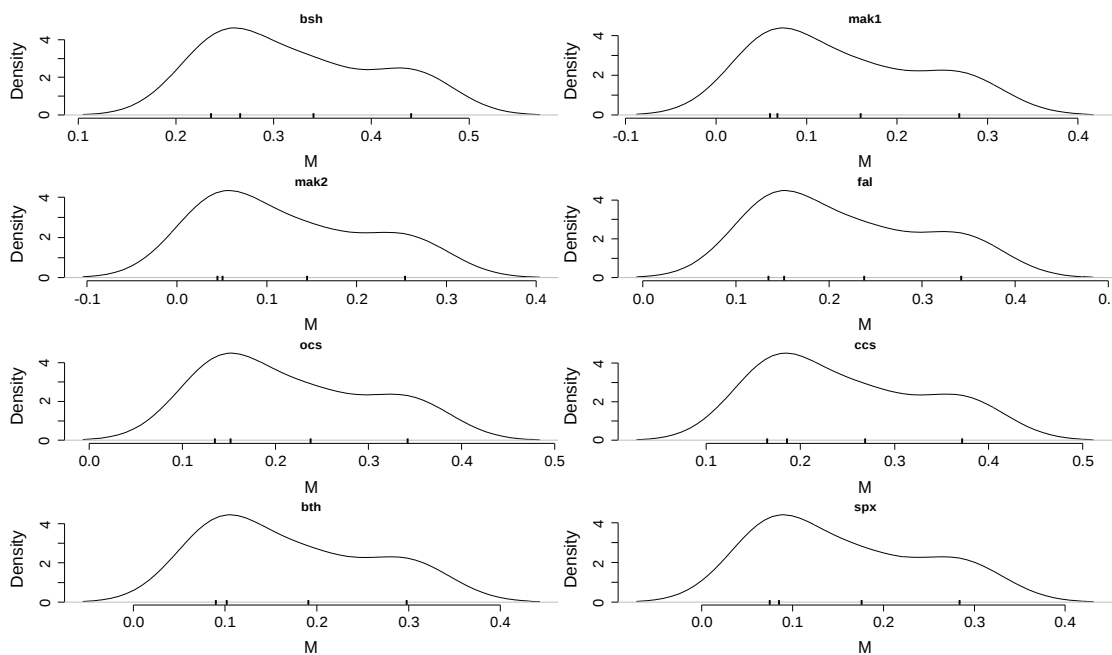


Figure SOM 9. Density plots natural mortality (M) estimated from empirical methods using Tmax (Table 3).

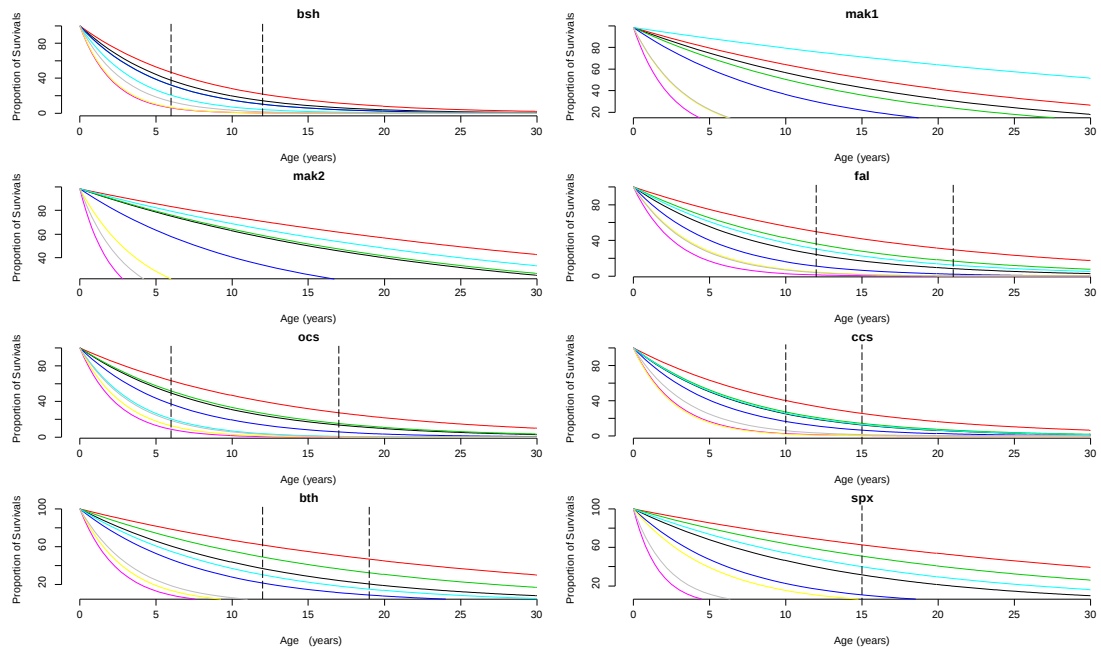


Figure SOM 10. Survivorships for natural mortality (M) estimated from empirical methods using multi parameters (Table 3).

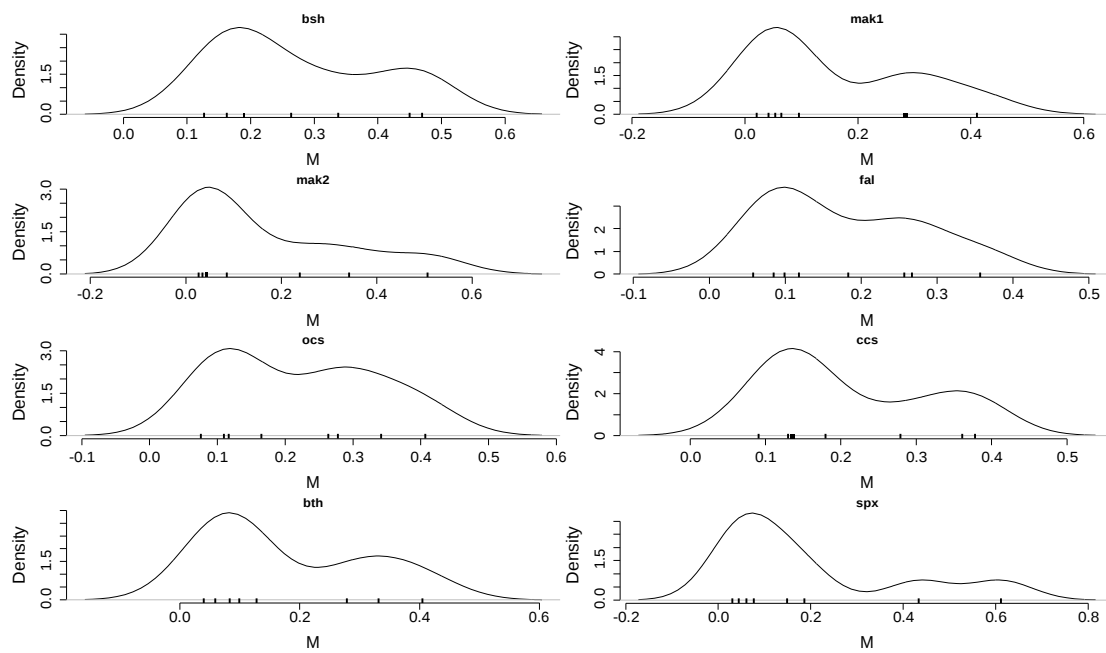


Figure SOM 11. Density plots of natural mortality (M) estimated from empirical methods using multiparameters (Table 3).

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5. Considerações finais

Nossos resultados tanto para a análise das fases de exploração e taxas de captura (cap. 2), quanto em relação a vulnerabilidade (cap. 3) foram consistentes no sentido de demonstrar que praticamente todas as espécies abordadas encontram-se sobreexplotadas e seriamente ameaçadas pelas pescarias de espinhel, que tendem a crescer em virtude do aumento de demanda comercial. Identificamos também, que a parcela das populações (todas as espécies) afetadas por estas pescarias são na maioria jovens, que ainda não atingiram maturidade, tendo sido esta parcela identificada como a principal responsável pelo controle dos níveis populacionais (elasticidades). Como ainda não existe um monitoramento efetivo, nem planos de manejo para estas espécies, consideramos que ações futuras devem focar na melhoria da estatística pesqueira e também no estabelecimento de medidas mitigadoras da mortalidade. Outrossim, tendo em vista que o Brasil deve reavaliar o estado de conservação das espécies nos próximos anos, sugerimos que os tubarões mako e lombo-preto sejam incluídos em categorias de ameaça compatíveis com o estado de conservação dessas espécies, como demonstrado no presente estudo, haja vista que na última avaliação Brasileira estas espécies foram consideradas como próximas a ameaça (NT).