Southern Hemisphere porbeagle shark stock status assessment

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Executive summary

This report presents the results of a Southern Hemisphere stock status assessment of porbeagle shark. The study, along with associated regional studies, was a collaborative one involving many countries with Southern Hemisphere fisheries that catch porbeagles. Participating scientists from Argentina, Chile, Japan, New Zealand and Uruguay contributed data analyses and abundance indices. Our approach combined indicator analyses and a spatially-explicit sustainability risk assessment. Indicator analyses were performed independently for different Southern Hemisphere fisheries and served to characterise local trends in relative abundance based on commercial catch per unit effort (CPUE) data, and trends in size and sex ratio based on biological data.

We limited our analyses to the region south of 30 ^{o}S which provided most of the available data, although the porbeagle shark's range extends slightly north of this latitude. Porbeagle sharks are taken in fisheries at least as far south as 56 ^{o}S. Southern Hemisphere population structure is not well understood, and we considered it unlikely that the population comprises a single well-mixed stock for management purposes. We subdivided the spatial domain of the assessment into five subpopulations or regions by longitude: 1) Western Atlantic Ocean; 2) Eastern Atlantic/Western Indian Ocean; 3) Eastern Indian Ocean; 4) Western Pacific Ocean; and 5) Eastern Pacific Ocean.

We applied different assessment methods by region, depending on data availability and quality. In the Eastern Atlantic/Western Indian Ocean, Eastern Indian Ocean, and Western Pacific regions, stock status assessment was performed using a spatially-explicit risk assessment. Indicator-based analyses were used to assess stock condition in the Eastern Pacific and the Western Atlantic, where there was limited information. We compared results from areas with varying levels of information, for greater insight into the status of the stock, levels of uncertainty, and data requirements for future studies.

Public domain surface longline data were obtained at a resolution of 5 x 5° grid by month by flag from regional fishery management organisations. Catch and effort data were also obtained from other trawl and longline fisheries known to take porbeagle sharks. Japanese observer data on catch and effort throughout the Southern Hemisphere were analysed to determine relationships between catch rates and the covariates year, quarter, latitude, hooks between floats, hooks, and sea surface temperature. These relationships were then used to predict relative abundance across the entire spatial domain, and combined with effort to predict surface longline catches. Catch estimates for other fisheries were obtained from the literature.

Most catch rate indicators were relatively short, variable, and uncertain, with the majority either stable or increasing. Length indicators were also variable. Only the Argentinian size and sex indicators showed temporal trends, with a small decline in sizes for both sexes, and a slight trend towards less female bias in the sex ratio index.

The indicator analyses, in addition to providing time series to monitor population change, revealed spatial patterns in size and sex distributions, and relationships with environmental variables. Such analyses are critical inputs to stock status assessments, because they help to determine model structure.

The risk assessment uses a quantitative framework to estimate spatially-explicit fishing mortality. It derives sustainability status as the ratio of total impact to a maximum impact sustainable threshold (MIST) reference point. The quantitative framework quantifies and propagates uncertainty throughout the assessment process. The risk assessment served to integrate selected CPUE indicators in the evaluation of risk from commercial pelagic longline fisheries to porbeagle shark, within an area subset of the Southern Hemisphere. The spatial domain of the risk assessment covered three regions: Eastern Atlantic/Western Indian Ocean, Eastern Indian Ocean, and Western Pacific Ocean, bounded at 30 ^{o}S and 60 ^{o}S. The Eastern Atlantic/Western Indian Ocean region was selected as the 'calibration region', being the most data-rich. A biomass dynamic model was fitted to the estimated catch and the abundance index for the calibration area. The model estimated a catchability parameter for the pelagic longline effort, which was used to estimate fishing mortality for the calibration area, and extended to other model areas.

Annual fishing mortalities (F) were greatest in the Eastern Atlantic/Western Indian Ocean, slightly lower in the Eastern Indian Ocean, and lowest in the Western Pacific Ocean. Median F decreased from the mid-1980s to 2014 in both the Eastern Atlantic/Western Indian Ocean and Eastern Indian Ocean regions. In the assessment area (three regions combined) in the last decade (2005 to 2014), median F values ranged from 0.0008 to 0.0015 (mean 0.0010).

Risk was determined from the relationship between total impact and the MIST limit reference point for the stock. We reported against three MIST values: F_{crash}, which is the instantaneous fishing mortality that will in theory lead to population extinction; F_{lim}, the instantaneous fishing mortality rate that corresponds to the limit biomass B_{lim}; and F_{msm}, instantaneous fishing mortality rate that corresponds to the maximum number of fish in the population that can be killed by fishing in the long term. Risk values were calculated both as an F-ratio (Impact/MIST) and the probability that F exceeds the MIST, for the period from 1992 onwards (the first year of Japanese CPUE data).

F-ratios for the assessment area declined by half from a 1992–2005 mean for the F_{crash} MIST of 0.068 (range 0.051–0.088), to a 2006–2014 mean of 0.032 (range 0.023–0.042). For the F_{lim} MIST the equivalent numbers were 0.090 (range 0.068–0.118) in 1992–2005 and 0.043 (range 0.031–0.056) in 2006–2014. For the F_{msm} MIST the F-ratios were 0.135 (range 0.102–0.176) in 1992–2005, and 0.063 (range 0.046–0.083) in 2006–2014.

The probability of F exceeding the F_{crash} MIST decreased by 95% from a 1992–2005 mean of 0.0084 (range 0.0015–0.0205), to a 2006–2014 mean of 0.0004 (range 0.0000–0.0013). The probability of F exceeding the F_{lim} MIST similarly decreased from a 1992–2005 mean of 0.0183 (range 0.0073–0.0358), to a 2006–2014 mean of 0.0016 (range 0.0005–0.0040). The probability of F exceeding the F_{msm} MIST decreased from a 1992–2005 mean of 0.0452 (range 0.0213–0.0778), to a 2006–2014 mean of 0.0066 (range 0.0023–0.0133).

In the last 10 years, the southern bluefin tuna (SBT) and albacore/SBT fisheries combined contributed about 75–80% of the fishing mortality in the Western Indian Ocean/Eastern Atlantic Ocean, 70–90% in the Eastern Indian Ocean, and 70–85% in the Western Pacific Ocean.

Thus, results from the risk assessment indicate low fishing mortality rates in the three regions comprising the assessment area, and low risk from commercial pelagic longline fisheries to porbeagle shark over the spatial domain of the assessment. These results are consistent with the trends observed in catch rate indicators over the entire Southern Hemisphere range of the porbeagle shark population, which in most cases show stable or increasing catch rates. Concern has previously been expressed about reduced catch rates in the Western Atlantic Ocean in the Uruguay longline fishery after 1993, but this concern is allayed by the re-analysis undertaken in collaboration with this project.

The population catchability was calibrated assuming that capture mortality was 100% (i.e., post-release survival is zero). Allowing for post-release survival would reduce these fishing mortality estimates, and reduce the estimated risk.

The catch rate indicators are the most important factors driving the results of the status assessment, and their reliability determines its reliability. The indicator trend in the calibration area is the most important factor determining the relatively low estimate of risk.

The risk assessment assumes that population density from 45 to 55 ^{o}S is the same as at 40 to 45 ^{o}S, and that density south of 55 ^{o}S is zero. We have evidence from fisheries and surveys that porbeagles occur south of 45 ^{o}S, but we do not have Japanese longline observer data with which to estimate density. This is an important assumption, because it implies that the low fishing effort south of 45 ^{o}S provides a refuge from fishing mortality for the population. Biological data, and estimated relationships between size and sea surface temperature, suggest that a high proportion of the adult population occurs at these latitudes.

Continued data collection by observers will improve the time series and provide better evidence about abundance trends. Maintaining collection and analysis of indicators from observer data is a key recommendation from this project. The following analyses could be carried out with currently available data:

- Explore assumptions about population density distribution and their effects on risk estimates, by rerunning the assessment with alternative density estimates.
- Explore selectivity at age in the Japanese pelagic longline data, which may permit estimation of the availability at age of the population to fishing. This analysis may permit two further developments: an age-structured analogue of the biomass dynamic risk assessment; and direct estimation of the proportion of the population south of 45 ^{o}S, removing the need to assume constant density from 45 to 55 ^{o}S.
- Further explore available biological data, to understand why patterns differ among areas. For example, it would be useful to model the effects of SST on size and sex patterns in the Chilean swordfish fishery.

The following recommendation would require further data collection:

- Compile biological and catch rate data from fisheries occurring south of 45 ^{o}S, such as the Chilean demersal longline fishery. Some data from this fishery are currently available, and data collection is ongoing.

The following recommendation would require additional, separate studies:

- Study porbeagle distribution using various tool (genetics, microchemistry, stable isotopes, parasites, conventional and electronic tags) to identify biologically-based boundaries.

The multiple indicators/risk assessment approach used in this study served to 1) source and synthesise available information on porbeagle shark at the scale of the Southern Hemisphere; 2) identify important data gaps (e.g., density distribution and life-stage specific vulnerability and overlap with fishing activities); 3) define a productivity-based reference point for the species; and 4) prioritise fishery areas for monitoring and management. This project has filled important information gaps by both directly analysing available life history information, and providing statistical support to the analyses by participating national fisheries scientists.

The project has provided the first assessment of the sustainability of the impact of fishing on the Southern Hemisphere porbeagle shark stock, and laid a foundation for future work. Results indicate that the impact of fishing is low across the entire Southern Hemisphere range of the porbeagle shark population.

1 Introduction

1.1 Historical background

The Western and Central Pacific Fisheries Commission (WCPFC) is one of five tuna Regional Fisheries Management Organisations (t-RFMOs) responsible for the sustainable use, conservation and management of highly migratory species taken by tuna fisheries. Unlike some of the other t-RFMOs, the WCPFC has explicit responsibility for assessing and managing not only tuna species, but also dependent and associated species under Articles 5(d) and 10.1(c) of its Convention. Recognition by the WCPFC of sharks as dependent and associated species in need of conservation and management has resulted in a list of fourteen shark species found in the Western and Central Pacific Ocean (WCPO) for which both data provision and assessment are required (Western and Central Pacific Fisheries Commission 2012). The WCPFC designated the porbeagle shark (*Lamna nasus*) as a key species at its seventh annual meeting in December 2010 but only in areas south of 20 °S due to concerns about species mis-identification in more northerly areas.

The designation of porbeagle as a key species by WCPFC may have been motivated by conservation and management proposals under other intergovernmental treaty organisations (see below) rather than by specific threats posed by WCPFC fisheries per se. This is because none of the WCPFC purse seine effort and only 7% of WCPFC longline fishing effort lies below 20 °S (based on SPC's Catch Effort Query system's raised aggregate data for 2013-2015 for longline effort, and Williams & Terawasi (2016)). As identified in a recent analysis of key shark species conducted by WCPFC's Scientific Services Provider, the Pacific Community (SPC), porbeagle sharks have been recorded in WCPFC observer datasets only in or immediately adjacent to the Australian and New Zealand Exclusive Economic Zones (EEZs; Rice et al. 2015), which represent only a small portion of the range of the Southern Hemisphere porbeagle stock. For this reason, while WCPFC committed to assessing the porbeagle shark's stock status by designating it as a key shark species, it was recognised that a broader regional approach would be necessary to undertake a comprehensive assessment (Rice et al. 2015).

At approximately the same time (March 2012), the Commission for the Conservation of Southern Bluefin Tuna (CCSBT) also identified porbeagle shark as a species of interest. In 2013, New Zealand compiled metadata on porbeagle biology, life history and catch and effort data to support an assessment (Clarke et al. 2013). Subsequently, the CCSBT Ecologically-Related Species Working Group (ERSWG) in March 2015 agreed to request the Common Oceans (Areas Beyond National Jurisdiction (ABNJ)) Tuna Project, through its Technical Coordinator-Sharks and Bycatch position based at the WCPFC, to progress this work with the ERSWG and across the joint t-RFMOs. The ERSWG made this request on the basis that it would facilitate access to a broader range of data sets than would be available through the ERSWG members alone, and importantly cover the whole stock for assessment.

The Common Oceans (ABNJ) Tuna Project (www.commonoceans.org) is a partnership between the five t-RFMOs, as well as governments, inter- and non- governmental organisations, and the private sector, aimed at sustainable and efficient tuna fisheries production and biodiversity conservation. It focuses its efforts on marine resources that do not fall under the responsibility of any one country, thus working both in coastal and high seas areas. One set of activities of this Global Environment Fund (GEF)-funded project aims at reducing the impact of tuna fisheries on biodiversity by improving data and assessment methods for sharks, thereby promoting their sustainable management. Within this set of activities WCPFC is leading four stock status assessment studies for Pacific-wide shark stocks. The first study, a stock status assessment of the bigeye thresher shark (*Alopias superciliosus*) was completed in September 2016 (Fu et al. 2016). The Southern Hemisphere porbeagle shark was selected as another species of interest as its distribution is not only pan-Pacific but global, making a cooperative, inter-regional approach particularly important. The objectives of the Common Oceans (ABNJ) Tuna Project assessments include evaluating whether current t-RFMO management schemes are adequate, supporting national management actions such as Convention on International Trade in Endangered Species (CITES) Non-Detriment Findings, and demonstrating new modes of international cooperation for the assessment of highly migratory species.

1.2 Biology and distribution

Porbeagle sharks are cold-temperate, wide-ranging, coastal and oceanic sharks (Compagno 2001). Recent studies show they undergo both diel and reverse diel vertical movement patterns, and exhibit coastal site fidelity as well as large-scale open ocean movement (Pade et al. 2009, Campana et al. 2010b, Saunders et al. 2011, Francis et al. 2015). This species is distributed in the Northern Hemisphere from approximately 20^{o}–75 ^{o}N but only in the Atlantic Ocean and Mediterranean Sea. It is absent from the North Pacific Ocean, where the closely related salmon shark, *Lamna ditropis*, fills its niche. In contrast, Southern Hemisphere porbeagle sharks have a circumpolar distribution (Last & Stevens 2009, Ebert et al. 2013). Although they are mainly caught between 30 ^{o}S and 50 ^{o}S, in the South Pacific porbeagles have sometimes been caught further north in the austral winter (June to August) and spring (September to November); in summer (December to February), they are not found north of about 35 ^{o}S. In summer and autumn, Southern Hemisphere porbeagle sharks appear to penetrate further south, and are found near many of the sub-Antarctic islands in the Indian and southwest Pacific Oceans (Francis & Stevens 2000).

The Northern and Southern hemisphere porbeagle shark populations are genetically and biologically distinct, and geographically disjunct (Figure 1; Testerman et al. 2007, Kitamura & Matsunaga 2010). As a result, the two populations have quite different life history characteristics: the Southern Hemisphere porbeagle is a smaller form that grows more slowly and lives twice as long as its northern conspecifics (Francis et al. 2007, Clarke et al. 2015). In the North Atlantic, porbeagles are often found close to shore but they also occur in the open ocean: mature females make long migrations into the subtropical waters of the central North Atlantic to give birth (Pade et al. 2009, Campana et al. 2010b, Saunders et al. 2011, Biais et al. 2017). In the Southern Ocean porbeagles are commonly caught in pelagic habitats far from shore, but also occur in coastal waters (Yatsu 1995, Francis & Stevens 2000, Semba et al. 2013, Francis et al. 2015). Limited tagging results from New Zealand confirm that Southern Hemisphere porbeagles undergo seasonal north-south movements and some make longitudinal movements of several thousand kilometres (Francis et al. 2015). It is not known whether there is a single circumpolar population in the Southern Hemisphere or whether there are multiple stocks or sub-stocks spread over this wide range.

Life history data for the Southern Hemisphere population derive primarily from studies in New Zealand and Australia; there is scant life history information from other Southern Hemisphere areas (International Commission for the Conservation of Atlantic Tunas 2010, Clarke et al. 2015). Length at birth is 58–67 cm fork length (FL; Francis & Stevens 2000); females mature at around 170–180 cm FL (age 13-16 years) and males at about 140–150 cm FL (age 6-8 years) (Francis & Duffy 2005, Francis 2015). Longevity is unknown but may be more than 65 years (Francis et al. 2007). Porbeagles are live-bearers (aplacental viviparous) and exhibit uterine oophagy with embryos feeding on other ova produced by the mother (Francis & Stevens 2000). The gestation period is about 8–9 months. Litter size is usually four embryos, with a mean litter size in the southwest Pacific of 3.75 (Francis & Stevens

2000). If the reproductive cycle lasts one year, annual fecundity would be about 3.7 pups per female (Francis et al. 2008).

Figure 1. World distribution of porbeagle shark. NB: The northern and southern range limits of the Southern Hemisphere population are not well known and may be unreliable. Source: IUCN (http://maps.iucnredlist.org/map.html?id=11200).

Two studies of the age and growth of New Zealand porbeagles produced growth curves that show that males and females grow at similar rates up to about 10 years of age (about 150cm FL) and diverge thereafter with male growth approaching an asymptote while females continue to grow at a similar rate (Figure 2). The second study (Francis 2015) obtained slightly younger ages for a given length than the earlier study (Francis et al. 2007) because of a modified vertebral band pair counting protocol. Both studies cautioned that growth parameters are probably only accurate for ages up to about 20 years (because growth bands in older sharks become too narrow to be resolvable with a light microscope) and require further validation (Clarke et al. 2015).

In New Zealand, porbeagle sharks recruit to commercial fisheries during their first year at about 70 cm FL, and much of the commercial catch is immature (Francis 2015). Most sharks caught by tuna longliners are 70-170 cm FL. The size and sex distribution of both sexes is similar up to about 150 cm, but larger individuals caught by New Zealand fisheries are predominantly male; few mature females are caught. Regional differences in length composition suggest segregation by size (Francis 2013). Porbeagles caught by the Argentinean surimi (trawl) fleet had median fork lengths of 182 cm and 167 cm for females and males respectively, and a relatively high proportion of adults (62% of females and 82% of males) (Cortés et al. 2017). Porbeagles are active pelagic predators mainly of fish, but squid are also commonly eaten especially by the small sharks (Griggs et al. 2007, Horn et al. 2013).

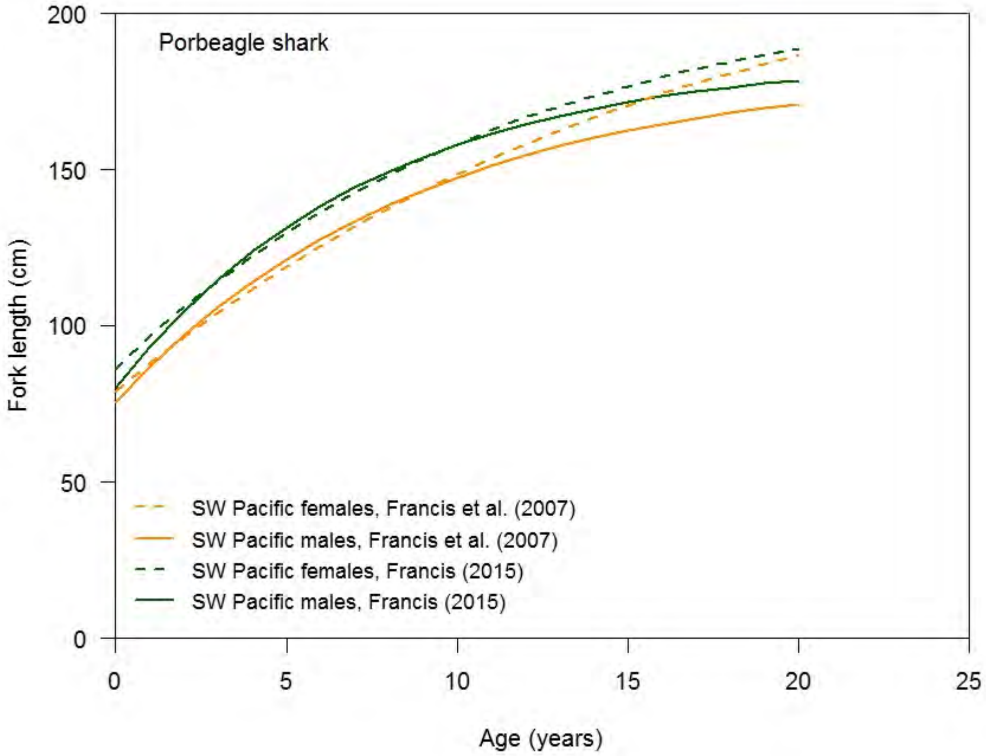

Figure 2. Growth curves for porbeagle shark (reproduced from Clarke et al. 2015).

A study of the intrinsic rate of population increase of 38 species of sharks indicated that porbeagle shark has low productivity, similar in reproductive potential to some of the coastal carcharhinids (such as *Carcharhinus plumbeus* (sandbar shark) and *C. obscurus* (dusky shark)) and the pelagic thresher shark (*Alopias pelagicus*) (Cortés 2002). A comparison of productivities of twenty pelagic shark stocks in the Atlantic suggested that the porbeagle shark has relatively lower productivity than most species examined (13th of 20), although it had higher productivity than the bigeye thresher (*Alopias superciliosus*) and higher or similar to mako sharks (*Isurus* spp.) (Cortés et al. 2015). In terms of overall vulnerability (i.e. productivity and susceptibility) the porbeagle was the third most vulnerable shark of the 20. It should be noted, however that these studies were based on life history characteristics of the Northern Hemisphere porbeagle population. A similar ecological risk assessment of the Southern Hemisphere population in the Indian Ocean suggested the porbeagle was the seventh most vulnerable of the 17 species considered (Murua et al. 2012). Given that the Southern Hemisphere porbeagle has a longer generation time than its Northern Hemisphere conspecific, it may be more vulnerable to depletion.

1.3 Population trends

To date, most population-level studies of porbeagle sharks have been conducted for the North Atlantic, where this species has been highly valued for its meat and as a popular target for recreational fishing for several decades. As a result, porbeagle stocks in the North Atlantic currently show signs of serious overfishing in the form of greatly diminished catches compared to peak periods (Campana et al. 2008, Food and Agriculture Organization of the United Nations 2017). There has been less attention to the status of Southern Hemisphere stocks, perhaps in part owing to catches from the southern stock being generally incomplete (Food and Agriculture Organization of the United Nations 2017). While there is no known, thriving market for its meat in the Southern Hemisphere, porbeagles have in the past been utilized for their fins (e.g. in the southern bluefin tuna fishery) as well as retained whole in countries such as New Zealand (Clarke et al. 2013, Food and Agriculture Organization of the United Nations 2017).

The most comprehensive attempt to assess porbeagle stocks was conducted by the International Commission for the Conservation of Atlantic Tunas (ICCAT) in 2009 (International Commission for the Conservation of Atlantic Tunas 2010). Separate analyses were conducted for the northeast, northwest and South Atlantic (Figure 3).

In the northeast Atlantic, there was considerable uncertainty in identifying the current stock status relative to virgin biomass because the peak of the fishery occurred well before the earliest points in the abundance indices. Nevertheless, the ICCAT assessment agreed with the view of the Northeast Atlantic Fisheries Commission (NEAFC, the Regional Fishery Body) that the stock was in a depleted state. It found that if catches were limited to zero, the stock would rebuild to its maximum sustainable yield level in 15–34 years, but if the current allowable catch was maintained rebuilding would require longer, possibly over 100 years (International Commission for the Conservation of Atlantic Tunas 2010).

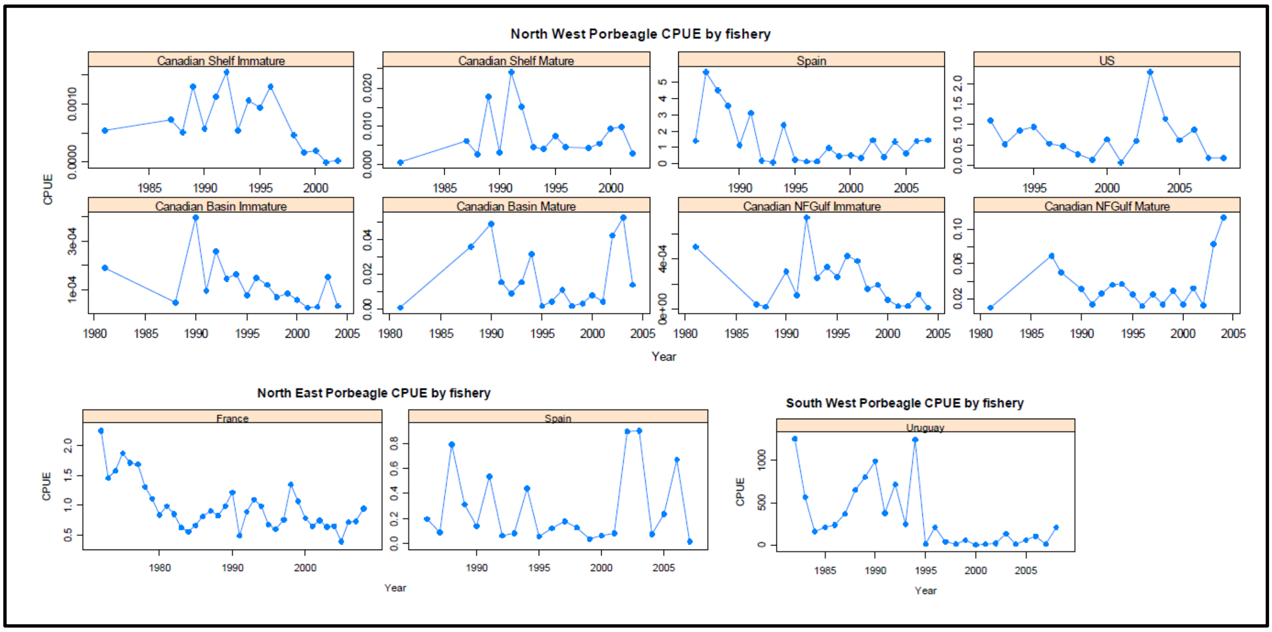

Figure 3. Catch per unit effort series for the northwest Atlantic (upper figures), northeast Atlantic (lower left figures) and southwest Atlantic (lower right figure) stocks (International Commission for the Conservation of Atlantic Tunas 2010).

In the northwest Atlantic, ICCAT's work was compared to, and found to agree with, an earlier Canadian stock assessment for coastal waters (Campana et al. 2010a). Both assessments concluded that the population is highly depleted but recovering under current management implemented by Canada and the United States. However, depending on the stock productivity and fishing mortality assumptions applied, recovery is projected to be achieved on the order of decades to over 100 years (International Commission for the Conservation of Atlantic Tunas 2010, Campana et al. 2012). There was no appreciable difference in the results when previously unaccounted for high seas catches were estimated and included in the model (International Commission for the Conservation of Atlantic Tunas 2010).

ICCAT's assessment for the South Atlantic was hampered by limited data for southwest and southeast regions. In the southwest there was an apparent decline in catch rates in the Uruguayan fleet, with models suggesting that overfishing was occurring and that the stock was overfished. In the southeast, catch rates appeared stable since the early 1990s but biomass levels could not be estimated. The overall result for both regions was that a robust conclusion on stock status could not be drawn (International Commission for the Conservation of Atlantic Tunas 2010).

A study based on a large Japanese observer and research survey dataset from 1982 to 2011 provided the most comprehensive view of the Southern Hemisphere stock (Figure 4) (Semba et al. 2013). One of the findings of that study was that large adult porbeagles penetrate into colder waters at higher latitudes, beyond the range of the southern bluefin tuna longline fishery, so they may not be subject to large-scale fishing pressure. In support of this theory the authors reported no declining trend in relative abundance in the Southern Ocean longline fishery from 1994 to 2011 (Semba et al. 2013).

In the Pacific, New Zealand has conducted indicator analyses for longline-caught porbeagle shark assessing trends in distribution, catch composition, abundance, size and sex ratios (Francis et al. 2014, Francis & Large 2017). There was some inconsistency among trends identified for porbeagle shark by the distribution and CPUE indicators, and by the standardised CPUE indices for the northern and southern New Zealand fisheries. Furthermore, some CPUE models fitted the data poorly and may be unreliable. Nevertheless, when taken as a group, the indicators suggested that the porbeagle population around New Zealand has been stable or increasing since 2005. Prior to that time observer data suggested a decline in abundance in the late 1990s and early 2000s followed by stability at a relatively low level.

SPC also conducted an indicator analysis which included porbeagle shark data for the wider South Pacific (Rice et al. 2015). That analysis benefitted from the inclusion of older records off Tasmania from the Australian observer programme in the 1990s (possibly identified as makos before then (Bruce 2014)), but mainly relied on the New Zealand observer records in more recent years (Figure 5). Not surprisingly, Rice et al. (2015) found the same pattern of high but variable porbeagle catch rates in the late 1990s followed by a low, fluctuating and slightly increasing catch rates thereafter. Rice et al. (2015) also concluded that most observed porbeagles were smaller than the size at maturity.

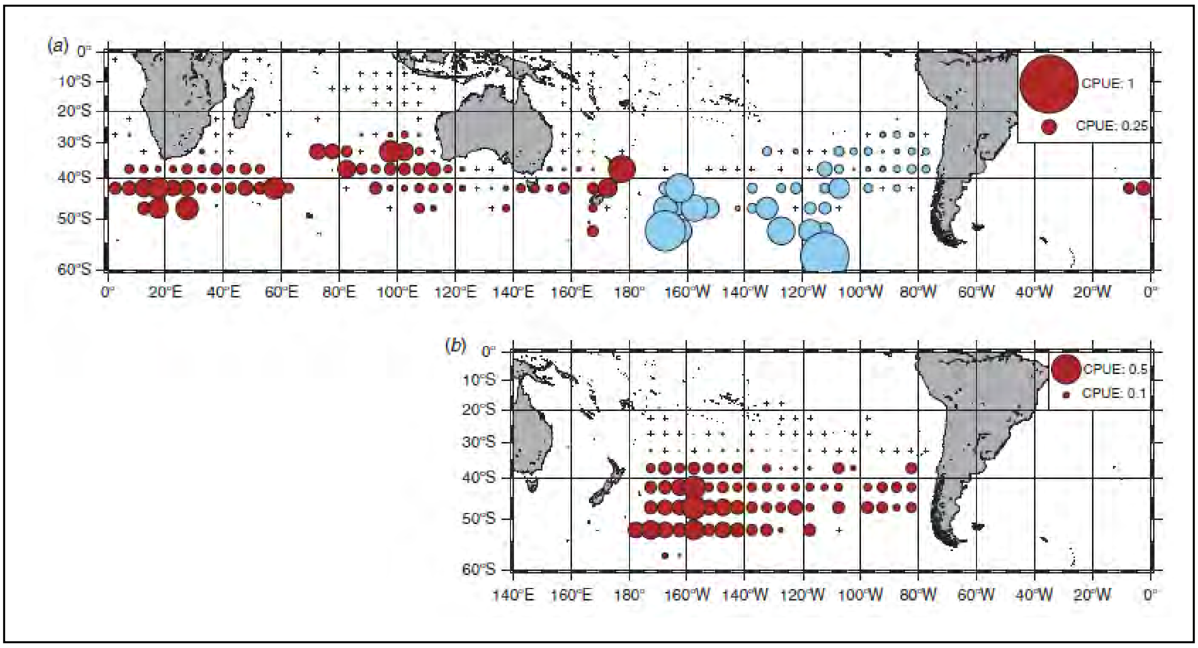

Figure 4. Catch per unit effort for porbeagle longline (top) and drift net (bottom) fishing gear in the Southern Ocean. In the top panel observer data are shown in red and survey data are shown in blue. Crosses denote no catch. (Semba et al. 2013).

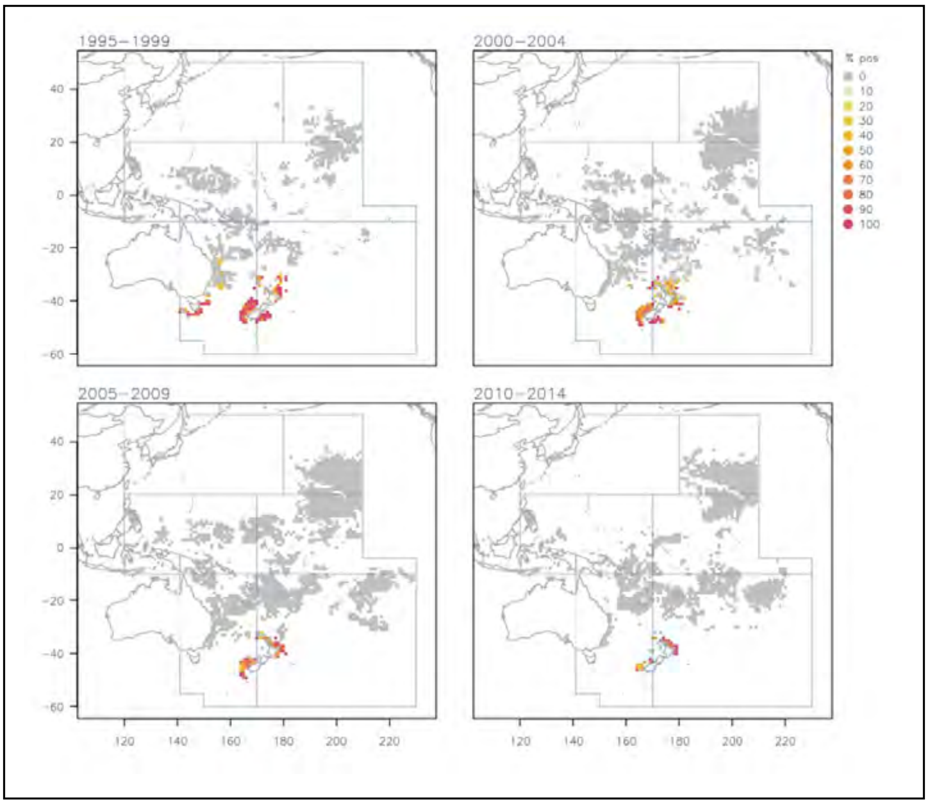

Figure 5. Spatial distribution of the proportion of longline sets for which one or more porbeagle sharks were caught for each five-year period between 1995 and 2014 (Rice et al. 2015).

1.4 Current conservation and management designations and measures

The IUCN Red List classifies porbeagle sharks as "Vulnerable" based on population trends from the Northern Hemisphere and Uruguay as of 2006 (Stevens et al. 2006). Since that assessment several international organisations have adopted protections for the porbeagle as follows:

2008 Listed on Appendix II of the Convention on the Conservation of Migratory Species of Wild Animals (CMS), which encourages international cooperation toward conservation.

2010 Added to the CMS Memorandum of Understanding (MOU) for Sharks, which will develop a Conservation Plan to guide cooperation between the signatories to CMS Convention as well as other interested stakeholders (Convention on Migratory Species 2016)

2011 NEAFC, citing the porbeagle shark's low productivity and high vulnerability to overfishing, prohibited directed fishing and mandated prompt release. This measure will remain in effect until the end of 2019 (Northeast Atlantic Fisheries Commission 2016).

2012 Added to Annex II of the Barcelona Convention. In response, the General Fisheries Commission for the Mediterranean (GFCM) agreed under GFCM/36/2012/3 to prohibit retention on board, trans-shipping, landing, transferring, storing, selling or displaying or offering for sale porbeagle specimens caught in the Mediterranean (Food and Agriculture Organization of the United Nations 2012).

2014 Listed on Appendix II of CITES, requiring that all exports of porbeagle sharks, including landings in non-flag State ports, be accompanied by permits issued by the flag state CITES Management Authority. Export permits are contingent upon legal acquisition and non-detriment findings (NDFs), the latter of which represents a certification by an authorized CITES Scientific Authority that the proposed export is not detrimental to the survival of the species (Clarke et al. 2014).

2015 ICCAT adopted a recommendation requiring that porbeagles be released promptly and unharmed, to the extent practicable (International Commission for the Conservation of Atlantic Tunas 2015).

Various countries also have adopted management measures for porbeagle specifically including:

European Union	Zero total allowable catch by European Union vessels (European Union 2015).
Canada and the United States	Catch limits for the northwest Atlantic stock (International Commission for the Conservation of Atlantic Tunas 2010, National Oceanic and Atmospheric Administration 2016).
New Zealand	Catch limits under a quota management system (Ministry for Primary Industries 2016).
Australia	A requirement to release porbeagles brought up alive in Australia (Australian Fisheries Management Authority 2017).
Uruguay	Since January 2013, a prohibition on retaining porbeagles by Uruguayan-flagged vessels and foreign vessels fishing in the Uruguayan EEZ fisheries (R. Forselledo, personal communication, July 2017).
Argentina	A prohibition of directed fishing and a requirement to release live porbeagles (and other shark species) longer than 1.6 m (Federal Fishery Council of Argentina 2013).

Other organisations with fishing grounds lying within the range of the Southern Hemisphere porbeagle population (assumed to be south of 20 ^{o}S) have enacted measures applicable to sharks in general. No-retention measures for all commercial take of sharks have been adopted by the French Overseas Territories of New Caledonia and French Polynesia, the British Overseas Territory of Pitcairn, and the Cook Islands. The Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) implemented a moratorium on all directed shark fishing in the Antarctic region in 2006 and encourages the live release of incidentally caught sharks (Commission for the Conservation of Antarctic and Marine Living Resources 2006).

In response to a petition to list the porbeagle under the United States' Endangered Species Act, the National Oceanic and Atmospheric Administration (NOAA) undertook a comprehensive status review of both the Northern and Southern hemisphere populations in 2016. This resulted in a finding that neither population is currently in danger of extinction throughout all or a significant portion of its range or likely to become so in the foreseeable future and thus listing was not warranted. With regard to the Southern Hemisphere population in particular, NOAA's review found that abundance is stable or increasing, but that there is some uncertainty about current stock status (Curtis et al. 2016, National Oceanic and Atmospheric Administration 2016).

1.5 Status evaluation

This report presents the results of a status assessment of Southern Hemisphere porbeagle shark. The study was a collaborative one involving many countries with Southern Hemisphere fisheries that catch porbeagles. Participating scientists from Argentina, Chile, Japan, New Zealand and Uruguay contributed data analyses and abundance indices used and considered during the risk assessment. The study team supported this work by providing analytical advice to participating scientists during the development of the indicators.

The indicators are described in separate papers, which have been submitted to WCPFC-SC13 as Information Papers. Analyses of Japanese longline data (Hoyle et al. 2017b) provided catch rate, size and sex indicators for the Eastern Atlantic/Western Indian Ocean, Eastern Indian Ocean, and Western Pacific regions; a basis for estimating the spatial distribution of porbeagle sharks worldwide; and a basis for imputing catches in pelagic longline fisheries. Analyses of New Zealand fisheries data provided catch rate, size, and sex indicators (Francis & Large 2017) and catch estimates (Francis 2017) for the Western Pacific region. Analyses of Chilean swordfish fishery data provided catch rate, size, and sex indicators and catch estimates for the Eastern Pacific region (Hoyle et al. 2017a). Analyses of Uruguayan longline data provided catch rate indicators (Forselledo et al. 2017) for the Western Atlantic region. Analyses of Argentinean surimi trawl fishery data (Cortés et al. 2017) provided catch rate, size, and sex indicators for the Western Atlantic region.

Our approach combined indicator analyses and a spatially-explicit sustainability risk assessment. Indicator analyses were performed independently for different Southern Hemisphere fisheries/study partners and served to characterise local trends in relative abundance based on commercial catch per unit effort (CPUE) data, and trends in size and sex ratio based on biological data. We also considered a more complex age and length-structured assessment, using Stock Synthesis software (Methot & Wetzel 2013).

Risk assessment tools have been developed in response to data limitation problems in the evaluation of fishing effects on non-target species, including sharks and other elasmobranch species (Stobutzki et al. 2002, Braccini et al. 2006, Griffiths et al. 2006, Cortés 2008, Zhou & Griffiths 2008, Cortés et al. 2010, Gallagher et al. 2012, Cortés et al. 2015). We adapted and modified the risk assessment methods developed by Fu et al. (2016) in a stock status assessment of bigeye thresher shark, *Alopias superciliosus*. The method uses a quantitative framework for estimating spatially-explicit fishing mortality and deriving a sustainability status for the species as the ratio of total impact to a maximum impact sustainable threshold (MIST) reference point. Rather than following a traditional stock assessment approach, which relies heavily on population processes that for sharks are often poorly understood, this spatially-explicit approach is based on species productivity, inferred distribution and data on the occurrence, characteristics and intensity of fishing. The quantitative framework allows uncertainty to be quantified and propagated throughout the assessment process. An important outcome is that impact, sustainability risk and uncertainty can be partitioned spatially and among fishery sectors, allowing more focused management. The risk assessment served to integrate selected CPUE indicators in the evaluation of risk from commercial pelagic longline fisheries to porbeagle shark within an area subset of the Southern Hemisphere having the greatest amount of data. Indicator-based analyses were then used to assess condition in the remainder of the Southern Hemisphere (see Section 2.1). This combined approach allowed us to integrate results from areas with varying levels of information, and to gain greater insight into the status of the stock, levels of uncertainty, and the data requirements for future studies.

2 Methods and Results

The overall approach to the risk assessment is presented in Figure 6, which summarises the data inputs, analytical methods and key parameters.

The Methods and Results section covers a wider range of issues and comprises seven parts: (2.1) Assessment stock structure, which describes the spatial configuration of the assessment and identifies which methods are applied by area; (2.2) Effort data, which describes the effort data; (2.3) Population distribution/density, in which we fit a model to Japanese observer data and use it to infer species distribution across the entire spatial domain; (2.4) Catch data and estimation, which provides catch estimates for all fisheries in all regions, based both on reported catch and inferred by combining effort with predicted catch rates; (2.5) Indicator analyses, which describes the development of population indicators, which are used both directly as indicators of population status, and within the risk assessment; (2.6) Risk assessment, which describes the risk assessment procedure applied to three of the five assessment regions; and (2.7) Quantitative stock assessment, which discusses the potential to apply age-structured modelling approaches to the Southern Hemisphere porbeagle shark population.

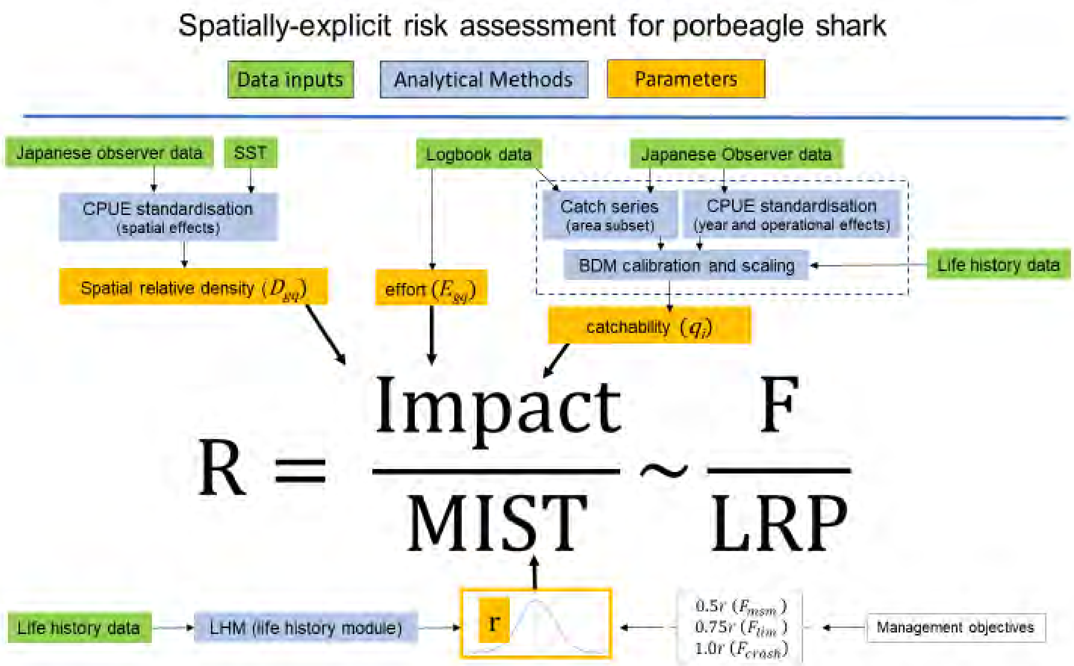

Figure 6. Conceptual representation of data inputs, analytical methods and key parameters used in spatially-explicit risk assessment of the Southern Hemisphere porbeagle shark. BDM = Bayesian state-space biomass dynamics model. The dashed outline box represents analytical methods applied to a region subset of the available data.

2.1 Assessment stock structure

This study covers the entire Southern Hemisphere porbeagle shark population. Porbeagles have been reported in fisheries or surveys circum-globally in the Southern Hemisphere, so all longitudes are included. Porbeagles are found as far north as 20 °S, though catch rates are very low north of 30 °S. Porbeagle catches have been reported from further north, including at the equator, but logbook data include reporting errors, and we considered these northernmost catches to be errors. We limited analyses, and therefore the assessment domain, to the area south of 30 °S to avoid problems fitting analyses to strata without any catch. In the Southern Hemisphere, most longline porbeagle catch is taken north of 45 °S, but this is probably due to the distribution of the southern bluefin tuna longline fishery rather than the distribution of porbeagle sharks.

Porbeagle sharks are also taken in fisheries at least as far south as 56 °S, such as the mackerel icefish and Patagonian toothfish trawl and longline fisheries in the Heard and McDonald Island EEZ, in New Zealand midwater trawl fisheries, in the Argentinian surimi trawl fishery, and in the Chilean demersal longline fishery. Porbeagle sharks were observed in the eastern Pacific to the southern limits of the JAMARC longline survey (60 °S) and the JAMARC gillnet survey (52.5 °S) (Yatsu 1995, Semba et al. 2013, Hoyle et al. 2017b). Porbeagle sharks may be found elsewhere, but few data are available.

We considered that Southern Hemisphere porbeagles are unlikely to comprise a single well-mixed stock for management purposes. Initial observations of trends in population indices from Japanese longline data (the most comprehensive dataset available) suggested that they may vary spatially (Hoyle et al. 2017b), although reanalysis of the Japanese observer data shows reasonably stable catch rates across three regions (see Section 2.5). Nevertheless, the spatial scale of the Southern Hemisphere is very large relative to observed longitudinal movement rates of porbeagles (Francis et al. 2015). Depletion of one longitudinal band may take considerable time to affect the population outside that area. Fisheries interactions suggest a higher incidence of juveniles in northern areas (Semba et al. 2013) and the majority of shark movements appear to be in the north−south direction (Francis et al. 2015), suggesting that mixing occurs across latitudes.

The potential for subdivision is apparent, but the population structure is not well understood. It is possible that there are subgroups within the population that have not been identified. There is some evidence for population structuring by age class and/or reproductive class by longitude as well as by latitude, based on analyses of size and sex patterns in Japanese and Chilean observer data (Hoyle et al. 2017b, Hoyle et al. 2017a). Genetic analyses have even suggested the possibility of independent populations within the South Atlantic (Kitamura & Matsunaga 2010).

Based on this understanding of stock structure, we subdivided the Southern Hemisphere porbeagle stock into five subpopulations or regions by longitude, with the divisions based on variation in fishing effort and on geographical features (Figure 7). For comparison, Northern Hemisphere porbeagle sharks in the North Atlantic are managed as two separate stocks, one on each side of 42 °W, given low levels of population interchange, and evidence for site fidelity and homing behaviour (Biais et al. 2017). The five subpopulations (hereafter called regions) of Southern Hemisphere porbeagle shark defined in this study were:

1. Western Atlantic Ocean (70° to 10° W);
2. Eastern Atlantic/Western Indian Ocean (10° W to 70° E);
3. Eastern Indian Ocean (70° to 140° E);

4. Western Pacific Ocean (140° to 180° E); and

5. Eastern Pacific Ocean (180° E to 70° W).

We applied different assessment methods by region, depending on data availability and quality. In the Eastern Atlantic/Western Indian Ocean, Eastern Indian Ocean, and Western Pacific regions, stock status assessment was performed using a spatially-explicit risk assessment. Indicator-based analyses were used to assess condition in the Eastern Pacific and the Western Atlantic, where there was limited information.

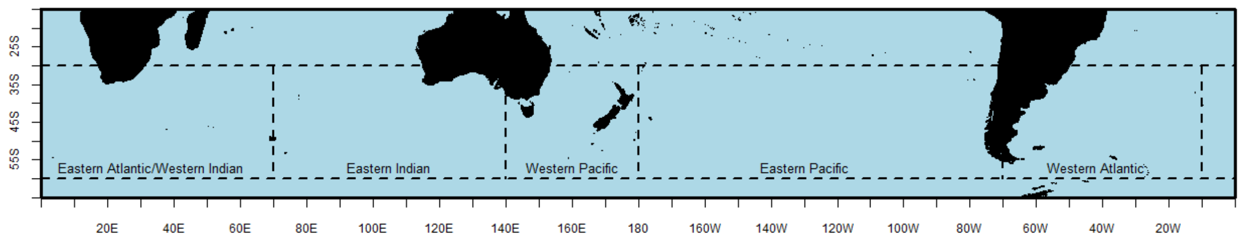

Figure 7: Spatial subdivision of the Southern Hemisphere porbeagle population into five regions.

2.2 Effort data

Public domain surface longline data were obtained at a resolution of 5 x 5° grid by month from the following regional fishery management organisations: the Western and Central Pacific Fisheries Commission (WCPFC), the Inter-American Tropical Tuna Commission (IATTC), the International Commission for the Conservation of Atlantic Tunas (ICCAT), the Indian Ocean Tuna Commission (IOTC), and the Commission for the Conservation of Southern Bluefin Tuna (CCSBT).

Each dataset was adjusted to the same reference frame, with location marking the centre of the 5 x 5° grid, and longitudes 0–360°. The WCPFC, CCSBT and ICCAT datasets were affected by the three-vessel confidentiality rule, according to which data are only reported for time-region strata that include data from at least three fishing vessels.

To address data loss due to the three-vessel rule, public domain catch and effort data for the Western and Central Pacific Ocean were requested from the WCPFC for the period 2004–2014, in two formats: a) stratified by year, month, 5 x 5° grid (latitude and longitude), and flag (WCP_FLAG), and b) stratified by year, quarter, and 5° latitude (WCP_LAT). Both public domain datasets omit strata that include fewer than three vessels, to avoid potential identification (Western and Central Pacific Fisheries Commission 2007), which meant that more data were omitted from the less aggregated dataset WCP_FLAG. However, WCP_FLAG has higher spatial resolution. We therefore used the WCP_LAT dataset to calculate a multiplier with which to scale up the effort in WCP_FLAG for each year, quarter and latitude band to match the total effort in WCP_LAT, while retaining the distribution by 5° grid and month. Public domain catch and effort data were also obtained from the WCPFC website for the period 1950–2014, stratified by year, month, and 5 x 5° grid.

Atlantic Ocean effort data were obtained from the Task II catch and effort database (https://www.iccat.int/en/accesingdb.htm) for the period 1961–2014. For longline, these data are aggregated by flag, year, month and 5 x 5° grids. ICCAT avoids identifying individual vessels by omitting strata with observations from fewer than three vessels. For this dataset no information on total effort was available for scaling, so total effort was underestimated.

Effort data for parties reporting to the CCSBT were obtained from the public domain file https://www.ccsbt.org/userfiles/file/data/CEData_Longline.xlsx for the period 1965–2015. These data are aggregated by flag, year, month and 5 x 5° grids. The CCSBT data are known to be affected by the three-vessel rule but could not be adjusted, so total effort was underestimated.

Indian Ocean effort data were obtained from the IOTC website (http://www.iotc.org/documents/ce-longline) for the period 1960–2015. These data are aggregated by flag, year, month and 5 x 5° grids. The IOTC further aggregates data prior to release at a coarser resolution wherever there would otherwise be potential to identify individual vessels. Thus, all catch and effort data were included in the IOTC dataset. A small amount of IOTC effort was reported in days rather than hooks, and this was omitted.

Effort data from the Eastern Pacific were obtained from the IATTC website for the period 1963–2014, aggregated by flag, year, month and 5 x 5° grids. This dataset includes all the longline data exactly as provided by the countries.

A check of CCAMLR data holdings in November 2016 revealed a total of three reported captures (and subsequent release) under the generic code 'sharks, skates and rays' in bottom longline fisheries (S. Mormede, NIWA, pers. comm.). Based on this, no further data were requested from CCAMLR.

Trawl data were provided at a resolution of 5 x 5° grid and month by the South Pacific Regional Fisheries Management Organisation (SPRFMO). No porbeagle catch was reported in this fishery.

2.3 Population distribution / density

The spatially-explicit risk assessment methodology uses the spatial overlap of fishing effort and population density to derive a risk metric. This requires estimation of relative population density over the spatial domain of the assessment. Spatially-explicit porbeagle density was estimated at the same spatial resolution as the available effort data (i.e., in 5 x 5 degree grids). Population distribution was inferred from the spatial component of catch rates in the Japanese tuna longline fishery. When standardising catch and effort data to produce indices of abundance, the spatial representation of the models included both latitude and longitude. However, this approach only allows relative distribution to be estimated for longitudes and latitudes for which we have Japanese longline effort data. By removing longitude from the models and including sea surface temperature, we could use known values of sea surface temperature to predict relative abundance for all locations between 30 °S and 45 °S, circum-globally.

Analyses used a delta lognormal approach, first modelling the probability of nonzero catch, and then modelling the distribution of catch rates in the nonzero catches.

$$por \neq 0 \sim yr + qtr + s(\,lat, k = 10) + s(hbf, k = 5) + s(hooks, k = 10) + s(SST, k = 10)$$

$$log\left(\frac{por}{hooks}\right) \sim yr + qtr + s(lat, k = 10) + s(hbf, k = 5) + s(SST, k = 10)$$

Latitude (*lat*), hooks between floats (*hbf*), hooks per set (*hooks*) and sea surface temperature (*SST*) were modelled as continuous variables using smoothers, which allow for nonlinear relationships. The term '*s*' refers to a one-dimensional thin-plate regression spline smooth, and *k* sets the upper limit on the degrees of freedom associated with the smooth. The *hooks* term was included in the delta component of the model because the probability of non-zero catch applies to the complete set, while the lognormal component measures catch per hook. Year (*yr*), and quarter (*qtr*) were modelled as categorical variables.

Residuals for the positive component of the model indicated that all variables were statistically significant (Table 1), with reasonable fit to the data but some skewness (Figure 8). Results indicated strong relationships between SST and catch rates, particularly for the probability of nonzero catch, and relatively stable catch rates through time (Figures 9 and 10).

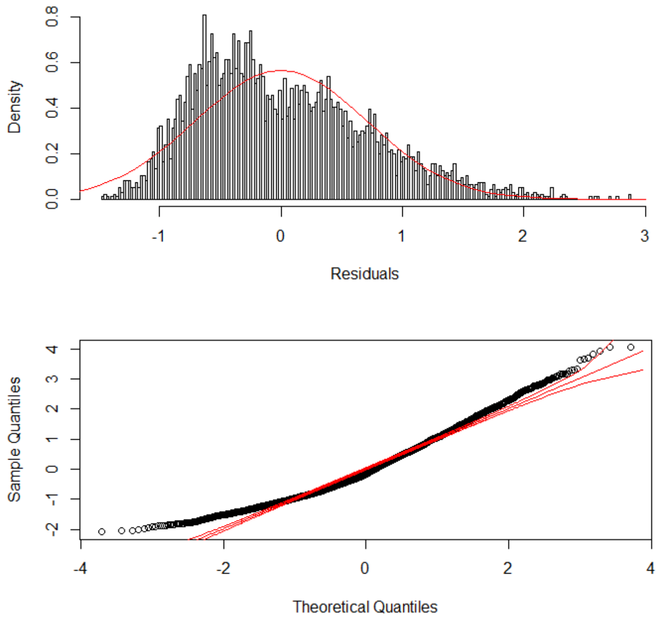

Figure 8: Residual distribution plots for the lognormal positive observer data analysis for catch prediction.

Table 1: ANOVA table for variables in the binomial and positive components of the delta lognormal standardisation model. For smooth variables degrees of freedom (DF) are the effective degrees of freedom calculated by the *mgcv* package.

Model type	Data type	Parameter	DF	F	p-value
Binomial	categorical	op_yr	22	292.0	< 2e-16
Binomial	categorical	qtr	3	52.4	2.52E-11
Binomial	smooth	s(lat)	5.1	52.5	2.32E-09
Binomial	smooth	s(hbf)	2.6	54.8	1.27E-11
Binomial	smooth	s(hooks)	7.9	68.2	2.95E-11
Binomial	smooth	s(SST)	7.8	844.0	< 2e-16
Positive	categorical	op_yr	22	14.0	< 2e-16
Positive	categorical	qtr	3	12.4	4.22E-08
Positive	smooth	s(lat)	8.6	16.0	< 2e-16
Positive	smooth	s(hbf)	3.2	6.3	9.29E-05
Positive	smooth	s(SST)	8.5	19.1	< 2e-16

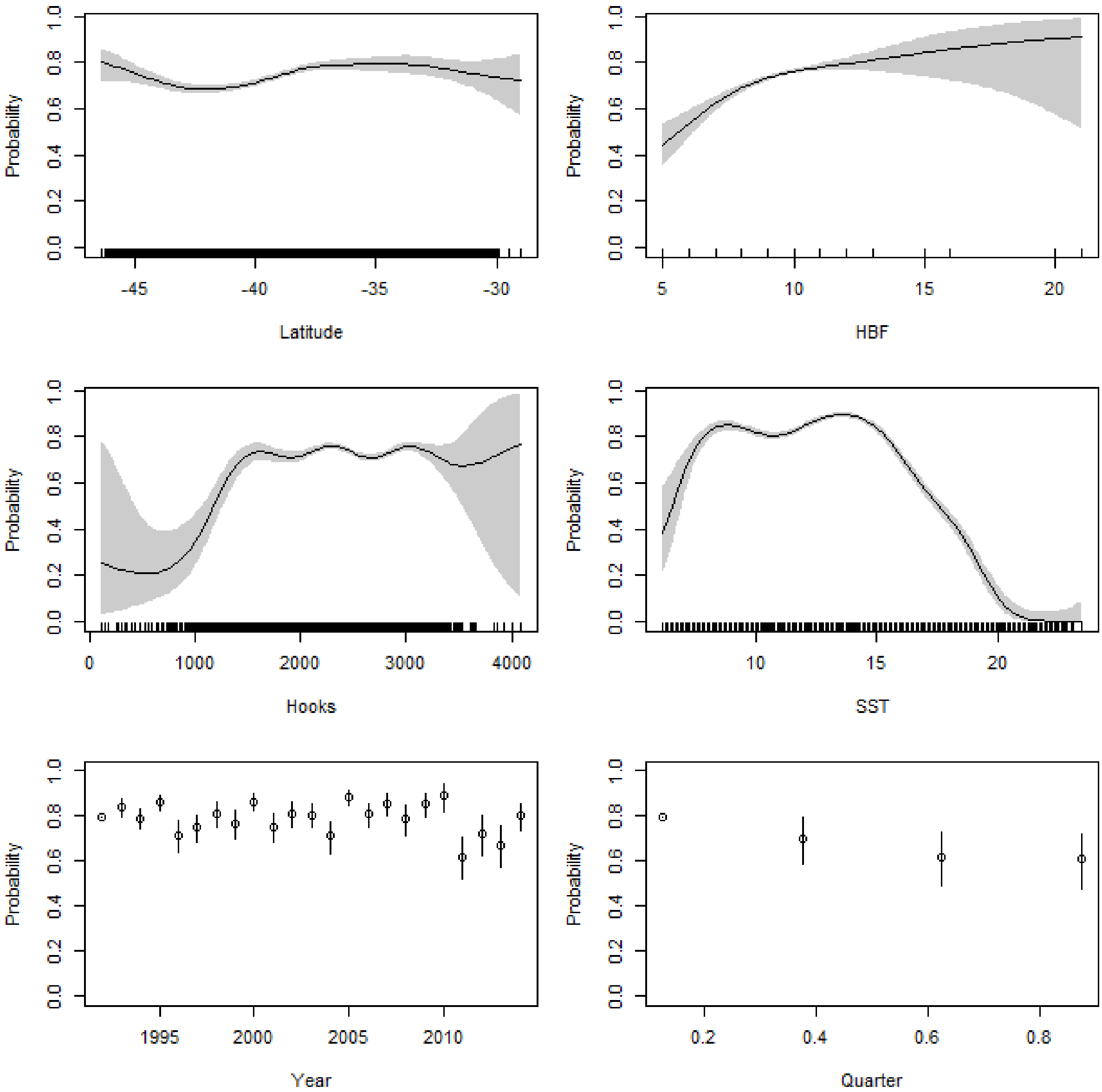

Figure 9: Probability of nonzero catch predicted for pelagic longline effort, for values of each covariate with other values held fixed. Fixed values were latitude 40 °S, HBF 11, SST 12 °C, Year 1992, and Quarter 0.125 (first quarter).

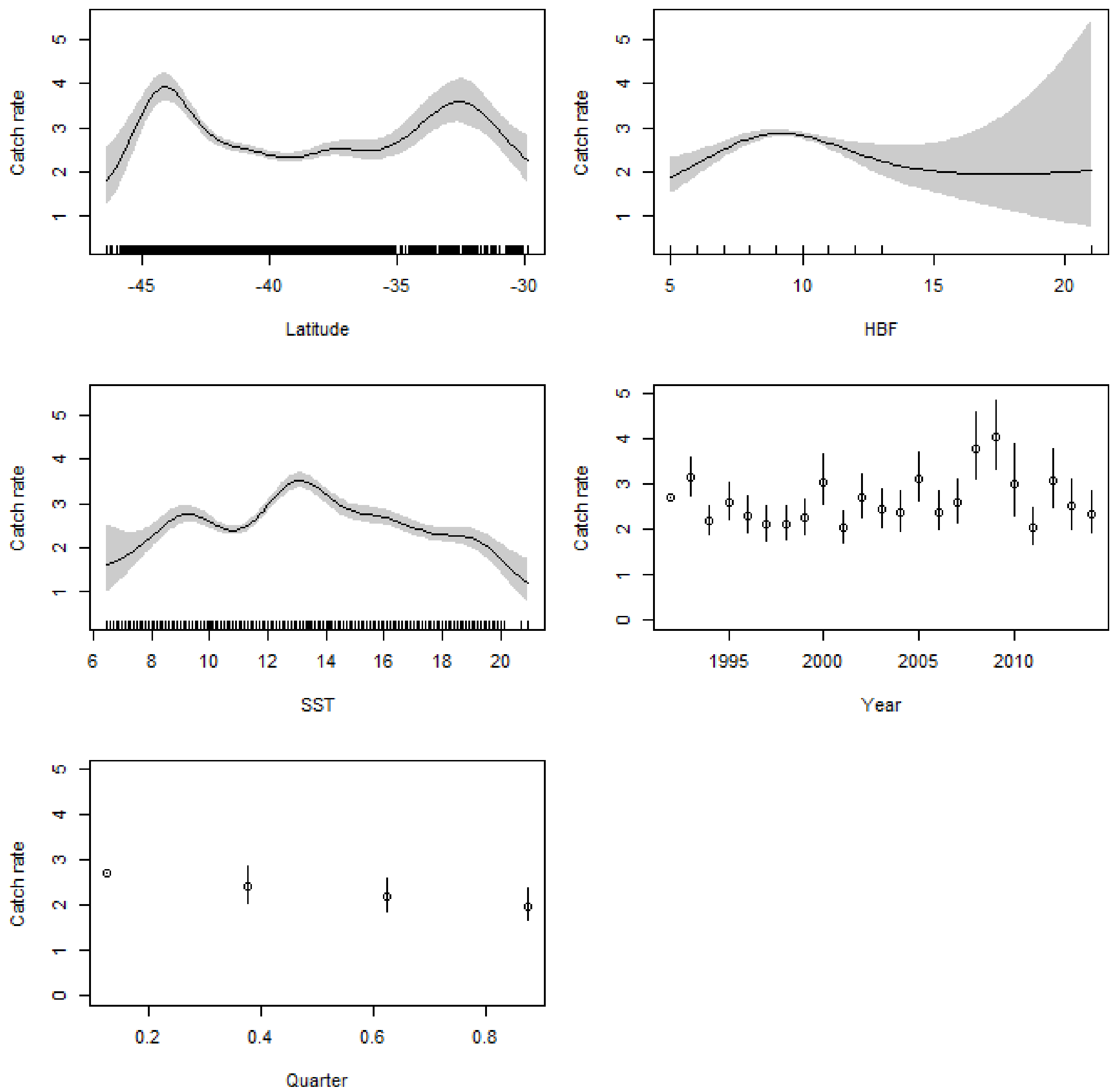

Figure 10: Predicted catch rate for nonzero pelagic longline effort, for values of each covariate with other values held fixed. Fixed values were latitude 40 °S, HBF 11, SST 12 °C, Year 1992, and Quarter 0.125 (first quarter).

Catch rates were predicted for the available pelagic longline effort at stratification of 5 x 5 grid by month, for a standard year. The original analysis used set level data, and predictions were made for a standard set. Each set was assumed to use 3000 hooks, with 11 hooks between floats. SST was predicted for the month and 5 x 5 grid based on the CSIRO CARS 2009 Atlas of Regional Seas (Ridgway et al. 2002), which provides monthly predictions averaged across years. Patterns of sea surface temperature varied strongly by latitude, longitude, and month (Figure 11). Please note that month is used to derive SST, but is not itself a covariate. Each month was allocated to a quarter, which was a covariate. Separate predictions were made for the probabilities of nonzero catch, and the catch rates in positive sets. For each stratum, the two predictions were multiplied to give expected catch rate. Each monthly distribution was normalized to have a maximum of 1.

We assumed that variation in catch rate with temperature and latitude was associated with relative abundance rather than catchability, and predicted relative density in space by month (Figure 12).

By using the same predictive model to estimate the population distribution for the entire spatial domain, this approach assumes that population densities, depletion levels, and trends are similar in all regions. The only differences are caused by SST and latitude. It also assumes that the relationships of latitude and SST with catch rate estimated from the Japanese longline data are applicable to all other pelagic longline effort. The resulting predictions suggest relatively consistent longitudinal gradients in porbeagle density within latitudinal bands over the Southern Hemisphere. This may or may not represent true population density, as porbeagle distribution may be patchy, with aggregation in some areas.

2.4 Catch data and estimation

For most of the pelagic longline effort, direct information on catches was not available. Reporting in Japanese logbooks of porbeagle catches was generally poor before 2008. We had little information about porbeagle catch rates for most other fleets. The best available information was derived from the Japanese observer data. Catches (in numbers of sharks) were therefore estimated from the expected catch rates estimated above (Section 2.3) and the observed effort (see Section 2.2). Catch rates were predicted as for the relative density prediction, but for all years rather than a standard year. Predictions could only be generated for years with catch rate estimates. For years without such estimates, the catch rates for 1992 were applied as a conservative assumption. Expected catches per stratum were estimated by multiplying the observed effort by the expected catch rate.

Porbeagle catch estimates for all fisheries in the New Zealand EEZ, including midwater trawl and longline fisheries, were provided by New Zealand (Francis 2017). Trawl effort and estimated porbeagle catches were provided for the Argentinian surimi fleet (Cortés et al. 2017).

Observer data for the Chilean swordfish fishery were provided for three sectors: industrial longline, artisanal longline, and artisanal gillnet (Hoyle et al. 2017a). Coverage for the artisanal sectors was reported to be 3% and for industrial longline 87%. Observed catches were scaled up to annual catch estimates by dividing the catch records by year and sector by the appropriate coverage rate.

Longline catches in the region of the Kerguelen and Crozet islands in the Southern Ocean were provided by the French Muséum National d'Histoire Naturelle, Paris (Guy Duhamel pers. comm.), at a resolution of year and FAO area. Catch data were reported by the Australian Bureau of Agricultural and Resource Economics and Sciences (Heather Chapman, pers. comm.) for the Heard and McDonald Island trawl fishery for 1996−97 to 2014−15, and for the Macquarie Island Patagonian toothfish longline fisheries for 2008−09 to 2014−15.

Combined catches per year by region are summarised in Figure 13.

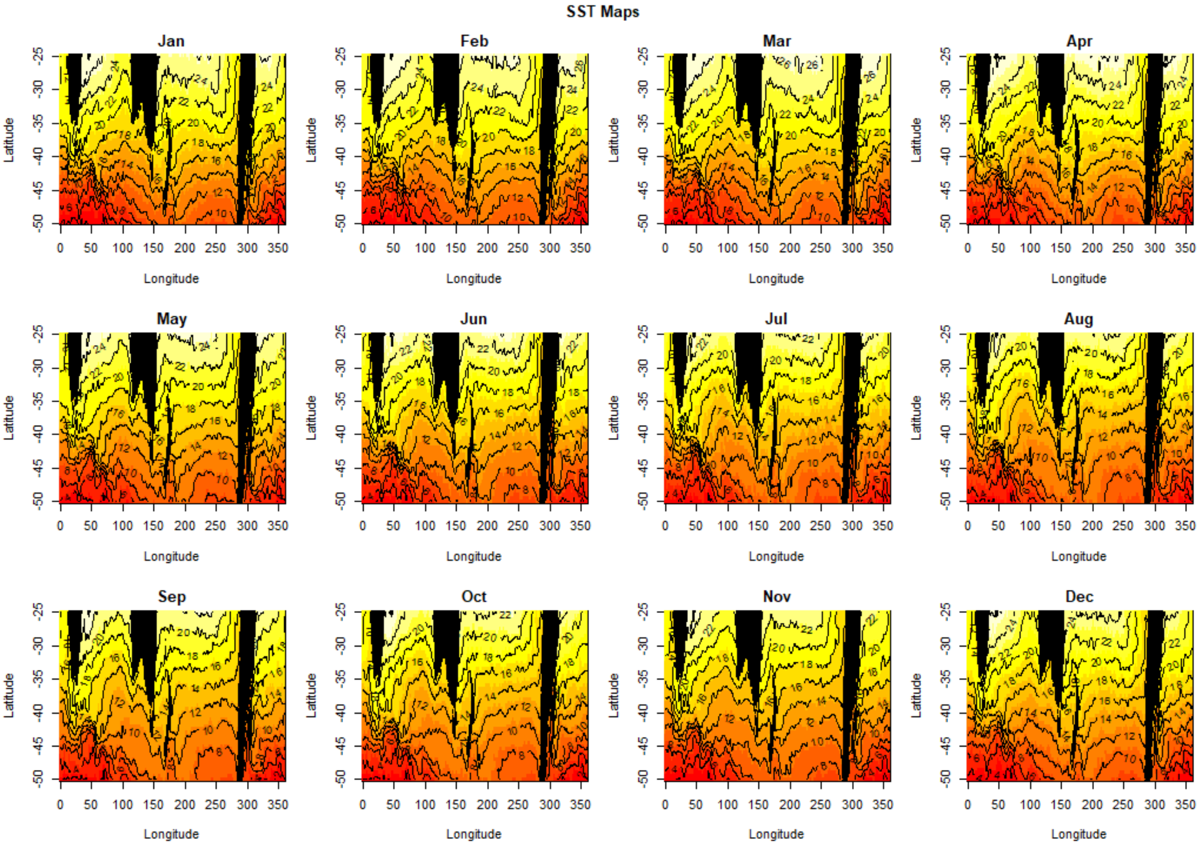

Figure 11: Sea surface temperature patterns by month from CARS data. Yellow represents higher SST, red lower SST, and black is land mass

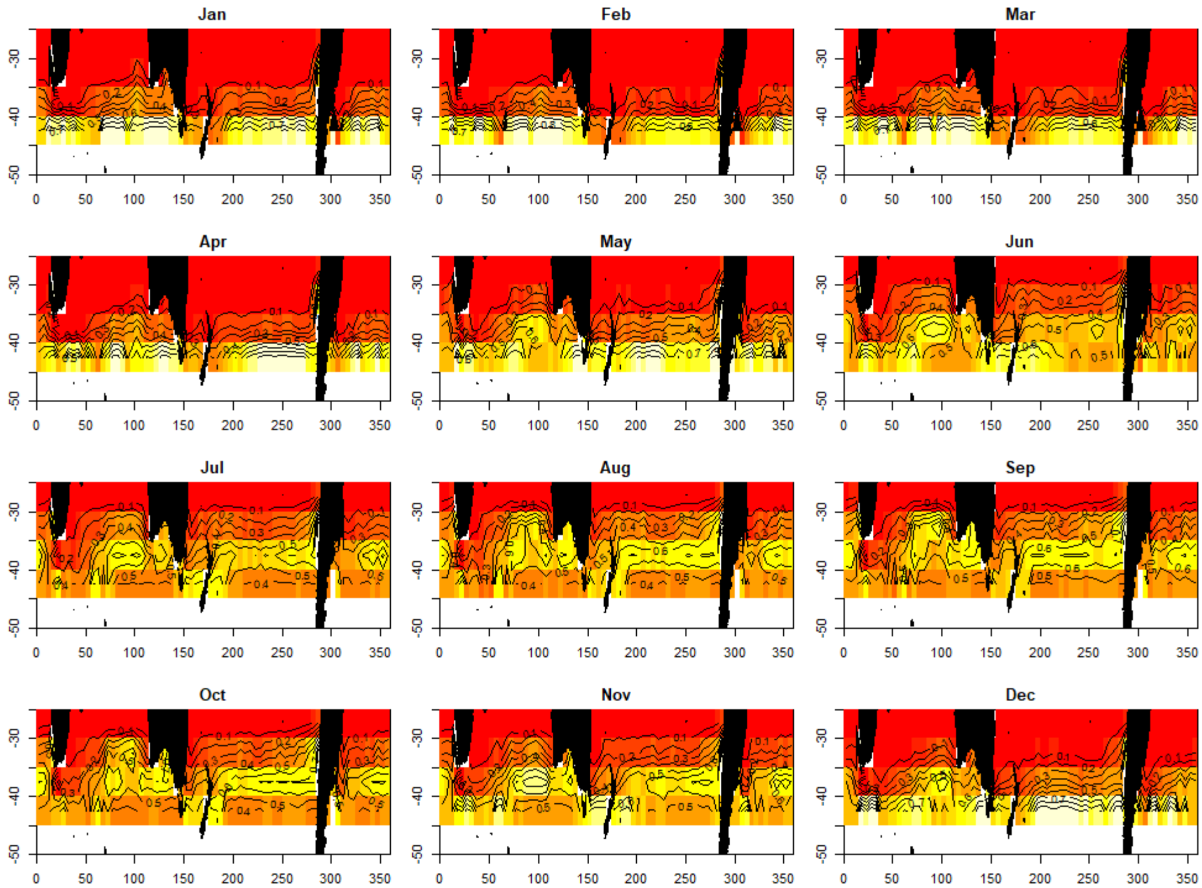

Figure 12: Predicted population relative distribution by month from the abundance prediction model. For each month, the area with highest density is assigned relative density of 1. Yellow indicates higher density, and red lower density. White indicates no information, and black is land mass.

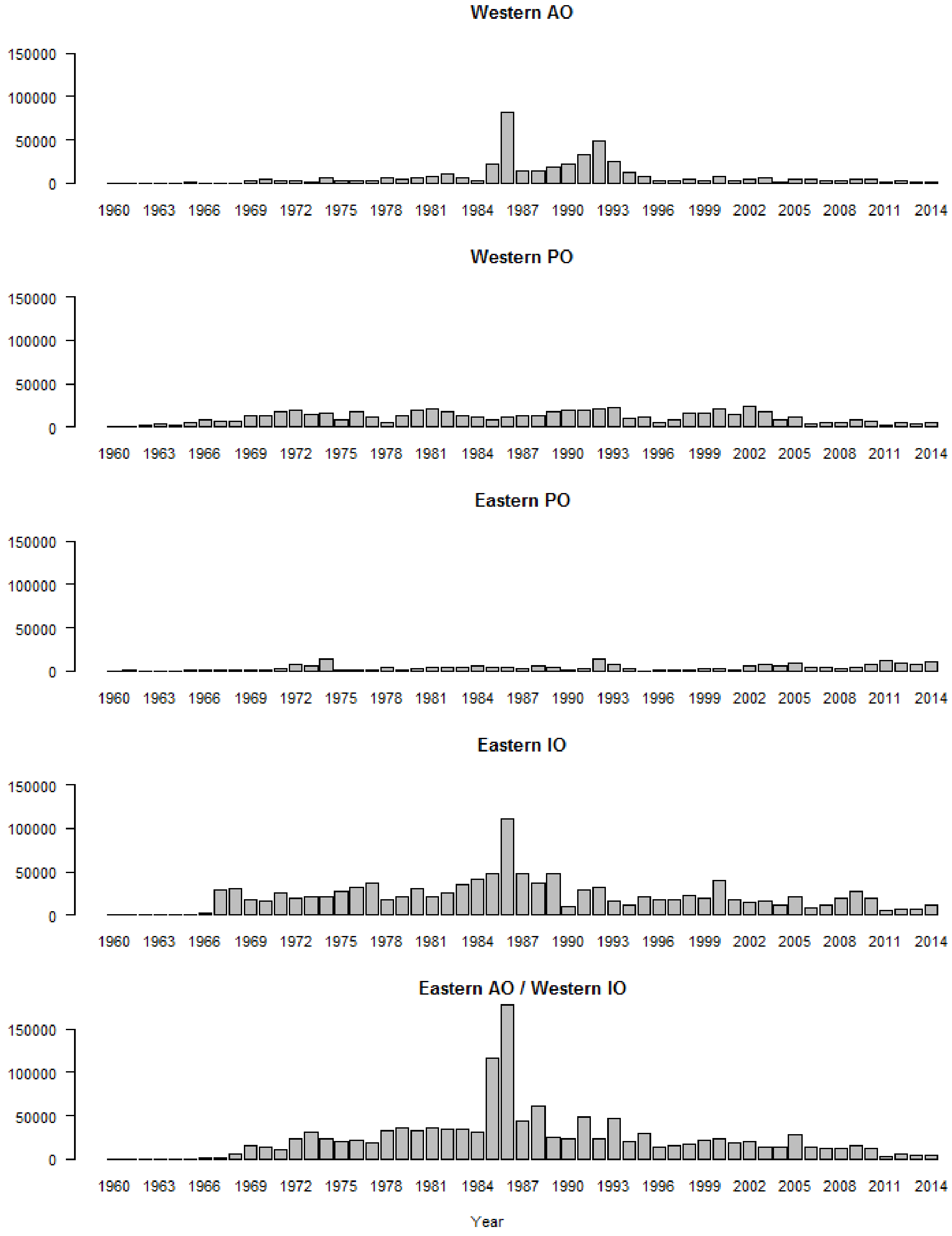

Figure 13: Total catch in number of porbeagle sharks per year by region.

2.5 Indicator analyses

Abundance indices through time were required as inputs into the risk assessment, and to serve as indicators of population trend and condition. The abundance indicators reported here are based on fisheries that operated within each of the five areas, and were taken to be representative of temporal trends in abundance. This contrasts with the estimates of relative abundance in space, estimated in a broad-based analysis across the 190 degrees of longitude for which we had access to Japanese observer data, presented above in Section 2.3.

For some regions, where available, we also present indicators of trends in size and sex ratio.

2.5.1 Eastern Atlantic/Western Indian Ocean

A reanalysis of the Japanese longline data is presented here, using a modification of the method used by Hoyle et al. (2017b) in order to address problems observed with non-normal distributions of the residuals. Most of the methods used remain the same, and apart from some necessary background information, only the approaches that were changed are described here.

Abundance indices for the Eastern Atlantic/Western Indian Ocean, Eastern Indian Ocean, and Western Pacific Ocean were estimated from catch rates in Japanese longline fisheries (Hoyle et al. 2017b). Data were grouped into fishing strategies using cluster analysis of species composition in the logbook data. Observer data and logbook data were linked based on the vessel callsign and set date.

We fitted generalized additive models in R (R Core Team 2017) using the package *mgcv* (Wood 2011). The previous approach used categorical variables for 5 x 5° spatial grids with generalized linear models, and failed to provide estimates for grids in which all observations had zero catch. Using the spatial smoothing available in *mgcv* avoided this problem. The previous approach also fitted indices for separate northern and southern areas in each region, but we combined these to give a single set of indices for each region, with better statistical precision and fewer missing values. Analyses used a delta lognormal approach, first modelling the probability of nonzero catch, and then modelling the distribution of catch rates in the nonzero catches.

$$por \neq 0 \sim yr + qtr + te\big(lon, lat, k = c(7,7)\big) + s(hbf, k = 5) + s(hooks, k = 10) + cl + s(SST, k = 5)$$

$$log\left(\frac{por}{hooks}\right) \sim yr + qtr + te(lon, lat, k = c(7,7)) + s(hbf, k = 5) + cl + s(SST, k = 5)$$

Latitude (*lat*), longitude (*lon*), hooks between floats (*hbf*) and sea surface temperature (*SST*) were modelled as continuous variables. Year (*yr*), quarter (*qtr*), and cluster (*cl*) were modelled as categorical variables. The term '*s*' refers to a one-dimensional thin-plate regression spline smooth, '*te'* refers to a two-dimensional tensor product smooth, and *k* sets the upper limit on the degrees of freedom associated with the smooth.

Residuals were more normally distributed (Figure 14) than those estimated in previous analyses (Hoyle et al. 2017b), which used categorical variables for spatial effects. Indices are variable and do not provide strong evidence of long-term trends (Figure 15).

Size and sex indicators are also available from this fishery (Figure 16).

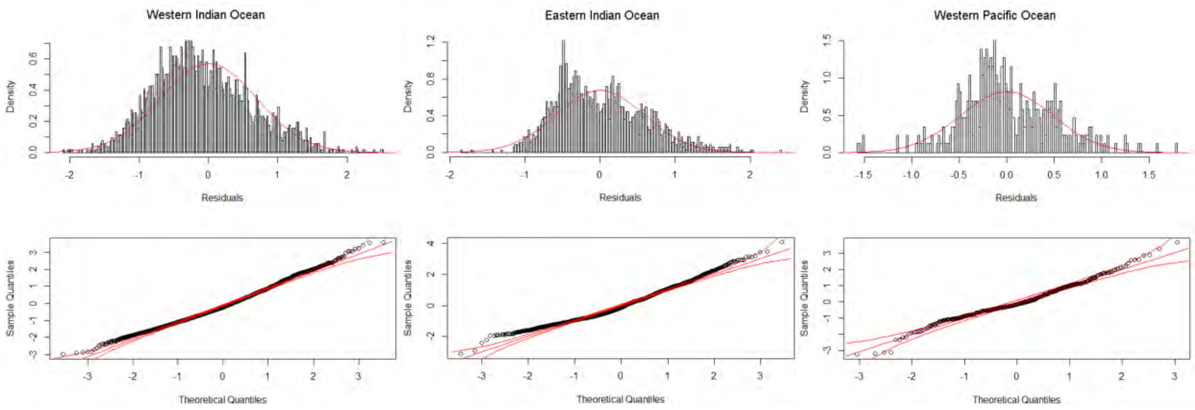

Figure 14: Residual distribution plots for lognormal positive observer data analyses for the Eastern Atlantic/Western Indian Ocean (labelled Western Indian Ocean), the Eastern Indian Ocean, and the Western Pacific Ocean.

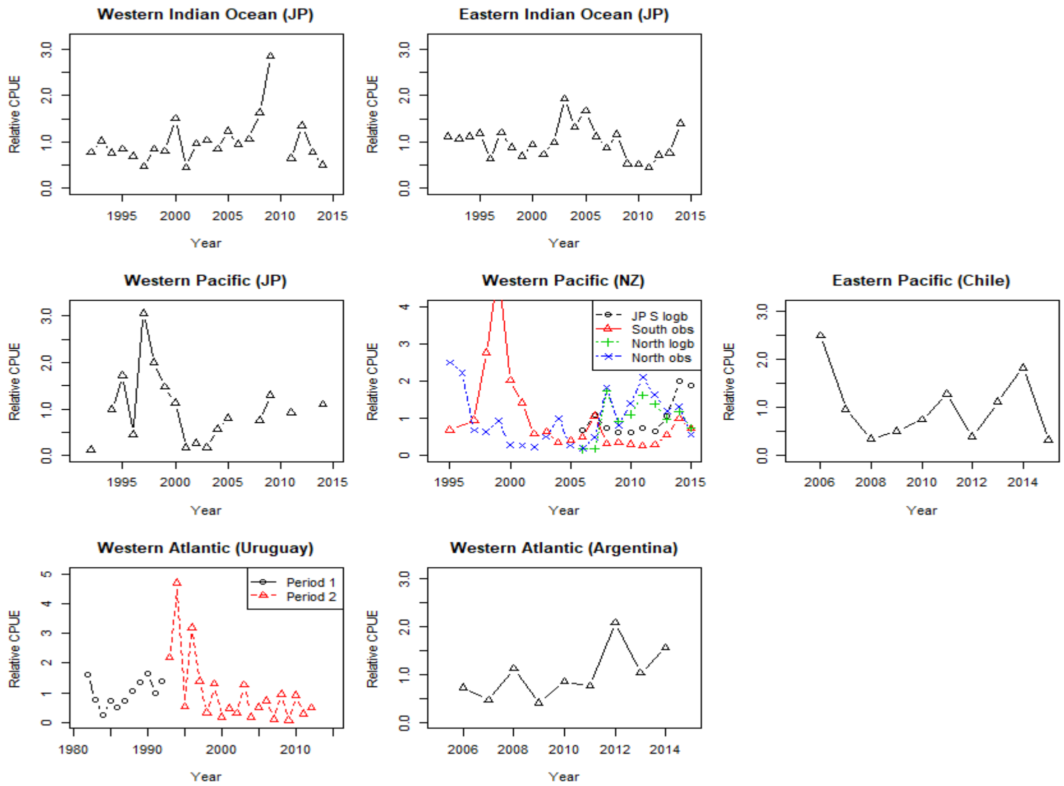

Figure 15: CPUE indices for each region and contributing country. The Eastern Atlantic/Western Indian Ocean region is labelled Western Indian Ocean. Sources: (Cortés et al. 2017, Forselledo et al. 2017, Francis & Large 2017, Hoyle et al. 2017b, Hoyle et al. 2017a).

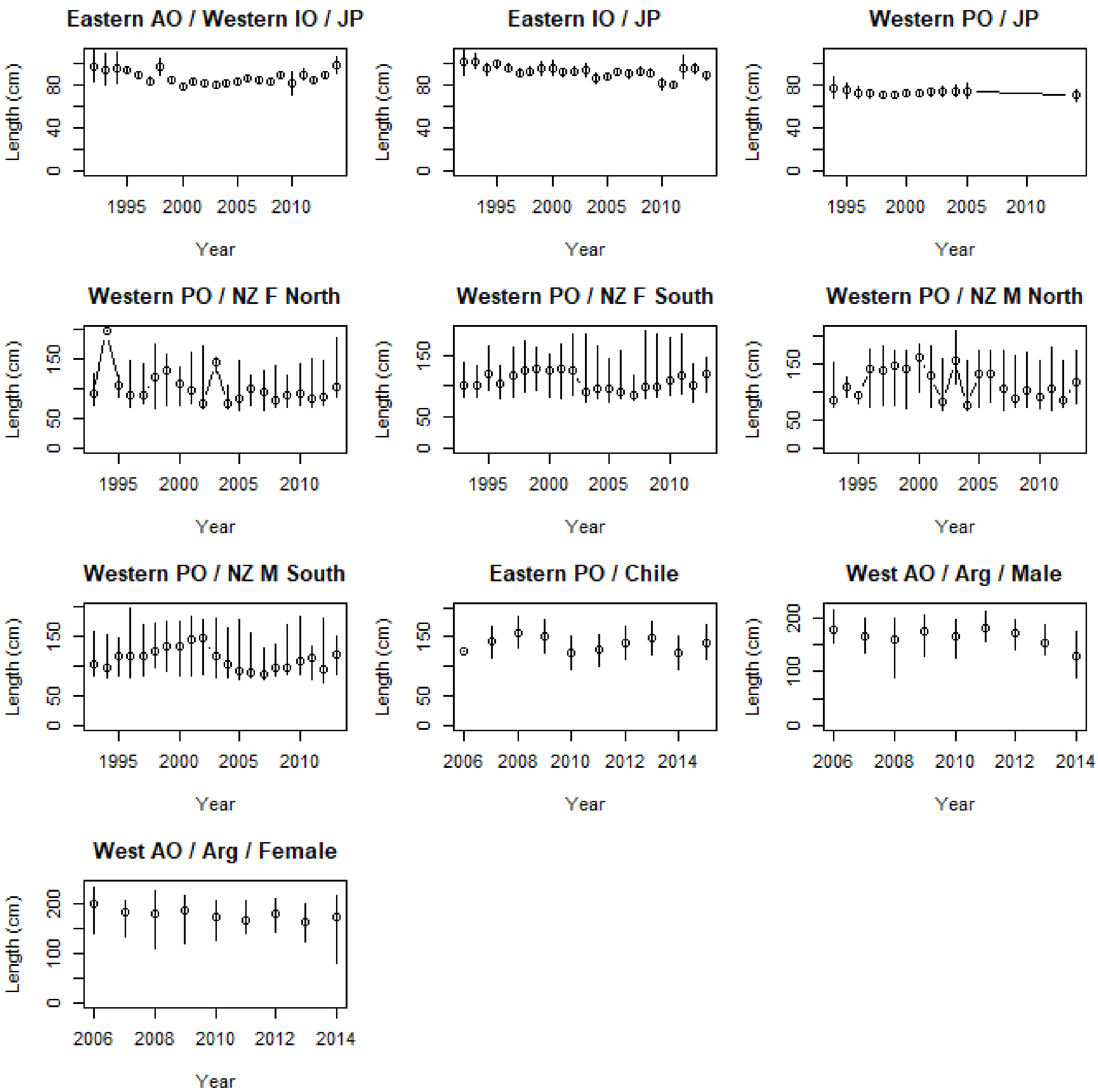

Figure 16: Standardized predictions (Japan (JP) and Chile) and annual measurements (New Zealand (NZ) and Argentina) of lengths in the catch, by region and contributing country. Sources: (Cortés et al. 2017, Francis & Large 2017, Hoyle et al. 2017b, Hoyle et al. 2017a). Size predictions were designed to display trends and may not provide unbiased estimates of median lengths.

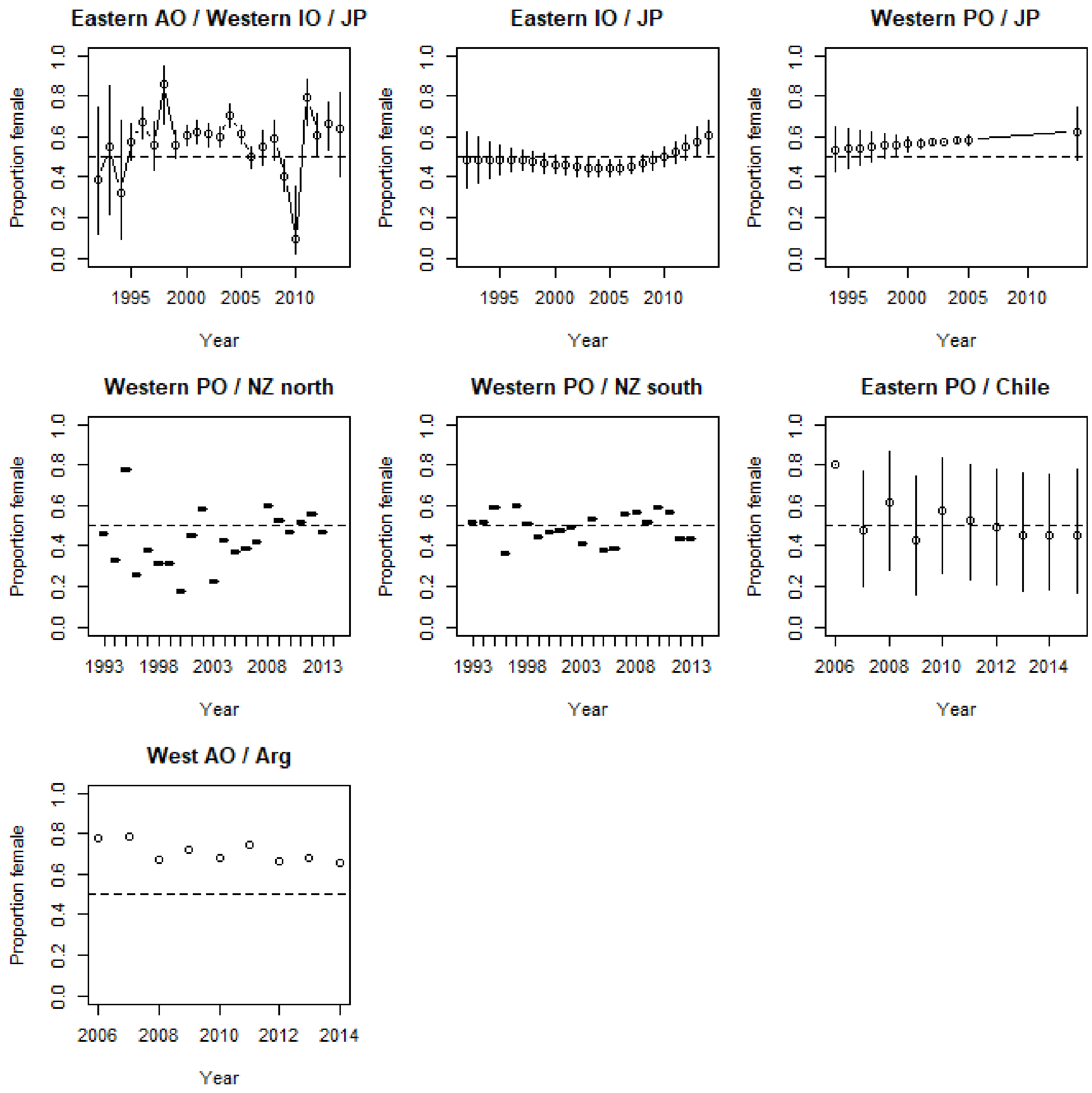

Figure 17: Standardized predictions (Japan (JP) and Chile) and annual measurements (New Zealand (NZ) and Argentina) of proportion female in the catch, by region and contributing country. Sources: (Cortés et al. 2017, Francis & Large 2017, Hoyle et al. 2017b, Hoyle et al. 2017a).

2.5.2 Eastern Indian Ocean

Indices for the Eastern Indian Ocean region were estimated in the same way as those for the Eastern Atlantic/Western Indian Ocean (Figure 15). Indices are variable and do not provide strong evidence of long-term trends.

Size and sex indicators are also available from this fishery (Figures 16 and 17).

2.5.3 Western Pacific Ocean

Two sets of indices were estimated for the Western Pacific. The first set was based on catch rates in the Japanese longline fishery, and was estimated in the same way as those for the Eastern Atlantic/Western Indian Ocean (Hoyle et al. 2017b).

The second set was based on analyses of catch rates in New Zealand longline fisheries. They were reported by Francis & Large (2017) in an update of the results of a previous analysis (Francis et al. 2014). CPUE indices were provided for catches in the Japanese charter tuna longline fishery in southern New Zealand (the Japan South fishery), separately for both logbook data and observer data. There were also indices from observer data in northern New Zealand for both domestic and Japanese charter vessels combined. Indices are shown in Figure 15. Indices are variable and do not provide strong evidence of long-term trends.

Size and sex indicators are also available from this fishery (Figures 16 and 17).

2.5.4 Eastern Pacific Ocean

Indices for the Eastern Pacific Ocean are based on catch rates in the Chilean swordfish fishery, which takes porbeagle sharks as bycatch, particularly in the southern portion of the fishery. The fishery comprises three components, the industrial longline, artisanal longline, and artisanal gillnet. The longline fishery data were combined and analysed to produce indices of abundance (Hoyle et al. 2017a) (Figure 15).

The Chilean index is relatively short and variable, reflecting the fact that data are sparse because porbeagles are only taken at the southern extreme of the swordfish fishery. There is no indication of a temporal trend in these indices.

Size and sex indicators are also available from this fishery (Figures 16 and 17).

2.5.5 Western Atlantic Ocean

Two sets of indices were estimated for the Western Atlantic. The first set was based on catch rates in the Uruguayan longline fishery (Forselledo et al. 2017). This was an update of a previous analysis (Pons & Domingo 2010). There were a number of changes from the 2010 analysis, the most important being breaking the index into two parts (1981−1991 and 1992−2012). Prior to 1992 all participants were large-scale freezer vessels using Japanese-style multifilament longlines (about 2000 hooks per set). From 1992 these vessels were replaced, mostly by small-scale fresh-fishing vessels using American-style monofilament longlines (about 900 hooks per set), and two vessels using Spanish-style multifilament longline. Porbeagle shark catch rates were much lower for vessels using the American style longlines.

Neither the early nor the late Uruguayan indices shows a clear trend through time. The later index has a very low catch rate and is also very variable.

The second set of indices was based on catch rates in the Argentinean surimi fishery (Figure 15). This trawl fishery has comparatively low catch rates, but provides a useful dataset. The fishery is further south than others reported here, and takes relatively large porbeagles, with catch rates increasing further south. Size and sex indicators are also available from this fishery (Figures 16 and 17).

The Argentinian index is short, but appears to show an increasing trend through time. Indices of mean size appear to decline slightly for both males and females. There is also a slight trend toward a lower proportion of females in the sex ratio index.

2.6 Risk assessment

The risk assessment methodology uses the spatial overlap of fishing effort and population density to derive a risk metric. This requires estimation of a catchability coefficient, which is achieved by fitting a logistic production model to available data in the most data-rich of the assessment regions. The catchability scalar is then applied to effort overlap in the other regions to estimate a fishing mortality. The sum of spatially-explicit, annual fishing mortality (annual impact) is compared to a maximum impact sustainable threshold (MIST), which is a limit reference point derived from the intrinsic rate of population growth. Risk is estimated from the ratio of annual impact to the MIST, and expresses the probability, given the uncertainty, that total impacts exceed the MIST.

We used the risk assessment model to evaluate the risk from commercial pelagic longline fisheries to porbeagle shark within a subarea of the Southern Hemisphere (hereinafter referred to as the spatial domain of the risk assessment). This area ranges from 10 °W to 180 °E longitude and from 30 to 60 °S latitude (see Figure 7) and corresponds to the region covered by the Japanese tuna longline fishery (Semba et al. 2013). The risk assessment was restricted to this area as it contained sufficient information to estimate key components of the risk assessment, namely the species distribution (population density) and population catchability. The quantitative risk approach assumes that true population abundance is unknown, but that catch and effort information from an area of highest abundance (and comparatively high data availability) can be used in conjunction with relative density estimates to calibrate a catchability parameter for the assessed population and estimate spatially-explicit relative fishing mortality and total impact from fisheries over the spatial domain of the assessment. Our approach assumes that porbeagle sharks are distributed over the spatial domain of the assessment, comprised of three regions/subpopulations (potentially distinct biological stocks) that undergo limited or no mixing: 1) Eastern Atlantic/Western Indian Ocean; 2) Eastern Indian Ocean; and 3) Western Pacific Ocean (see Section 2.1). We estimated fishing mortality and calculated risk separately for each of the assessment regions, and across the whole spatial domain.

The risk assessment is spatially-explicit and quantitative and distinguishes impact and risk among fishery sectors (i.e., fleet components characterised by differences in operational practice and therefore, different catchability). Annual impacts were estimated over a spatial grid of 5° latitude x 5° longitude, corresponding to the spatial resolution of the catch and effort data available for assessment. The timeframe of the biomass dynamic model (BDM) assessment (see Section 2.6.2) was the period of commercial effort (logsheet) data from 1960 to 2014, though for the risk assessment metrics we give greater weight to the period with better data, starting in 1992. The fishing effort data were modelled as a single fleet (with hooks per set standardised to that of a standard vessel from the Japanese fleet). Only seasonal (year-quarter) variability in species distribution and operational practice (e.g., gear type and targeting strategies) were considered in the assessment. Uncertainties in species distribution within season were not considered. Uncertainty in the catchability parameter and population productivity parameter were estimated and propagated into the evaluation of risk.

2.6.1 Preparation of spatial data

The spatial domain of the assessment covered three regions: Eastern Atlantic/Western Indian Oceans, Eastern Indian Ocean, and Western Pacific Ocean, bounded at 30 °S and 60 °S (Figure 18). The Eastern Atlantic/Western Indian Ocean region is also referred to as the 'calibration region', being the most data-rich. Each region was divided into 5 x 5° grids, and all spatial data were available at this resolution.

Three sources of spatial data were used:

1) The ocean area of each grid g in each region r: A_{rg} accounting for area variation with latitude, and non-ocean (land) area. This was calculated using the R code in Appendix A. The projection was the Lambert azimuthal equal-area projection.
2) The year-invariant relative density of porbeagle shark by grid and quarter q so that $\sum_g D_{rgq} = 1$ for all quarters across regions (see Section 2.3). Because no density estimates were available south of 45 °S, we assumed that the density in the latitude band 45–55 °S was the same as the density in the 40–45 °S band immediately to the north (Figure 18). There is limited information about porbeagle shark populations south of 55 °S, and the assessment makes the conservative assumption that the density south of 55 °S is zero. This assumption represents a relatively pessimistic scenario which, depending on the densities south of 55 °S, may bias the estimate of fishing mortality upwards. Fishing effort is low at these latitudes.
3) The absolute fishing effort between 1960 and 2015, by grid and quarter, summed over all fleets combined (Figure 18). All effort was converted to the number of sets, with each set standardised to 3000 hooks. This is the assumed effort unit for the Japanese CPUE data.

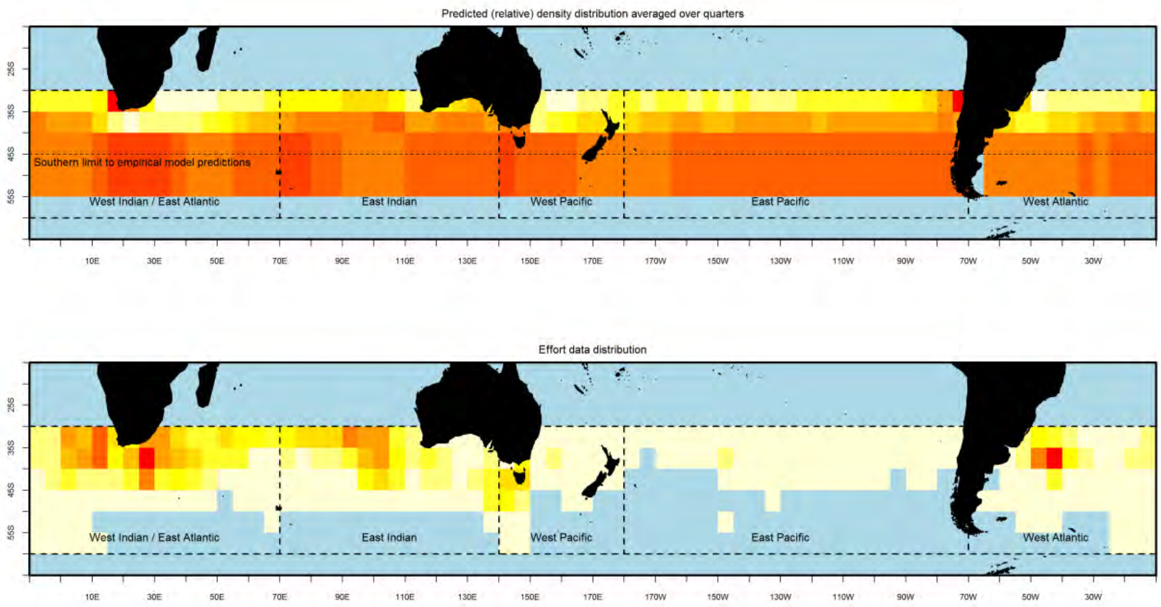

Figure 18. Illustrative map of five regions showing (top) the spatial coverage of relative density data of porbeagle shark, averaged across quarters, and (bottom) the sum of absolute fishing effort between 1960 and 2015. Scale is from red (high) to yellow (low).

2.6.2 Fitting of biomass dynamic model to catch and abundance in the calibration area

Relative abundance indices from the Eastern Atlantic/Western Indian Ocean region (section 2.5.1) were used to calibrate a population catchability parameter for porbeagle shark over the spatial domain of the assessment. To estimate a posterior distribution for the catchability coefficient q, we fitted a biomass dynamic model (BDM) to Japanese observer CPUE index for the calibration region (Eastern Atlantic/Western Indian Ocean region), using a reconstructed catch for the whole region (Section 2.4, Figure 13).

The biomass dynamic, state-space model was implemented using the R package 'bdm' (Edwards 2017) which is written in the Bayesian modelling language Stan (Stan Development Team 2014).

The model takes the form:

$$\mu_{t+1} = x_t + r.x_t.(1 - x_t) - C_t/K$$
$$x_t \sim Log - Normal\left(\ln(\mu_t) - \sigma^2_{[p]}/2\,, \sigma^2_{[p]}\right)$$
$$I_t \sim Log - Normal\left(\ln(q.x_t)\ - \sigma^2_{[o]}/2\,, \sigma^2_{[o]}\right)$$

where x is the unobserved biomass depletion relative to the carrying capacity K, C is the catch, μ is the expected value (i.e. $\mu = E[x]$), I is the observed CPUE index, $\sigma^2_{[p]}$ is process variance, $\sigma^2_{[o]}$ is observation variance, and q is the catchability scalar. Estimated parameters within the model are r, K and q. We assumed a uniform prior on log(K) with a plausible upper bound (based on expert opinion) equivalent to 5 individuals per km^2, which is equivalent to 28 million individuals for the calibration area (i.e. the upper bound on K implies that log(K) < 17.2). This was a relatively arbitrary value designed to be consistent with the K upper limit of 1 per km^2 assumed for the much rarer bigeye thresher (Fu et al. 2016). The prior for the intrinsic growth rate r was derived from life-history data (Table 2) using the approach of McAllister et al. (2001) , in which r is obtained as a solution to the Euler-Lotka equation (giving a prior mean=0.033 and cv=0.55). The catchability q was estimated as a nuisance parameter (i.e. fixed analytically at its maximum likelihood value). The standard error terms were fixed on input at: $\sigma^2_{[p]}$ = 0.05 and $\sigma^2_{[o]}$ = 0.20. The model was run with 4 chains of 2000 samples each. A burn-in period of 1000 samples from each chain was discarded, leaving 4000 samples in total.

The catch and abundance data are shown in Figure 19. Trace outputs from the Monte Carlo Markov Chain (MCMC) model fit, the derived fit to the CPUE index, and the posterior distributions or q, r and log(K) are shown in Figure 20. Convergence of the MCMC chains was inferred from visual inspection of multiple independent chains, which can be seen to mix well and generate overlapping samples from the posterior. No formal statistical measures of convergence were generated, because they are unreliable.

Table 2. Input life history information used to develop a prior for the maximum intrinsic population growth rate (*r*) of porbeagle shark within the spatial domain of the assessment. Maturation, growth and recruitment parameters were based on available information for females only. Parameter values were reviewed by Clarke et al. (2015) and those values were either adopted here (with original sources given) or modified according to the listed source.

Process	Parameter	Value	CV	Reference
Longevity	A_{inf} (yr)	75		Francis et al. (2007)
Maturation	A_{50} (yr)	14.5	0.25	Francis (2015)
	delta	1.5	0.25	
Growth	L_{inf} (cm, FL)	211	0.3	Francis (2015)
	k	0.086	0.3	Francis (2015)
	t_0	-6.1	0.3	Francis (2015)
Length-weight	a	2.14289E-05	0.1	Ayers et al. (2004)
	b	2.924	0.1	Ayers et al. (2004)
Recruitment	α (no.)	3.75		Clarke et al. (2015)
Mortality	M (yr^{-1})	0.09	0.42	Averaged from four empirical equations: 1. Hoenig (1983): $\ln(M)=0.941-0.873 \ln(A_{inf})$; 2. Campana et al. (2001): $M=-\ln 0.01/A_{50}$; 3. Jensen (1996): $M=1.65/A_{50}$; 4. Jensen (1996): $M=1.6k$

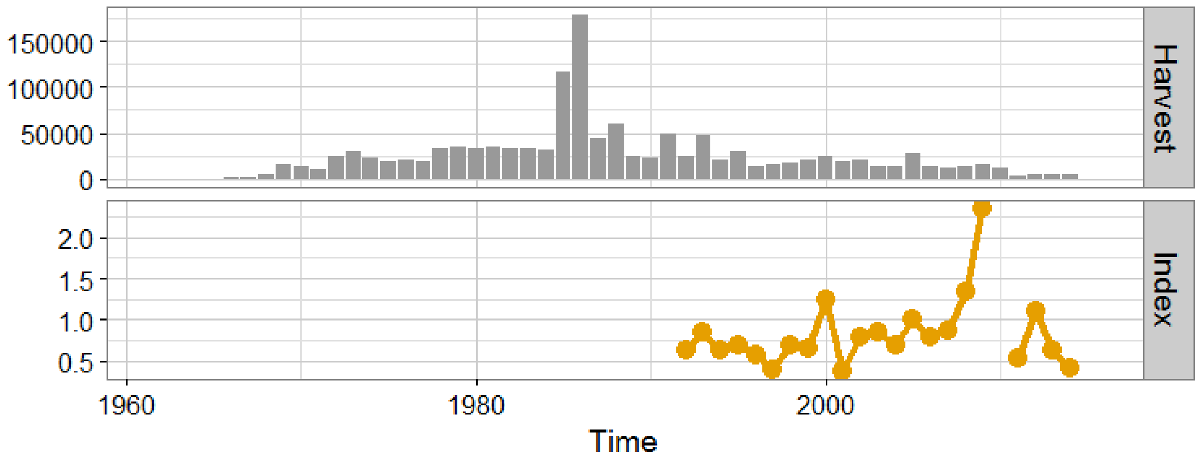

Figure 19. Catch (top) and abundance (bottom) data.

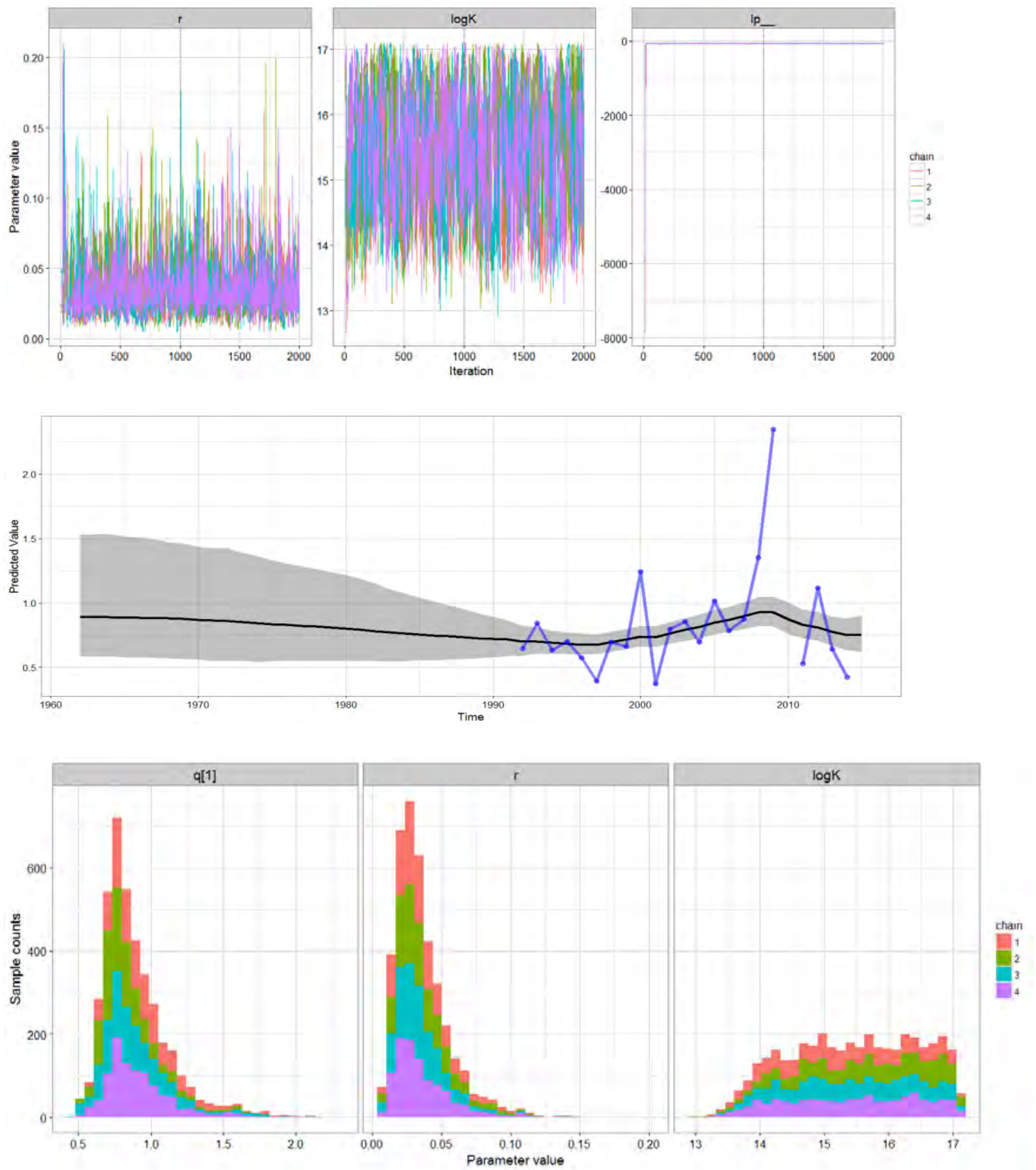

Figure 20. Top: Trace outputs from the MCMC model fit showing estimates of *r* and log*(K)* alongside the log-posterior (*lp__*). Middle: Derived fit to the CPUE index. Bottom: Stacked histograms of MCMC samples, representing the posterior distributions of *q*, *r* and log*(K).*

2.6.3 Estimation of the fishing mortality for each assessment region

We first convert the catchability estimated using the BDM into a catchability that can be used in the risk assessment. For BDM the catch equation is:

$$C = q^{[bdm]} \cdot \frac{N}{K} \cdot E$$

where *N* is the total number in the assessment area, *K* is the carrying capacity (total number), and *E* is the effort in set units; whereas for the fisheries risk assessment (*fra*), the catch is a function of the relative density in numbers per unit area $D = N/A$, with area expressed in km^2, rather than the depletion N/K, and the catch equation is therefore:

$$C = q^{[fra]} \cdot \frac{N}{A} \cdot E$$

Our estimate of the catchability for the risk assessment is therefore:

$$q^{[fra]} = q^{[bdm]} \cdot \frac{A}{K} = q^{[bdm]} \cdot \frac{1}{\widetilde{K}}$$

where $\widetilde{K}$ is the carrying capacity expressed in number per km^2 (i.e. the absolute density at carrying capacity). The area *A* is taken to be a summation across grids with non-zero density (i.e. $A = \sum_g A_g \,|D_g > 0$). We ignore seasonality in our calculation of A because any grid with non-zero density is non-zero for the whole year. This derivation of the catchability is valid for all of the assessment regions, using the approximation that the density at carrying capacity is constant across regions.

To derive a fishing mortality, we first estimate catches by year y, grid g, quarter q and iteration i for each posterior sample of the catchability, where catch C is a function of the relative density D and effort E:

$$C_{ygqi} = q_i^{[fra]} \cdot D_{gq} \cdot E_{ygq}$$

We then estimate total numbers for each year, grid and quarter as the product of the relative density and the area A_g in km^2 for each grid:

$$N_{gq} = D_{gq} \cdot A_g$$

To estimate the harvest rate U using the risk assessment methodology, we calculate:

$$U_{yqi} = \frac{\sum_g C_{ygqi}}{\sum_g N_{gq}}$$

Assuming an exponential model of instantaneous mortality we can then write the fishing mortality as:

$$F_{yqi} = -\ln(1 - U_{yqi})$$

Finally, we calculate the fishing mortality rate (impact) per year by taking the sum across quarters:

$$F_{yi} = \sum_q F_{yqi}$$

Annual fishing mortalities are shown in Figure 21 and Appendix B. Annual *F* values were greatest in Eastern Atlantic/Western Indian Ocean, slightly lower in the Eastern Indian Ocean, and lowest in the

Western Pacific Ocean. Annual median F decreased from the mid-1980s to 2014 in both the Western Indian Ocean/Eastern Atlantic Ocean and Eastern Indian Ocean regions. In the assessment area (three regions combined) in the last decade (2005 to 2014), median F values ranged from 0.0008 to 0.0015 (mean 0.0010). Higher F values in the Western Indian Ocean/Eastern Atlantic Ocean and Western Pacific Ocean regions were consistently associated with the second quarter (April–June months or austral autumn season) (quarterly F estimates not shown). In contrast, the third and fourth quarters (July–September (austral winter) and October-December (austral spring)) contributed higher F values in the Eastern Indian Ocean region, however with a shift to higher F values in the second and third quarter (and a drop in fourth quarter F) over the recent period (2010-2014).

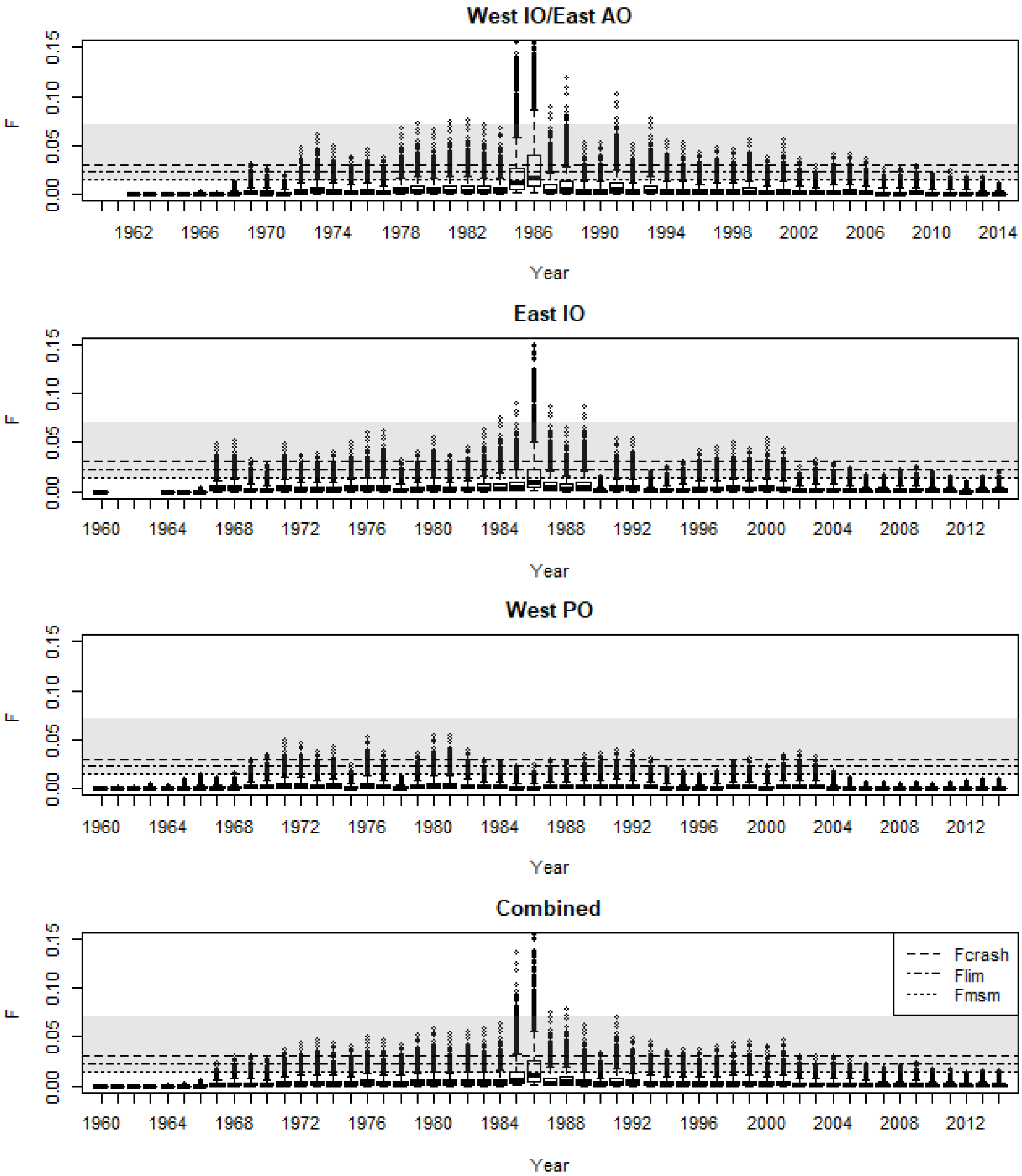

Figure 21. Estimated fishing mortalities by year and region. The horizontal dotted lines indicate the median MIST values (F_{crash}, F_{lim}, and F_{msm}), but note that the MIST varies among realisations of the model, with the grey band showing the 90% distribution for F_{crash}. The boxes show the interquartile range of the F estimates, with a line at the median of each. The whiskers extend up to 1.5 x the interquartile range. Dots mark points beyond the whiskers.

2.6.4 Estimation of risk

The risk metric R is calculated as the ratio between impact (sum of spatially-explicit relative *F* estimates) and our limit reference points for the stock (the MIST values). We reported against three MIST values, as described by Clarke and Hoyle (2014) (based on Zhou & Griffiths 2008, Zhou et al. 2011): F_{crash}, which is the instantaneous fishing mortality that will in theory lead to population extinction; F_{lim}, the instantaneous fishing mortality rate that corresponds to the limit biomass B_{lim} (where B_{lim} is assumed to be half of the biomass that supports a maximum sustainable fishing mortality); and F_{msm}, the instantaneous fishing mortality rate that corresponds to the maximum number of fish in the population that can be killed by fishing in the long term. F_{crash} was set equal to *r*, F_{lim} to 3*r*/4; and F_{msm} to *r*/2. The values of *r* were obtained from the BDM fit. For each of these limit reference points (*LRP*), the risk metric is then:

$$R = \frac{Impact}{MIST} = \frac{F}{F^{[LRP]}}$$

which was calculated for each year and iteration of the full posterior distributions of both *F* and *r*

Risk values are shown both as *F*-ratios and the probabilities that *F* exceeds the MIST in Figures 22 and 23, and for the period from 1992 onwards (the first year of Japanese CPUE data) in Tables 3 and 4. *F*-ratios for the assessment area declined by half from a mean for the F_{crash} MIST of 0.068 (range 0.051–0.088) in 1992–2005, to a mean of 0.032 (range 0.023–0.042) in 2006–2014 (Table 3). For the F_{lim} MIST the equivalent numbers were 0.090 (range 0.068–0.118) in 1992–2005, to a mean of 0.043 (range 0.031–0.056) in 2006–2014 (Table 3). For the F_{msm} MIST the *F*-ratios were 0.135 (range 0.102–0.176) in 1992–2005, to a mean of 0.063 (range 0.046–0.083) in 2006–2014 (Table 3).

The probability of *F* exceeding the F_{crash} MIST decreased by 95% from a mean of 0.0084 (range 0.0015–0.0205) in 1992–2005, to a mean of 0.0004 (range 0.0000–0.0013) in 2006–2014 (Table 4). The probability of *F* exceeding the F_{lim} MIST decreased by 95% from a mean of 0.0183 (range 0.0073–0.0358) in 1992–2005, to a mean of 0.0016 (range 0.0005–0.0040) in 2006–2014 (Table 4). The probability of *F* exceeding the F_{msm} MIST decreased by 95% from a mean of 0.0452 (range 0.0213–0.0778) in 1992–2005, to a mean of 0.0066 (range 0.0023–0.0133) in 2006–2014 (Table 4).

2.6.5 Contributions to fishing mortality

We estimated the proportional contributions to fishing mortality of each fishery by year in each region, and for the different fishing strategies within the pelagic longline fishery. Fishing strategies (which we have also called 'fisheries') were determined based on cluster analysis of species composition data in the Japanese fleet (see Hoyle et al. (2017b) for a description of the approach). For each grid cell by year-quarter stratum, the Japanese longliners' proportional allocations of effort by cluster were noted, and assigned to all effort in that stratum. For effort records in strata that included no Japanese effort, the same approach was applied at coarser stratification: latitude (within region) by year-quarter. For the few records still not assigned, the same approach was applied with stratification at latitude (within region) by quarter.

Predicted longline catch was then calculated by region, year, and cluster by summing across grid cells and quarters. Other catch types available as total catch were then added to the dataset. Proportional contributions to *F* were calculated as catch(fishery, year) / total catch(year). Please note that uncertainties in predicted catches can significantly affect these estimates of proportional contributions to *F*, which should be regarded as approximate and indicative.

The greatest contributions to *F* were made by the pelagic longline fisheries, with the largest contribution by the SBT fishery and the mixed ALB/SBT fisheries (Figure 24). In the last 10 years, those fisheries contributed about 75–80% of the fishing mortality in the Western Indian Ocean/Eastern Atlantic Ocean, 70–90% in the Eastern Indian Ocean, and 70–85% in the Western Pacific Ocean. The contribution of ALB/SBT increased significantly from about 2004 in the Western Indian / Eastern Atlantic Ocean region, and from 2005 in the Eastern Indian Ocean region, but has been consistently important in the Western Pacific since 1996. The more northern longline fishery that takes albacore, bigeye and yellowfin tunas made a generally smaller but still substantial contribution in recent years. In the Western Pacific region, the New Zealand midwater trawl fishery has also contributed to the fishing mortality.

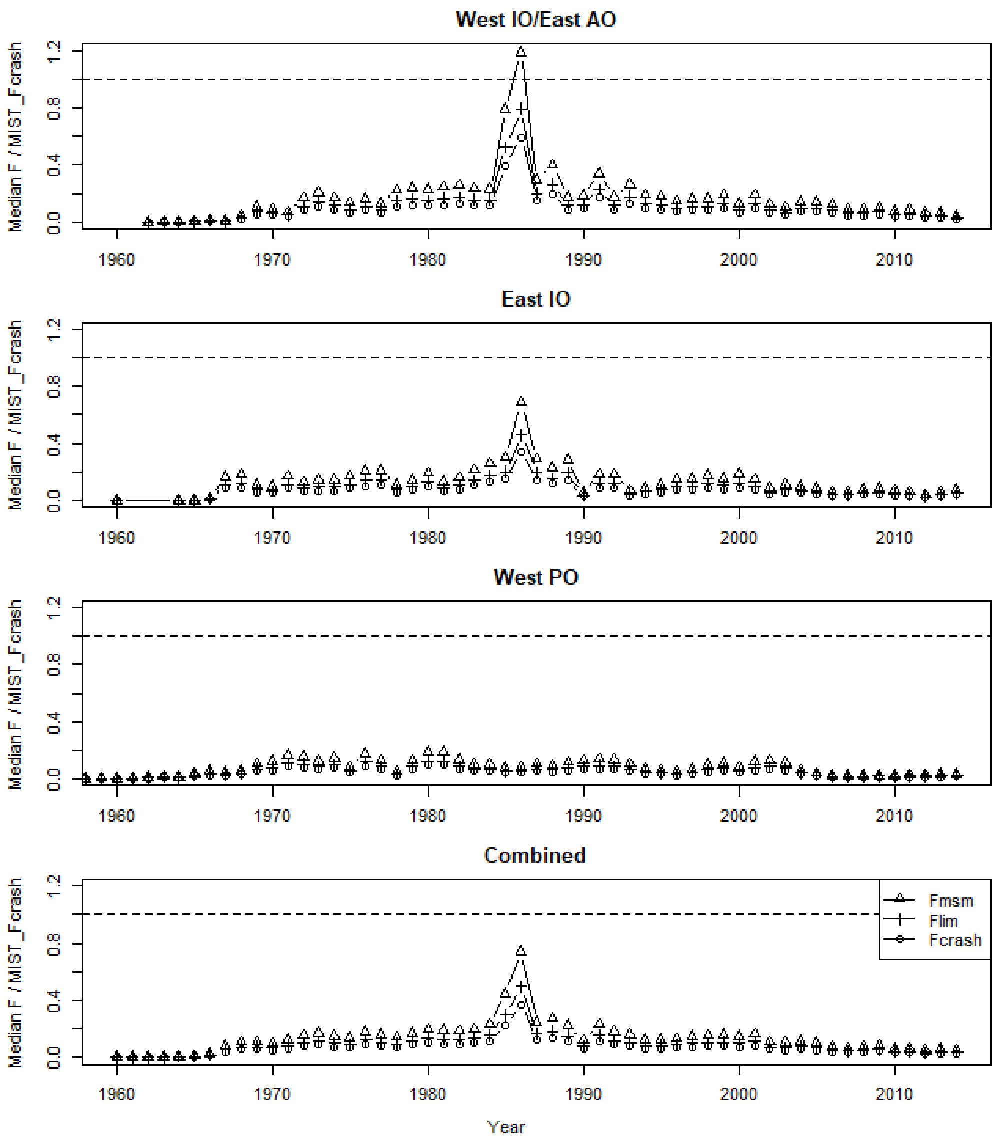

Figure 22: F-ratio plots showing the median values of *F* / MIST by year, for the three versions of the MIST (F_{crash}, F_{lim}, and F_{crash}), for the three regions separately and combined (the assessment area). Note that the *F*-ratio is almost always below 1, indicated by the horizontal dotted line.

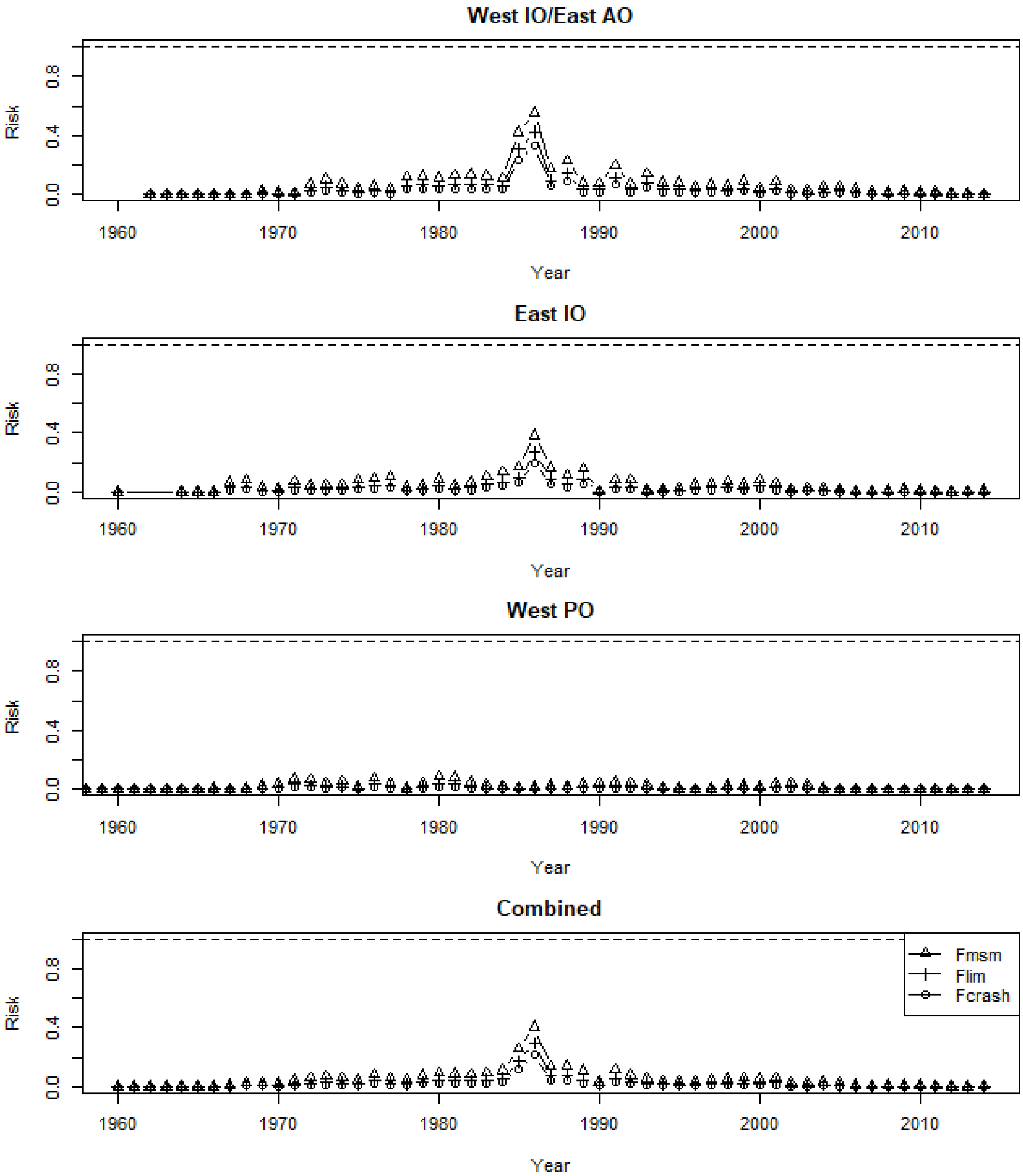

Figure 23: Risk plots showing the probability that *F* exceeds the MIST by year, for the three versions of the MIST (F_{crash}, F_{lim}, and F_{crash}), for the three regions separately and combined (the assessment area).

Table 3a: *F*-ratio$_{crash}$ metric for the impact of pelagic longline fisheries on porbeagle shark in three regions, and in the combined regions (the assessment area). The *F*-ratio$_{crash}$ metric is the ratio of *F* to the F_{crash} limit reference point (MIST), rounded to three decimal places. 95% confidence intervals for these values are shown in Appendix C.

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	0.088	0.090	0.067	0.088
1993	0.132	0.036	0.053	0.080
1994	0.095	0.045	0.037	0.061
1995	0.090	0.055	0.032	0.063
1996	0.074	0.073	0.024	0.064
1997	0.082	0.077	0.033	0.073
1998	0.081	0.086	0.049	0.075
1999	0.094	0.078	0.054	0.079
2000	0.065	0.092	0.041	0.073
2001	0.095	0.074	0.061	0.081
2002	0.063	0.043	0.065	0.053
2003	0.050	0.055	0.056	0.051
2004	0.071	0.051	0.031	0.055
2005	0.070	0.043	0.021	0.051
2006	0.062	0.030	0.011	0.037
2007	0.045	0.031	0.012	0.031
2008	0.046	0.039	0.011	0.038
2009	0.052	0.044	0.013	0.042
2010	0.037	0.035	0.011	0.030
2011	0.042	0.029	0.013	0.030
2012	0.033	0.019	0.015	0.023
2013	0.032	0.029	0.017	0.028
2014	0.022	0.038	0.017	0.026

Table 3b: F-ratio$_{lim}$ metric for the impact of pelagic longline fisheries on porbeagle shark in three regions, and in the combined regions (the assessment area). The F-ratio$_{lim}$ metric is the ratio of F to the F_{lim} limit reference point (MIST), rounded to three decimal places. 95% confidence intervals for these values are shown in Appendix C.

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	0.117	0.120	0.089	0.118
1993	0.176	0.048	0.070	0.106
1994	0.127	0.060	0.049	0.081
1995	0.120	0.073	0.042	0.084
1996	0.098	0.098	0.033	0.085
1997	0.109	0.103	0.044	0.097
1998	0.107	0.115	0.065	0.100
1999	0.125	0.104	0.072	0.105
2000	0.087	0.123	0.055	0.097
2001	0.126	0.098	0.081	0.108
2002	0.084	0.058	0.087	0.071
2003	0.067	0.073	0.075	0.069
2004	0.095	0.067	0.041	0.074
2005	0.094	0.058	0.028	0.068
2006	0.083	0.040	0.015	0.050
2007	0.060	0.041	0.016	0.042
2008	0.061	0.052	0.014	0.051
2009	0.069	0.059	0.017	0.056
2010	0.050	0.047	0.014	0.040
2011	0.056	0.038	0.018	0.040
2012	0.044	0.025	0.020	0.031
2013	0.043	0.039	0.022	0.038
2014	0.029	0.050	0.023	0.035

Table 3c: *F*-ratio$_{msm}$ metric for the impact of pelagic longline fisheries on porbeagle shark in three regions, and in the combined regions (the assessment area). The *F*-ratio$_{msm}$ metric is the ratio of *F* to the F_{msm} limit reference point (MIST), rounded to three decimal places. 95% confidence intervals for these values are shown in Appendix C.

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	0.176	0.180	0.134	0.176
1993	0.263	0.072	0.105	0.159
1994	0.190	0.089	0.074	0.122
1995	0.180	0.109	0.063	0.125
1996	0.147	0.146	0.049	0.128
1997	0.163	0.154	0.067	0.146
1998	0.161	0.173	0.098	0.150
1999	0.188	0.156	0.109	0.157
2000	0.130	0.184	0.083	0.146
2001	0.189	0.148	0.121	0.162
2002	0.126	0.087	0.130	0.106
2003	0.100	0.110	0.112	0.103
2004	0.142	0.101	0.062	0.111
2005	0.141	0.087	0.042	0.102
2006	0.124	0.060	0.022	0.075
2007	0.089	0.061	0.024	0.062
2008	0.092	0.078	0.021	0.077
2009	0.103	0.088	0.025	0.083
2010	0.075	0.070	0.021	0.059
2011	0.083	0.058	0.027	0.060
2012	0.066	0.038	0.031	0.046
2013	0.064	0.059	0.033	0.056
2014	0.044	0.076	0.035	0.053

Table 4a: F_{crash} risk metric for the impact of pelagic longline fisheries on porbeagle shark in three regions, and in the combined regions (the assessment area). The F_{crash} risk metric is the probability that F is greater than the F_{crash} limit reference point, rounded to four decimal places.

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	0.0170	0.0190	0.0060	0.0205
1993	0.0450	0.0013	0.0038	0.0115
1994	0.0195	0.0018	0.0008	0.0065
1995	0.0198	0.0043	0.0003	0.0063
1996	0.0110	0.0100	0.0000	0.0070
1997	0.0130	0.0115	0.0008	0.0090
1998	0.0130	0.0160	0.0020	0.0095
1999	0.0208	0.0108	0.0033	0.0118
2000	0.0088	0.0218	0.0015	0.0090
2001	0.0248	0.0120	0.0075	0.0148
2002	0.0055	0.0010	0.0075	0.0018
2003	0.0025	0.0045	0.0035	0.0035
2004	0.0090	0.0035	0.0003	0.0045
2005	0.0105	0.0018	0.0000	0.0015
2006	0.0053	0.0008	0.0000	0.0008
2007	0.0025	0.0003	0.0003	0.0003
2008	0.0010	0.0008	0.0000	0.0008
2009	0.0033	0.0025	0.0000	0.0013
2010	0.0005	0.0008	0.0000	0.0000
2011	0.0013	0.0003	0.0000	0.0000
2012	0.0000	0.0000	0.0000	0.0003
2013	0.0003	0.0000	0.0000	0.0003
2014	0.0000	0.0013	0.0000	0.0000

Table 4b: F_{lim} risk metric for the impact of pelagic longline fisheries on porbeagle shark in three regions, and in the combined regions (the assessment area). The F_{lim} risk metric is the probability that F is greater than the F_{lim} limit reference point, rounded to four decimal places.

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	0.0335	0.0358	0.0158	0.0358
1993	0.0778	0.0023	0.0098	0.0238
1994	0.0400	0.0063	0.0030	0.0165
1995	0.0378	0.0083	0.0010	0.0135
1996	0.0225	0.0225	0.0000	0.0145
1997	0.0320	0.0275	0.0030	0.0223
1998	0.0270	0.0303	0.0088	0.0243
1999	0.0425	0.0233	0.0083	0.0238
2000	0.0175	0.0395	0.0045	0.0223
2001	0.0398	0.0283	0.0153	0.0270
2002	0.0123	0.0050	0.0175	0.0083
2003	0.0083	0.0103	0.0085	0.0073
2004	0.0200	0.0098	0.0010	0.0095
2005	0.0208	0.0055	0.0000	0.0075
2006	0.0143	0.0015	0.0000	0.0018
2007	0.0058	0.0010	0.0003	0.0013
2008	0.0060	0.0028	0.0000	0.0040
2009	0.0078	0.0038	0.0000	0.0030
2010	0.0035	0.0030	0.0000	0.0013
2011	0.0035	0.0013	0.0000	0.0005
2012	0.0020	0.0003	0.0000	0.0008
2013	0.0018	0.0008	0.0000	0.0005
2014	0.0003	0.0035	0.0000	0.0008

Table 4c: F_{msm} risk metric for the impact of pelagic longline fisheries on porbeagle shark in three regions, and in the combined regions (the assessment area). The F_{msm} risk metric is the probability that F is greater than the F_{msm} limit reference point, rounded to four decimal places

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	0.0713	0.0835	0.0433	0.0778
1993	0.1443	0.0105	0.0280	0.0573
1994	0.0838	0.0180	0.0098	0.0368
1995	0.0828	0.0288	0.0068	0.0398
1996	0.0545	0.0570	0.0015	0.0390
1997	0.0720	0.0598	0.0080	0.0503
1998	0.0600	0.0688	0.0258	0.0540
1999	0.0900	0.0535	0.0248	0.0588
2000	0.0458	0.0803	0.0140	0.0560
2001	0.0865	0.0598	0.0378	0.0608
2002	0.0345	0.0158	0.0415	0.0213
2003	0.0243	0.0265	0.0270	0.0230
2004	0.0565	0.0290	0.0060	0.0303
2005	0.0553	0.0153	0.0005	0.0273
2006	0.0375	0.0055	0.0000	0.0093
2007	0.0215	0.0068	0.0008	0.0058
2008	0.0188	0.0118	0.0000	0.0093
2009	0.0250	0.0195	0.0000	0.0133
2010	0.0133	0.0125	0.0000	0.0080
2011	0.0138	0.0035	0.0000	0.0058
2012	0.0065	0.0018	0.0000	0.0023
2013	0.0068	0.0053	0.0000	0.0025
2014	0.0020	0.0105	0.0005	0.0028

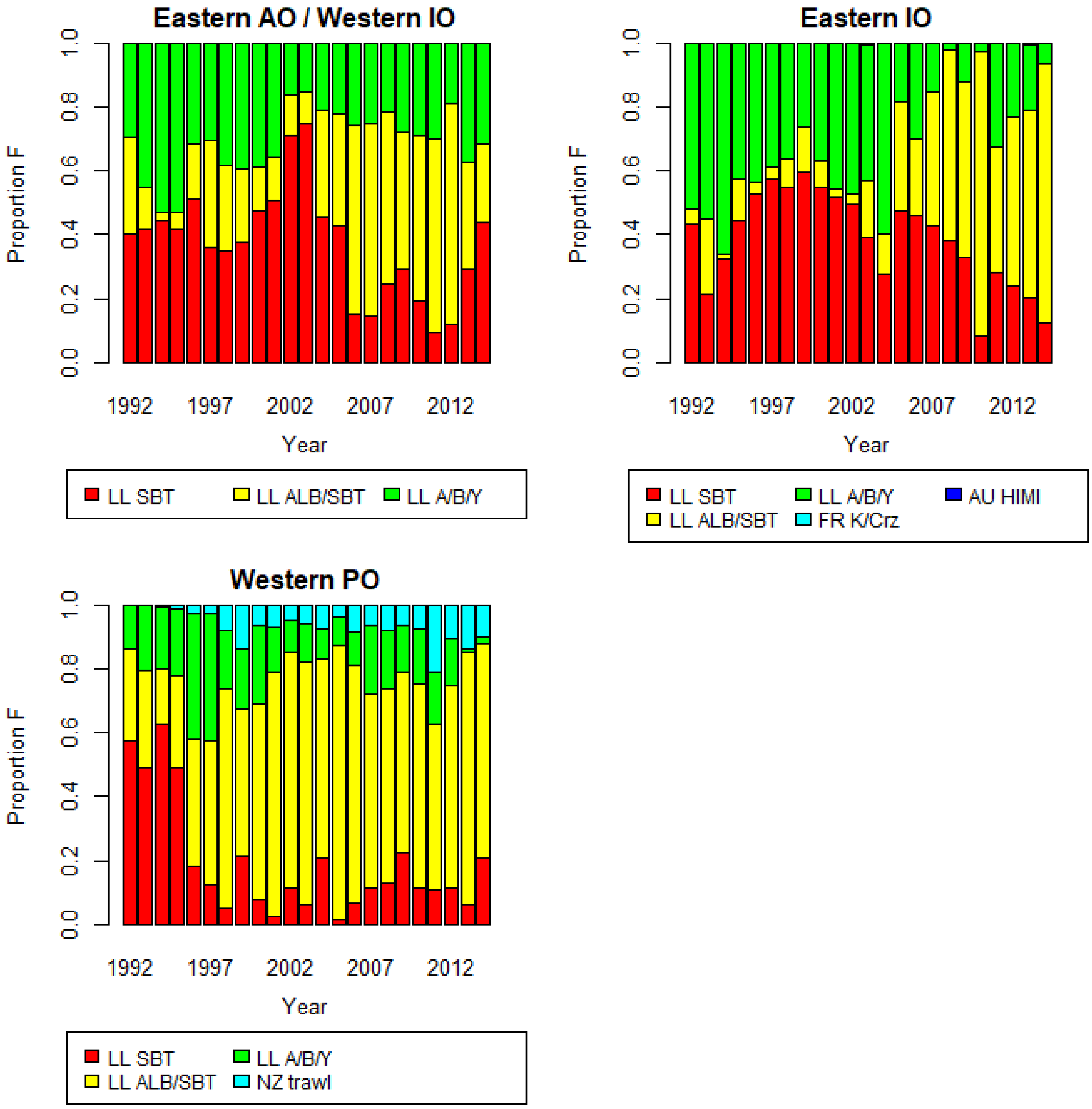

Figure 24: Proportional contributions by year and region to fishing mortality, for the fisheries operating in each region. The pelagic longline (LL) effort is broken into three fishing strategies based on cluster analysis (southern bluefin tuna (SBT), albacore and SBT (ALB/SBT), and ALB, bigeye, and yellowfin tuna (A/B/Y), see Hoyle et al. 2017b). Note that the French (Kerguelen and Crozet Islands, FR K/Crz) and Australian (Heard Island and McDonald Islands, AU HIMI) sub-Antarctic fisheries' contributions to *F* are too small to be visible on the plots.

2.7 Quantitative stock assessment

We explored the potential to develop a quantitative stock assessment model using Stock Synthesis (Methot & Wetzel 2013) for the porbeagle shark stock. Age structured approaches have the advantage that they may provide more reliable inferences than simpler biomass dynamic models by using more data types and modelling important processes that biomass dynamic models cannot represent. For example, when selectivity changes through time it changes the productivity of the population. Representing such changes is difficult with a biomass dynamic model, and failing to represent them often introduces bias (Wang et al. 2014). However, age structured models have the disadvantages that they require more data than biomass dynamic models, and are time-consuming to develop.

The porbeagle shark stock has significant spatial size structure, with smaller sharks observed in warmer waters in the north and larger sharks further south. Furthermore, the northern areas with smaller sharks experience more fishing effort than the rest of the distribution, so selectivity is expected to be biased towards smaller sharks. There is potential to estimate selectivity parameters, because information on catch at size is available from observer data on size (and sex) in the Japanese, New Zealand, Argentinian, and Chilean fisheries. There is also information on relationships between capture size in longline fisheries and SST for the Japanese fishery (Hoyle et al. 2017b), which could be used to impute the size structure taken by all longline fisheries, which together dominate the catch.

On the other hand, the spatial distribution of fishing effort has been relatively stable through time, so it is likely that changes in selectivity through time have been relatively small. It is thus anticipated that this source of potential bias is not a major influence on the results of the assessment.

The decision to structure the spatial domain of the risk assessment model as multiple regions (rather than a single region), implies the need for a multi-regional age structured model. This additional work on the age structured model proved impractical in this study, however, it could be valuable in future to explore alternative hypotheses regarding fishery selectivity and population structure.

3 Discussion

This risk assessment for porbeagle shark in the Southern Hemisphere treats the population as five separate subpopulations, and provides estimates of the risk of exceeding the MIST limit reference point for the three subpopulations with sufficient data. Risk was also estimated for the three together. The remaining two subpopulations have less data, and their status has been interpreted based on relative abundance indices.

Results indicate low fishing mortality rates in the three regions comprising the assessment area, and low risk from commercial pelagic longline fisheries to porbeagle shark over the spatial domain of the assessment. These results are consistent with the trends observed in catch rate indicators over the entire Southern Hemisphere range of the porbeagle shark population, which in most cases show stable or increasing catch rates. Concern has previously been expressed about reduced catch rates in the Uruguay longline fishery after 1993 (International Commission for the Conservation of Atlantic Tunas 2010), but the re-analysis undertaken in collaboration with Uruguayan researchers indicates that in 1993 both the vessels fishing and their fishing methods changed almost completely (Forselledo et al. 2017). After allowing for this change, a decline was no longer evident.

Most catch rate indicators were relatively short, variable, and uncertain, with the majority either stable or increasing. Length indicators were also variable. Only the Argentinian size and sex indicators showed temporal trends, with a small decline in sizes for both sexes, and a slight trend towards less female bias in the sex ratio index.

The indicator analyses, in addition to providing time series to monitor population change, revealed spatial patterns in size and sex distributions, and relationships with environmental variables. Such analyses are critical inputs to stock status assessments, because they help to determine model structure.

The catch rate indicators are by far the most important inputs to this status assessment, and their reliability determines the reliability of the assessment. Stable or increasing observed population trends, under fishing pressure, constrain the risk assessment model to estimate levels of catchability and population density that would allow the population to be stable or increasing. Thus the indicator trend in the calibration area is the most important factor determining the relatively low estimate of risk. Continued data collection by observers will improve the time series and provide better evidence about abundance trends. Maintaining collection and analysis of indicators from observer data is a key recommendation from this project.

Furthermore, the population catchability was calibrated assuming that capture mortality was 100% (i.e., zero post-release survival). In recent years many fleets have released porbeagles, and many of these released sharks are likely to have survived (Campana et al. 2016). Allowing for post-release survival would reduce these fishing mortality estimates, and reduce the estimated risk below the low risk levels estimated here.

The risk assessment assumes that catchability estimates from Japanese vessels are applicable to other fleets. The three targeting strategies identified for the Japanese fleet, which were explored by using clusters in the CPUE standardisation, had quite similar catch rates. This observation provides some reassurance about the applicability of our catch rate estimates to other fleets. Although it is possible that some other fleets' targeting strategies may have very different catch rates for porbeagles, we have no evidence for this.

This approach also assumes that SST estimates from observers on Japanese vessels are equivalent to modelled SST estimates from the CSIRO CARS database, the CARS estimates being averages across multiple years with no allowance for inter-year variability. There is likely to be significant divergence between local, vessel-based observations of SST, and averaged model predictions, but we do not think this sufficient to substantially change the stock status results.

We also assumed an upper prior bound for *K* equivalent to a maximum plausible density of 5 sharks per km^2. The correct level for this assumption is unknown, and alternative values may slightly affect the outcomes.

The risk assessment results are based on strong assumptions about the population density distribution. We have assumed that density is driven primarily by SST, with smaller effects due to season, latitude and year, which implies that depletion is the same in all areas. This assumption was required to extend the analysis to areas and times without reliable catch data and thus cover the entire spatial domain. However, it is likely that spatial variation in historical effort has depleted some areas more, which would consequently have lower densities than other areas, and lower densities than we have assumed. Since catch is the product of effort and density, we would also have overestimated their pelagic longline catches. However, given the approach used in the assessment, this effect does not bias the estimates of fishing mortality. Further exploration of this issue could involve applying our SST-based spatial-environmental model to the Japanese longline data by region, predicting distribution and catches independently for each region, and allowing re-estimation of catchability for the calibration region.

The risk assessment assumes that population density from 45 to 55 ^{o}S is the same as at 40 to 45 ^{o}S, and that density south of 55 ^{o}S is zero. We have evidence from fisheries and surveys that porbeagles occur south of 45 ^{o}S, but we do not have Japanese longline observer data with which to estimate density. This is an important assumption, because it implies that the low fishing effort south of 45 ^{o}S provides a refuge from fishing mortality for the population. Unfortunately, there is little information about patterns of population density south of 45 ^{o}S, but there is considerable evidence that porbeagles are found there, with observations recorded for most of the regions considered here. In the Eastern Pacific catch rates were observed increasing to the southern limits of the JAMARC longline survey (60 ^{o}S) and were relatively stable to the southern limits of the JAMARC gillnet survey (52.5 ^{o}S) (Yatsu 1995, Semba et al. 2013, Hoyle et al. 2017b). Also in the Eastern Pacific, porbeagles have been caught In the Chilean demersal longline fishery to at least 56 ^{o}S. In the Western Pacific they have been taken in the New Zealand midwater trawl fishery to 53 ^{o}S (Francis 2013). In the Western Indian Ocean, porbeagles have been taken in the mackerel icefish and Patagonian toothfish trawl and longline fisheries in the Heard and McDonald Island EEZ (53 ^{o}S) (ABARES data, Heather Chapman, pers. comm.). The analysis of data from the Western Atlantic Argentinian surimi fleet (Cortés et al. 2017) conducted for this study shows catch rates increasing to the south, but this result may be due to the local geography, with strong currents, temperature variation, and depth changes around Cape Horn. We recommend further work to understand this southerly population, such as future analysis of bycatch information being collected from the Chilean demersal longline fishery, which fishes as far south as 56 °S.

We also recommend exploring selectivity at age in the Japanese pelagic longline data, which may permit estimation of the availability at age of the population to fishing. This analysis may permit two further developments: an age-structured version of the BDM biomass dynamic risk assessment (Edwards 2017); and direct estimation of the proportion of the population south of 45 ^{o}S, removing the need to assume constant density from 45 to 55 ^{o}S.

This analysis assumed separation of the population into five regions, but there is little information available with which to determine appropriate stock boundaries. We recommend analyses of distribution using various tools (genetics, microchemistry, stable isotopes, parasites, conventional and electronic tags) to identify biologically-based boundaries.

The multiple indicators/risk assessment approach served to 1) source and synthesise available information on porbeagle shark at the scale of the Southern Hemisphere; 2) identify important data gaps (e.g., density distribution and life-stage specific vulnerability and overlap with fishing activities); 3) define productivity-based reference points for the species; and 4) prioritise fishery areas for monitoring and management. This project has filled important information gaps by both directly analysing available life history information, and providing statistical support to the analyses by participating national fisheries scientists.

The project has provided the first assessment of the sustainability of the impact of fishing on the Southern Hemisphere porbeagle shark stock, and laid a foundation for future work. Results indicate that the impact of fishing is low across the entire Southern Hemisphere range of the porbeagle shark population.

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Appendix A R code for calculating areas of grid cells

```
windows()
m <- map(regions = c("South Africa", "Australia", "New Zealand", "Argentina",
"Chile", "Antarctica", "Uruguay", "Botswana", "Namibia", "Zimbabwe",
                     "South Georgia", "Falkland Islands", "Mozambique",
"Madagascar","Brazil", "Paraguay", "French Southern and Antarctic Lands", "Heard
Island"), fill = TRUE)
#m <- map(fill = TRUE)
#identify.map(m)
res <- expand.grid(ln = seq(-180, 180, 5), lt = seq(-60, -30, 5), garea = NA, areax
= NA)
i=1
for (i in 1:length(res$lt)) {
  lt = res$lt[i]; ln = res$ln[i]
  geo_str <- paste0("+proj=laea +lon_0=",ln," +lat_0=",lt," +datum=WGS84")
  crs.geox <- CRS(geo_str)
  grcl <- data.frame(lon = c(ln, ln+5, ln+5, ln), lat = c(lt+5, lt+5, lt, lt))
  coordinates(grcl) <- ~ lon + lat
  projection(grcl) <- "+init=epsg:4326"
  grclx <- Polygon(grcl)


  cells <- Polygons(list(grclx), "onecell")
  cells2 <- SpatialPolygons(list(cells))
  projection(cells2) <- crs.geo

  m.spx <- map2SpatialPolygons(m, IDs=m$names,proj4string=crs.geox)
  m.spx <- gSimplify(m.spx, tol = 0.00001)
  m.spx <- gBuffer(m.spx, byid=TRUE, width=0)
  m.sp <- map2SpatialPolygons(m, IDs=m$names,proj4string=crs.geo)
  m.sp <- gSimplify(m.sp, tol = 0.00001)

  m.diff <- gDifference(cells2, m.sp)
  if(is.null(m.diff)) {
    res$garea[i] <- 0
    res$areax[i] <- 0
  } else {
    md2 <- spTransform(m.diff, crs.geox)
    res$garea[i] <- gArea(md2)/1e6
    res$areax[i] <- areaPolygon(m.diff)/1e6
  }
  print(i); flush.console()
}
```

Appendix B Fishing mortality estimates

Median *F* in each region and the assessment area (three regions combined) (rounded to four decimal places).

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	0.0026	0.0028	0.0020	0.0025
1993	0.0040	0.0011	0.0016	0.0024
1994	0.0028	0.0014	0.0011	0.0019
1995	0.0028	0.0016	0.0009	0.0019
1996	0.0022	0.0022	0.0008	0.0019
1997	0.0025	0.0023	0.0010	0.0021
1998	0.0024	0.0026	0.0015	0.0023
1999	0.0029	0.0023	0.0016	0.0024
2000	0.0020	0.0028	0.0013	0.0021
2001	0.0029	0.0023	0.0018	0.0024
2002	0.0019	0.0013	0.0020	0.0016
2003	0.0015	0.0016	0.0017	0.0016
2004	0.0021	0.0015	0.0009	0.0016
2005	0.0022	0.0013	0.0006	0.0015
2006	0.0019	0.0009	0.0003	0.0012
2007	0.0014	0.0009	0.0004	0.0010
2008	0.0014	0.0012	0.0003	0.0011
2009	0.0016	0.0014	0.0004	0.0012
2010	0.0011	0.0011	0.0003	0.0009
2011	0.0013	0.0009	0.0004	0.0009
2012	0.0010	0.0006	0.0005	0.0007
2013	0.0010	0.0009	0.0005	0.0008
2014	0.0007	0.0011	0.0005	0.0008

Annual upper 95% CIs for *F* in each region and the assessment area (three regions combined) (rounded to four decimal places).

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	0.0007	0.0007	0.0005	0.0006
1993	0.0010	0.0003	0.0004	0.0006
1994	0.0007	0.0003	0.0003	0.0005
1995	0.0007	0.0004	0.0002	0.0005
1996	0.0006	0.0005	0.0002	0.0005
1997	0.0006	0.0006	0.0002	0.0005
1998	0.0006	0.0006	0.0004	0.0006
1999	0.0007	0.0006	0.0004	0.0006
2000	0.0005	0.0007	0.0003	0.0005
2001	0.0007	0.0006	0.0005	0.0006
2002	0.0005	0.0003	0.0005	0.0004
2003	0.0004	0.0004	0.0004	0.0004
2004	0.0005	0.0004	0.0002	0.0004
2005	0.0005	0.0003	0.0002	0.0004
2006	0.0005	0.0002	0.0001	0.0003
2007	0.0003	0.0002	0.0001	0.0002
2008	0.0003	0.0003	0.0001	0.0003
2009	0.0004	0.0003	0.0001	0.0003
2010	0.0003	0.0003	0.0001	0.0002
2011	0.0003	0.0002	0.0001	0.0002
2012	0.0002	0.0001	0.0001	0.0002
2013	0.0002	0.0002	0.0001	0.0002
2014	0.0002	0.0003	0.0001	0.0002

Annual lower 95% CIs for *F* in each region and the assessment area (three regions combined) (rounded to four decimal places).

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	0.0203	0.0213	0.0153	0.0196
1993	0.0309	0.0084	0.0124	0.0181
1994	0.0217	0.0104	0.0086	0.0145
1995	0.0212	0.0126	0.0073	0.0149
1996	0.0173	0.0170	0.0058	0.0147
1997	0.0193	0.0179	0.0077	0.0162
1998	0.0186	0.0200	0.0113	0.0176
1999	0.0225	0.0177	0.0123	0.0184
2000	0.0154	0.0215	0.0096	0.0165
2001	0.0220	0.0177	0.0142	0.0185
2002	0.0143	0.0101	0.0150	0.0126
2003	0.0119	0.0126	0.0129	0.0122
2004	0.0164	0.0118	0.0071	0.0126
2005	0.0167	0.0099	0.0048	0.0115
2006	0.0142	0.0069	0.0026	0.0089
2007	0.0105	0.0072	0.0028	0.0076
2008	0.0105	0.0091	0.0025	0.0083
2009	0.0120	0.0104	0.0030	0.0095
2010	0.0087	0.0082	0.0025	0.0072
2011	0.0097	0.0066	0.0031	0.0071
2012	0.0077	0.0044	0.0036	0.0055
2013	0.0074	0.0066	0.0039	0.0063
2014	0.0051	0.0086	0.0040	0.0062

Appendix C *F*-ratios

Annual upper 95% CIs for *F*-ratio$_{crash}$ metric, for the impact of pelagic longline fisheries on porbeagle shark in each region and the assessment area (three regions combined) (rounded to three decimal places).

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	0.828	0.882	0.634	0.906
1993	1.345	0.370	0.517	0.731
1994	0.923	0.437	0.327	0.596
1995	0.904	0.545	0.298	0.602
1996	0.721	0.711	0.250	0.596
1997	0.822	0.774	0.323	0.721
1998	0.769	0.817	0.503	0.738
1999	0.916	0.714	0.496	0.734
2000	0.678	0.944	0.405	0.710
2001	0.994	0.784	0.598	0.784
2002	0.579	0.409	0.622	0.474
2003	0.498	0.525	0.522	0.487
2004	0.696	0.541	0.287	0.526
2005	0.685	0.431	0.216	0.509
2006	0.602	0.298	0.110	0.361
2007	0.460	0.303	0.117	0.307
2008	0.436	0.384	0.104	0.335
2009	0.497	0.456	0.126	0.392
2010	0.375	0.373	0.098	0.325
2011	0.400	0.272	0.131	0.278
2012	0.310	0.178	0.150	0.244
2013	0.285	0.272	0.164	0.264
2014	0.204	0.356	0.176	0.245

Annual lower 95% CIs F-ratio$_{crash}$ metric, for the impact of pelagic longline fisheries on porbeagle shark in each region and the assessment area (three regions combined) (rounded to three decimal places).

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	0.014	0.015	0.011	0.014
1993	0.021	0.006	0.009	0.012
1994	0.014	0.007	0.006	0.010
1995	0.014	0.008	0.005	0.011
1996	0.012	0.012	0.004	0.010
1997	0.014	0.013	0.005	0.011
1998	0.013	0.014	0.008	0.012
1999	0.016	0.012	0.008	0.013
2000	0.011	0.014	0.007	0.011
2001	0.015	0.012	0.010	0.014
2002	0.010	0.007	0.010	0.009
2003	0.008	0.009	0.009	0.008
2004	0.011	0.008	0.005	0.008
2005	0.011	0.007	0.003	0.008
2006	0.010	0.005	0.002	0.006
2007	0.007	0.005	0.002	0.005
2008	0.007	0.006	0.002	0.006
2009	0.008	0.007	0.002	0.007
2010	0.006	0.006	0.002	0.005
2011	0.007	0.005	0.002	0.005
2012	0.005	0.003	0.002	0.004
2013	0.005	0.004	0.003	0.004
2014	0.003	0.006	0.003	0.004

Annual upper 95% CIs for *F*-ratio$_{lim}$ metric, for the impact of pelagic longline fisheries on porbeagle shark in each region and the assessment area (three regions combined) (rounded to three decimal places).

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	1.104	1.177	0.846	1.208
1993	1.793	0.493	0.690	0.974
1994	1.230	0.582	0.436	0.794
1995	1.205	0.727	0.397	0.803
1996	0.961	0.947	0.333	0.794
1997	1.096	1.032	0.430	0.961
1998	1.025	1.089	0.670	0.984
1999	1.221	0.952	0.661	0.978
2000	0.904	1.258	0.540	0.947
2001	1.326	1.045	0.798	1.045
2002	0.772	0.545	0.830	0.632
2003	0.664	0.700	0.695	0.649
2004	0.928	0.721	0.382	0.702
2005	0.913	0.574	0.287	0.678
2006	0.803	0.397	0.147	0.482
2007	0.614	0.405	0.156	0.409
2008	0.581	0.512	0.138	0.447
2009	0.662	0.609	0.168	0.523
2010	0.500	0.498	0.131	0.433
2011	0.533	0.362	0.174	0.370
2012	0.414	0.237	0.200	0.326
2013	0.381	0.362	0.219	0.352
2014	0.272	0.475	0.235	0.326

Annual lower 95% CIs F-ratio$_{lim}$ metric, for the impact of pelagic longline fisheries on porbeagle shark in each region and the assessment area (three regions combined) (rounded to three decimal places).

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	0.018	0.019	0.014	0.018
1993	0.028	0.008	0.012	0.017
1994	0.019	0.009	0.008	0.013
1995	0.019	0.011	0.007	0.014
1996	0.016	0.016	0.006	0.013
1997	0.018	0.017	0.007	0.015
1998	0.018	0.019	0.010	0.016
1999	0.021	0.017	0.011	0.017
2000	0.014	0.019	0.009	0.015
2001	0.020	0.016	0.013	0.018
2002	0.013	0.010	0.014	0.012
2003	0.011	0.012	0.013	0.011
2004	0.015	0.011	0.007	0.011
2005	0.015	0.009	0.004	0.010
2006	0.013	0.006	0.002	0.008
2007	0.010	0.006	0.003	0.007
2008	0.010	0.008	0.002	0.008
2009	0.011	0.010	0.003	0.009
2010	0.008	0.007	0.002	0.007
2011	0.009	0.007	0.003	0.007
2012	0.007	0.004	0.003	0.005
2013	0.007	0.006	0.004	0.006
2014	0.005	0.008	0.004	0.006

Annual upper 95% CIs for F-ratio$_{msm}$ metric, for the impact of pelagic longline fisheries on porbeagle shark in each region and the assessment area (three regions combined) (rounded to three decimal places).

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	1.655	1.765	1.268	1.813
1993	2.689	0.739	1.035	1.462
1994	1.845	0.874	0.654	1.191
1995	1.807	1.091	0.596	1.204
1996	1.441	1.421	0.500	1.191
1997	1.644	1.549	0.645	1.442
1998	1.538	1.633	1.006	1.476
1999	1.832	1.428	0.992	1.468
2000	1.356	1.887	0.811	1.420
2001	1.989	1.567	1.197	1.567
2002	1.159	0.818	1.245	0.948
2003	0.997	1.051	1.043	0.974
2004	1.392	1.081	0.574	1.053
2005	1.369	0.862	0.431	1.017
2006	1.204	0.595	0.220	0.723
2007	0.920	0.607	0.234	0.614
2008	0.871	0.768	0.208	0.671
2009	0.993	0.913	0.252	0.785
2010	0.749	0.747	0.196	0.649
2011	0.799	0.543	0.261	0.555
2012	0.621	0.356	0.300	0.489
2013	0.571	0.543	0.329	0.528
2014	0.408	0.712	0.352	0.489

Annual lower 95% CIs F-ratio$_{msm}$ metric, for the impact of pelagic longline fisheries on porbeagle shark in each region and the assessment area (three regions combined) (rounded to three decimal places).

Year	Eastern Atlantic Ocean/Western Indian Ocean	Eastern Indian Ocean	Western Pacific Ocean	Combined
1992	0.028	0.029	0.021	0.027
1993	0.042	0.012	0.018	0.025
1994	0.028	0.014	0.012	0.020
1995	0.029	0.017	0.010	0.021
1996	0.024	0.024	0.008	0.020
1997	0.027	0.026	0.011	0.022
1998	0.027	0.028	0.015	0.025
1999	0.032	0.025	0.017	0.025
2000	0.021	0.029	0.013	0.023
2001	0.031	0.024	0.020	0.027
2002	0.020	0.014	0.020	0.018
2003	0.017	0.018	0.019	0.016
2004	0.022	0.016	0.010	0.017
2005	0.022	0.014	0.007	0.016
2006	0.020	0.010	0.004	0.012
2007	0.014	0.010	0.004	0.010
2008	0.014	0.013	0.003	0.011
2009	0.016	0.015	0.004	0.013
2010	0.012	0.011	0.003	0.010
2011	0.013	0.010	0.004	0.010
2012	0.011	0.006	0.005	0.008
2013	0.010	0.009	0.005	0.009
2014	0.007	0.012	0.006	0.008