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Bycatch in longline fisheries for tuna and tuna-like species

A global review of status and mitigation measures

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BYCATCH IN LONGLINE FISHERIES FOR TUNA AND TUNA-LIKE SPECIES: A GLOBAL REVIEW OF STATUS AND MITIGATION MEASURES

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This publication is the third in a series of FAO Fisheries and Aquaculture Technical Papers reviewing bycatch in global tuna fisheries and it deals with longline fisheries. Its scope is defined taxonomically to comprise only non-tuna and non-tuna-like species, i.e. elasmobranchs, sea turtles, seabirds, marine mammals and other (non-target) bony fishes.

This document was written and edited by Dr Shelley Clarke, with the exception of Chapter 4, which was written by Drs Mayumi Sato, Cleo Small, Ben Sullivan, Yukiko Inoue and Daisuke Ochi.

Abstract

This publication is the third in a series on bycatch in global tuna fisheries. Dealing with longline fisheries, its scope is defined taxonomically to comprise only non-tuna and non-tuna-like species.

The history of longline fishing illustrates the role of new technologies, the expansion of fishing grounds, and the operational characteristics of the fleets in shaping today's fishery. More recently, management regulations, the price of oil, the cost of labour, and market demand have also exerted an influence. No more than 23 percent of the tuna in each ocean is longline-caught. However, there may be up to 7 500 tuna longliners globally with almost 60 percent of them less than 24 m in length.

Available data suggest that elasmobranch catches have fallen 14 percent since their peak in 2003. In longline fisheries, shark catch rates may be determined by bait type, soak time, hook shape, leader length and material, depth at which the hook is fished, and whether special gear is deployed to target sharks. Vulnerability to hooking, and resilience to haulback and handling, vary by species, size, area and fleet operational practices. Tuna regional fisheries management organizations (t-RFMOs) assess the status of shark populations but data limitations often hinder firm conclusions. There is little information on the implementation or effectiveness of finning bans and no-retention measures. Mitigation measures have been tested but results vary.

Six of the seven species of sea turtles are threatened with extinction, and while longline fisheries may have less impact than net-based fisheries, significant population-level impacts may be occurring in some regions. The greatest concern is associated with loggerhead–longline interactions in the Atlantic. Circle hooks and using finfish bait have proved effective mitigation techniques either by reducing hooking or hook swallowing. Other methods require further development.

Interactions with pelagic longline fisheries kill 50 000–100 000 seabirds annually. Many of these species, particularly albatrosses, are threatened with extinction. Recent advances in tracking technologies have facilitated mapping of where interactions are most likely. The Western and Central Pacific contains more than 45 percent of the global total albatross and giant petrel breeding distributions. The most promising mitigation methods appear to be night setting, side-setting, line weighting and streamer lines, but further research is needed. All five t-RFMOs require use of one or more of these methods in areas that overlap albatross distributions. However, compliance data are limited and improved observer coverage is essential.

Marine mammals' interactions with longline fisheries are detrimental to the fishery but may be positive or negative for the mammals. Although it is often unclear which species are involved, pilot whale interactions in the western Atlantic and false killer whale interactions off Hawaii have triggered national mitigation plans. No t-RFMO has adopted management measures for marine mammal interactions. Research and testing of mitigation measures continue in order to ameliorate both marine mammal impacts and economic losses to industry from depredation.

At least 650 species of other bony fishes may be caught in association with pelagic longline fisheries, e.g. dolphinfish, opah, oilfish, escolar and ocean sunfish. Some of these stocks are important as local food supplies. However, it is unclear whether these stocks or the ecosystem they help structure is at risk. More attention should focus on improving fishery statistics and initiating basic monitoring of these stocks' status.

The diversity of pelagic longline gear designs and fishing methods, the variety of habitats they are deployed in, the thousands of marine species they may interact with,

and the different mechanisms and behaviours that govern those interactions provide an array of topics to be addressed in any discussion of bycatch mitigation. Scientific and technical issues in mitigation including effects across taxa, effects of combinations of measures, economic and safety considerations, underlying biological mechanisms, handling and post-release mortality, and non-fishery impacts must all be addressed. In addition, it is also necessary to consider issues such as who takes the lead for ensuring mitigation is sufficient for the population as a whole, how to devise effective mitigation implementation strategies, and whether gear modification should be used in concert with more sweeping measures.

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Abbreviations and acronyms

ACAP	Agreement for the Conservation of Albatrosses and Petrels
ADD	acoustic deterrent device
AFMA	Australian Fisheries Management Authority
AHD	acoustic harassment device
AIDCP	Agreement on the International Dolphin Conservation Programme
BMIS	Bycatch Mitigation Information System
BPUE	bycatch per unit effort
CCAMLR	Commission for the Conservation of Antarctic Marine Living Resources
CCSBT	Commission for the Conservation of Southern Bluefin Tuna
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
CMM	Conservation and Management Measure
CMS	Convention on the Conservation Migratory Species (Bonn Convention)
Code	FAO Code of Conduct for Responsible Fisheries
COP	Conference of the Parties (CITES)
CWBR	Consortium for Wildlife Bycatch Reduction
DAFF	Department of Agriculture, Forestry and Fisheries (Australia)
EDD	echolocation disruption device
EPO	Eastern Pacific Ocean
ERSWG	Ecologically Related Species Working Group (CCSBT)
FIGIS	Fisheries Global Information System (FAO)
IAC	Inter-American Convention for the Protection and Conservation of Sea Turtles
IATTC	Inter-American Tropical Tuna Commission
ICCAT	International Commission for the Conservation of Atlantic Tunas
IOSEA	Indian Ocean South-east Asian Marine Turtle Memorandum of Understanding
IOTC	Indian Ocean Tuna Commission
IPOA	international plan of action
IPOA-Seabirds	International Plan of Action for Reducing Incidental Catch of Seabirds in Longline Fisheries
ISC	International Scientific Committee (North Pacific)
ISSF	International Seafood Sustainability Foundation
IUCN	International Union for the Conservation of Nature
IUU	illegal, unreported and unregulated (fishing)
IWC	International Whaling Commission
LED	light-emitting diode
MADE	Mitigation of Adverse Ecological Impacts of Open Ocean Fisheries (project)
MSY	maximum sustainable yield
NEI	not elsewhere indicated
NGO	non-governmental organization
NMFS	National Marine Fisheries Service (United States of America)
NOAA	National Oceanic and Atmospheric Administration (United States of America)
NPOA	national plan of action
PBR	potential biological removal

PFRP	Pelagic Fisheries Research Program
PLC	programmable logical controller
PSA	productivity–susceptibility analysis
RMU	regional management unit
ROP	Regional Observer Programme
SBWG	Seabird Bycatch Working Group
SPC	Secretariat of the Pacific Community
SST	sea surface temperature
t-RFMO	tuna regional fisheries management organization
TRP	take reduction plan
WPEB	Working Party on Ecosystems and Bycatch (IOTC)
WCPFC	Western and Central Pacific Fisheries Commission
WCPO	Western and Central Pacific Ocean

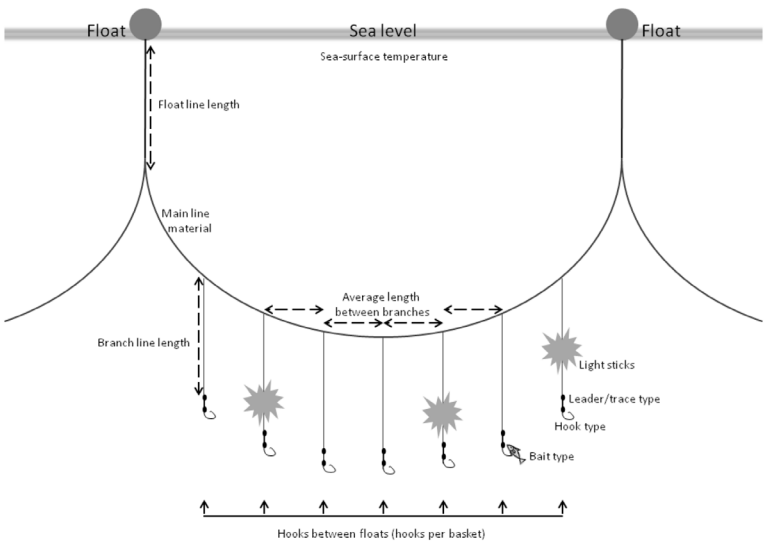
1. Introduction

1.1 BACKGROUND AND DEFINITION OF SCOPE

This paper is the third in a series of FAO Fisheries and Aquaculture Technical Papers reviewing the status and trends for bycatch in global tuna fisheries. Previous papers in the series have dealt with small-scale (undecked and/or unpowered) tuna fisheries of all gear types (Gillett, 2011) and tropical tuna purse seine fisheries (Hall and Roman, 2013). This paper completes the set by reviewing bycatch issues associated with large-scale pelagic longline fishing. Each paper has addressed a different set of challenges when presenting its global review. For small-scale fisheries, the diversity of gear types and the anecdotal nature of the data led to a focus on estimating bycatch quantities and identifying priority issues at the national level. For purse seine fisheries, the emphasis was on characterizing the various types of purse seine operations and providing new insights into how the behaviour of bycatch species shapes the nature of the interactions. As a result of longstanding concern about longline bycatch, as well as the wide range of selectivities of longline gear of various configurations, bycatch issues in longline fisheries are perhaps simultaneously the most complex and the best studied of all gear types. The challenge for this paper is thus to summarize the vast body of literature on longline interactions and to point to the most pressing needs and promising methods for mitigating impacts.

In delineating the scope of this study, it is necessary to define both pelagic longline fishing and which subset of its catches should be discussed as bycatch. Longline gear used in fisheries for tuna and tuna-like species consists of a mainline set in the water column from surface floats, with baited hooks attached by means of thinner branch lines, also called leaders, snoods or gangions (Figure 1). The gear operates by attracting fish to the bait from as far away as several hundred metres (Bjordal, 2009).

FIGURE 1
Schematic of a pelagic longline



Notes: The length of the branch lines and the average length between branches are not to scale. In addition, light sticks are used only when targeting swordfish, and sets targeting tunas may deploy considerably more branch lines.
Source: IOTC Resolution 13/03.

Beyond this simplistic description lies a wide variety of material types, hooks, baits, set configurations and operational practices (Beverly, Chapman and Sokimi, 2003) that influence the diversity, size range and amount of target and non-target catches. Longline fishing gear has been depicted at two extremes as highly selective (Bjordal and Løkkeborg, 1996; Smith, 1998) and unselective/non-selective (Pauly and Froese, 2001). Part of the explanation for this dichotomy is the relative and target-specific nature of the evaluation of selectivity. In a study of discard rates (in tonnes) by different gear types, average discard rates for tuna and highly migratory species from longlines (28.5 percent) were found to be second only to shrimp trawls (62.3 percent), but when examined individually some longlines had lower discard rates than some forms of pots, dredges and gill (or drift) nets (Kelleher, 2005). While pelagic longlines are thus relatively selective compared with trawls and some other gear types, they are relatively unselective in terms of total discards when compared with other major tuna fishing gear types such as purse seines (average 5.1 percent discards) and pole and line (average 0.4 percent discards) (Kelleher, 2005). A study of discarded fish and invertebrates in United States fisheries also suggested that pelagic longline bycatch rates are low (3.9 percent) compared with six other gear types (4.5–46.9 percent), although they are higher than rates for gillnets and purse seines (0.7–1.2; Harrington et al. 2005). From another perspective, it has been known for some time that longline gear is more selective for larger fish which are not in schools (Brock 1962, Bjordal 2009). Because longline gear avoids wasteful catches of large quantities of very small, unusable tunas, when considering tuna species per se it could be argued that longlines are more selective than purse seine gear. These comparisons reveal that the evaluation of gear selectivity, and thus the definition of bycatch, will depend on how narrowly the range of targets are delineated in any given fishing operation.

Defining what is meant by "bycatch in longline fisheries for tuna and tuna-like species" for this paper is both essential and problematic. From among a range of alternative meanings for the term "bycatch" (e.g. see Alverson *et al*., 1994, Gilman, Passfield and Nakamura, 2013), one paper in this series has defined bycatch from tuna fisheries by means of taxonomy as "non-tuna species" whether retained or discarded (Gillett, 2011). Noting the problems associated with establishing which species are target species, even in tuna-targeting fisheries, the other paper in this series defined bycatch by means of fate as "dead discards" regardless of species (Hall and Roman, 2013). It is known that in longline fisheries, small vessels making shorter trips are more likely to have fewer discards (i.e. are more likely to retain all catch) than larger distant-water vessels (Kelleher, 2005) even though they might both nominally be targeting tuna and be catching similar numbers and types of associated species. Such differences in catch retention practices may occur not only among fleets but among vessels in the same fleet, or even among sets from a single vessel. This variation in discarding practices, in combination with the current poor state of knowledge of post-release mortality for many species, suggests it would be difficult to attempt to define longline bycatch on the basis of dead discards.

Therefore, this paper adopts a taxonomic approach to the definition of bycatch by excluding tuna and tuna-like species regardless of size (see discussion of longline selectivity above). Target species of longline fisheries for tuna and tuna-like species are thus defined as the 51 species of the family Scombridae (mackerels, Spanish mackerels, bonitos and tunas), and the 13 species of the families Istiophoridae and Xiphiidae (billfishes) (FAO, 2000). While many of these species may not be intentionally targeted by pelagic longline fleets owing to their low abundance or low commercial value, their biology and behaviour are similar to the primary market species. Conversely, species such as sharks have distinctly different biology and behaviour from tuna and tuna-like species, and are thus discussed in this report as bycatch even if they are

sometimes target species. (In order to avoid confusing species that are included in this report because they fit the taxonomic definition of bycatch, but which are actually important components of the longline catch, this report, whenever possible, refers to "interactions" rather than bycatch and also specifies whether the animal is retained or discarded.) This taxonomic approach to defining bycatch supports this study's focus on characterizing interactions and evaluating mitigation measures, both of which are biologically driven and require different strategies for different taxa.

Having defined the subset of catch that is of interest to this study, it is also necessary to define the subset of longline fisheries worldwide to be discussed. For general scoping purposes, it is assumed that the global longline fishery for tuna and tuna-like species consists of those longliners that are authorized by their flag States to fish in the convention areas of the five global tuna regional fisheries management organizations (t-RFMOs). However, some caveats apply. First, it is recognized that some of the more than 10 000 longliners on the t-RFMO lists may not be primarily targeting tuna, e.g. some may be targeting swordfish, sharks and/or other bony fishes and catching tunas incidentally. Second, different t-RFMOs have different criteria for their authorized vessel lists: some list all vessels capable of fishing for tuna or tuna-like species regardless of size, while others specify vessel size limits or are limited to those fishing outside of national waters (Table 1). Applying these criteria in this study may to varying extents exclude small longliners, but those that are open- or partially-decked have been covered in the small-scale fisheries study (Gillett, 2011). Third, for this paper, a liberal approach was taken in compiling information to characterize longline bycatch. For this characterization, literature and data were compiled for longline fishing operations resembling those fishing for tuna or tuna-like species regardless of whether or not they actually fished under the jurisdiction of the t-RFMOs. The compiled information often did not provide details of the size of fishing vessels or the location (e.g. distance from shore) of their operations. Therefore, characterizations in this paper represent a mix of small coastal longline fisheries and large distant-water longline fisheries, which may have very different bycatch profiles.

The scope of this paper on bycatch in pelagic longline fisheries is thus defined as interactions, management and mitigation for species other than tuna and tuna-like species (families Scombridae, Istiophoridae and Xiphiidae) caught by longliners authorized to fish in or capable of fishing in the convention areas of the t-RFMOs. The focus of this document is on the management measures adopted by, and the data reported to, the t-RFMOs. However, in some cases, national regulations or management programmes are also discussed to illustrate specific issues or innovative solutions.

The remainder of this introduction describes the data sets and sources used to prepare this review and provides a summary of the current characteristics of the global pelagic longline fleet. The following chapters are organized taxonomically to discuss elasmobranchs (sharks and rays), sea turtles, seabirds, marine mammals and other bony fishes. The concluding chapter summarizes the key themes across taxa in both technical and policy fields.

TABLE 1
Criteria for vessels required to be listed on the record of fishing vessels for each t-RFMO and the number of longline vessels listed

t-RFMO	Criteria	Number
Commission for the Conservation of Southern Bluefin Tuna (CCSBT)	All vessels to be used for fishing for Southern bluefin tuna	640
Inter-American Tropical Tuna Commission (IATTC)	All vessels to be used for fishing for species under the IATTC competence in the Convention Area beyond the flag State's area of national jurisdiction	2 459
International Commission for the Conservation of Atlantic Tunas (ICCAT)	All vessels ≥ 20 m in length overall (and all vessels regardless of size when outside the flag State's area of national jurisdiction) to be used for fishing tuna in the Convention Area, and all vessels fishing for Eastern bluefin tuna, Mediterranean swordfish and Northern albacore	1 607
Indian Ocean Tuna Commission (IOTC)	All vessels >24 m length overall to be used for fishing for species under the IOTC competence in the Convention Area, and all vessels <24 m length overall to be used for fishing for species under the IOTC competence in the Convention Area beyond the flag State's area of national jurisdiction	2 205
Western and Central Pacific Fisheries Commission (WCPFC)	All vessels to be used for fishing for species under the WCPFC competence in the Convention Area beyond the flag State's area of national jurisdiction	3 766

Source: Adapted from Joint Tuna RFMOs (2013a).

1.2 DATA SOURCES AND DATA QUALITY ISSUES

1.2.1 T-RFMO data sets relevant to longline bycatch

The most comprehensive sources of information on pelagic longline fisheries are the records maintained by the t-RFMOs. However, each t-RFMO has its own data reporting requirements, and the quantity and quality of data reported vary in their compliance with these requirements by fishery and flag State. Moreover, in terms of bycatch, t-RFMO data holdings are considerably less complete than for tuna and tuna-like species. This situation arises owing to both lack of bycatch data collection by fishers and flag States, and lack of provision of existing bycatch data by flag States to t-RFMOs.

T-RFMO data relevant to longline bycatch are contained in logsheet, observer and port sampling databases or other reports (e.g. members' annual reports) held by the t-RFMOs. Of these, the most useful source of information is observer data, as many bycatch species are either not recorded on logsheets or recorded in non species-specific categories. Exceptions to this include sharks and other fishes, particularly those of commercial value, which may be recorded in either logbook or port sampling databases. However, even if some bycatch species are recorded, this is often done in a sporadic manner depending on national reporting requirements, vessel operator policies, the amount of time available for deck handling of less valuable species, and other factors (Walsh, Kleiber and McCracken, 2002). Although the observer data provide the most complete record of interactions with bycatch species, it is important to note that observer coverage in most longline fleets, particularly those with small vessels, is typically 5 percent or less of the total fishing effort. As the observer data sets are often skewed, either toward larger vessels (owing to the fact that smaller vessels may have difficulties accommodating observers) or fleets from countries with more developed observer programmes, or are otherwise uneven spatially or temporally,

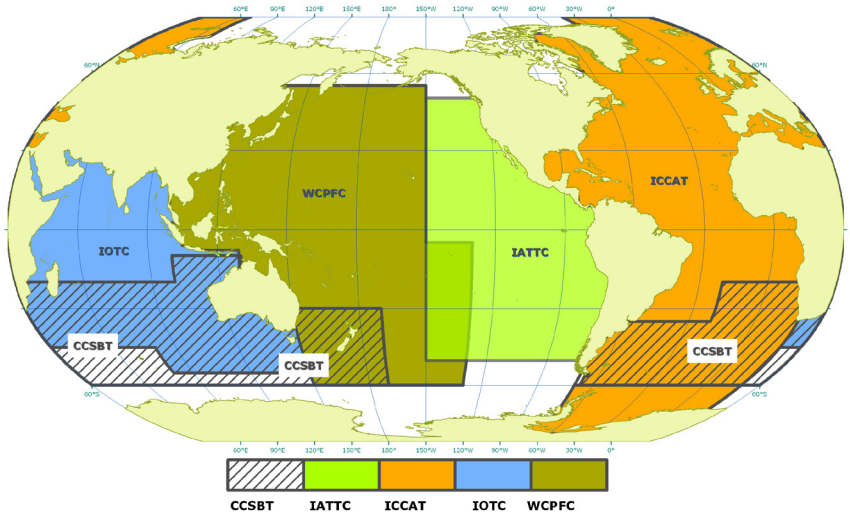
extrapolation from existing observer data to the entire fishery can be problematic. Data gaps in and uncertainties arising from available bycatch data are discussed for each taxa in the following chapters.

Beyond observer programmes, and the standard reporting of catch and effort data, which may or may not include bycatch species, some t-RFMO taxa-specific management measures require annual reporting of bycatch interactions wherever they occur. The amount and availability of these data vary among t-RFMOs. Most t-RFMOs require some form of annual bycatch interaction reporting, but most of these data are not in the public domain, and in most cases overall estimates of mortality and population impact are not undertaken or published. Brief summaries of the bycatch data reporting requirements and holdings for each t-RFMO are presented below. Details of individual bycatch management measures for each RFMO are discussed in subsequent chapters.

1.2.1.1 Commission for the Conservation of Southern Bluefin Tuna

The Commission for the Conservation of Southern Bluefin Tuna (CCSBT) was established to manage and conserve southern bluefin tuna (*Thunnus maccoyii*) throughout its range. It held its first meeting of the Ecologically Related Species Working Group (ERSWG) in 1995 (CCSBT 1995). It has adopted one binding measure for mitigating impacts on seabirds from longlines and various other non-binding measures for other species caught in association with southern bluefin tuna. Its members are also required to comply with all applicable measures adopted by the International Commission for the Conservation of Atlantic Tunas (ICCAT), Indian Ocean Tuna Commission (IOTC) and Western and Central Pacific Fisheries Commission (WCPFC) when operating in southern bluefin tuna fishing grounds overlapping any of these three t-RFMOs (Figure 2).

FIGURE 2
Competence areas of the t-RFMOs



Source: FAO (2012–2014)

The CCSBT agreed a data exchange (data provision) measure for ecologically related species at its annual meeting in October 2012. Under the adopted measure, data on captures, mortalities and live releases for sharks (blue, mako, porbeagle and other), sea turtles (seven species) and seabirds (large albatrosses, dark-coloured albatrosses, other albatrosses, giant petrels, and other) are to be reported in 2013 for 2010–12 and

annually thereafter, and will be held confidentially by the CCSBT. These data, or more detailed data provided by members, may be used in bycatch assessments conducted by the CCSBT ERSWG on an ad hoc basis (CCSBT, 2012a). Past ERSWG assessment results are not in the public domain. The CCSBT has set a target for observer coverage of 10 percent, which is implemented through members' national programmes (CCSBT, 2013), but the CCSBT does not hold these observer data centrally. The achieved longline observer coverage is generally below the target level (CCSBT, 2012b). Discussions are currently under way regarding the development of CCSBT regional observer programme (CCSBT, 2012c).

1.2.1.2 Inter-American Tropical Tuna Commission

The Inter-American Tropical Tuna Commission (IATTC) established a working group on bycatch in 1997 but began placing observers in the purse seine fisheries as early as the 1970s (Hall and Roman, 2013). Until recently, most IATTC activities relating to bycatch impact assessment and mitigation focused on dolphins associated with the purse seine fishery through the Agreement on the International Dolphin Conservation Programme (AIDCP). The longline fishery in the Eastern Pacific Ocean (EPO) is responsible for about 20 percent of the tuna catch in the IATTC convention area and is less well documented than the purse seine fishery (IATTC, 2012a). Gear testing and hook exchange programmes to increase the use of circle hooks have been a major area of focus for longline bycatch mitigation since 2005 (IATTC, 2012a).

The IATTC is the only t-RFMO in which the observer programme is fully coordinated by the secretariat (Gilman, 2011). However, most of the bycatch data held by the IATTC have been generated by observers in the purse seine fishery where coverage of vessels with a capacity of more than 363 tonnes is 100 percent (IATTC, 2009, Hall and Roman, 2013). There is currently no IATTC-coordinated longline observer programme, but requirements for 5 percent longline coverage for vessels longer than 20 m, and recording of seabird, sea turtle and shark interactions by national observer programmes, were implemented in January 2013 (IATTC, 2011a). Owing to a lack of data available to assess the vulnerability of non-target species through more data-intensive approaches such as stock assessments, the IATTC has recently initiated productivity–susceptibility assessments, but these have been limited to purse seine bycatch species only (IATTC 2012b). An ongoing IATTC silky shark stock assessment has required several years of effort owing to difficulties in overcoming data gaps (IATTC, 2013a).

1.2.1.3 International Commission for the Conservation of Atlantic Tunas

The International Commission for the Conservation of Atlantic Tunas (ICCAT) established a subcommittee on bycatch in 1995, and this subcommittee was combined with its subcommittee on the environment in 2005 to form a subcommittee on ecosystems. In recent years, shark stock assessments have been conducted by a shark-specific species working group under the ICCAT Standing Committee on Research and Statistics. These shark stock assessments have used catch and effort data submitted by ICCAT members as part of their general fishery statistics, as well as other data such as market and trade statistics. Other bycatch taxa have been assessed through ecological risk assessment. Where more specific assessments are called for, these are based on voluntary submissions by ICCAT members and consultations with taxa experts. Assessments undertaken by ICCAT are subsequently published and are thus in the public domain. For all taxa, a lack of data has hindered estimates of total impacts (Joint Tuna RFMOs, 2011).

The International Commission for the Conservation of Atlantic Tunas requires national observer programme coverage of at least 20 percent on longliners actively fishing for bluefin tuna and at least 5 percent on other longline fleets (IATTC, 2009;

ICCAT, 2010). Proposals for an ICCAT scientific observer programme, including required coverage of 8 percent, have been repeatedly considered since 2007 but have not been adopted. Under the current observer programme model, some national observer programmes implement relatively high levels of coverage, while others report lower levels or do not report their coverage in a manner that allows an evaluation of overall coverage (ICCAT, 2012a). A recent survey of national observer programmes indicated that, of 12 respondents, 11 monitor sea turtles, 10 monitor sharks, 10 monitor mammals, and 7 monitor seabirds (ICCAT, 2012a). However, only one national observer programme reports bycatch species interaction data to ICCAT (Cotter, 2010), and few provide summarized information on the number of interactions (e.g. Birdlife International, 2011).

1.2.1.4 Indian Ocean Tuna Commission

The Indian Ocean Tuna Commission (IOTC) first formed a working group to discuss non-target, associated and dependent species in 2002, but this group did not meet until 2005 and was reorganized in 2007 as the Working Party on Ecosystems and Bycatch (WPEB). The WPEB considers bycatch assessments submitted by members, but these assessments are not in the public domain. Half of the tuna catch in the convention area derives from small-scale, artisanal fleets whose catches are not well characterized (Joint Tuna RFMOs, 2011). Several sharks are required to be reported at the species or genus level as part of standard catch and effort logsheet reporting for longline fisheries. In addition, IOTC members are required to report all data on sea turtle and seabird interactions to the secretariat each year. Despite these reporting requirements, the WPEB acknowledges that many of the potential data on bycatch interactions are missing or incomplete, and it is therefore planning to undertake ecological risk assessments (Joint Tuna RFMOs, 2011; IOTC, 2012a).

The IOTC established a regional observer programme in 2010 and extended it to vessels less than 24 m operating outside of their flag State's national waters in January 2013. Observers are supplied by national programmes and coverage is set at 5 percent, but actual coverage has been curtailed by an increasing incidence of piracy in the region and a need to use onboard berths for security personnel (Joint Tuna RFMOs, 2010). Observer trip reports must be submitted to the secretariat, but the number of submitted trip reports has been very low (11, 31 and 1, respectively, in 2010–12). The WPEB has stated that this low level of implementation and reporting undermines its ability to estimate incidental catches of non-targeted species as requested by the IOTC (IOTC, 2012a).

1.2.1.5 Western and Central Pacific Fisheries Commission

The Western and Central Pacific Fisheries Commission (WCPFC) is the newest of the five t-RFMOs and held its first commission meeting in December 2004. The scientific committee's ecosystem and bycatch specialist working group (now the Ecosystems and Bycatch Mitigation Theme) has met annually since 2005 to discuss assessments of bycatch species provided both by WCPFC members and by the WCPFC's scientific services provider, the Secretariat of the Pacific Community (SPC). At its meeting in August 2013, the working group considered more than 20 papers relevant to bycatch, all of which are in the public domain. Members of the WCPFC are required to report WCPFC "key shark species" to species or genus on logsheets. Since 2007, all interactions with seabirds have been required to be reported in members' Annual Reports Part 1 (which are in the public domain), and seabird mitigation measures applied have been required to be reported in Annual Reports Part 2 (which are not in the public domain). Since 2008, all interactions with sea turtles have been required to be reported in members' Annual Reports Part 2.

A broad-scale observer programme in Pacific island countries has been in operation since 1995, and was incorporated into a regional observer programme in January 2009. All observer data collected under the regional observer programme are held centrally by the commission. Observer coverage for longline fleets was required to be 5 percent by June 2012, with the exception of some small fleets in the North Pacific that must achieve this coverage by the end of 2014. Compliance with the longline observer coverage requirements has not been verified owing to a lack of submitted data (WCPFC, 2012a; Williams, Cole and Falasi, 2013). Ecological risk assessments were conducted for a comprehensive suite of bycatch species in 2006–09. Largely on the basis of the rich holdings of observer data for some parts of the convention area, indicator analyses and stock assessments are being conducted for five groups of WCPFC key shark species (Clarke and Harley, 2010; Harley, Rice and Williams, 2013).

1.2.1.6 Joint t-RFMOs and other inter-RFMO cooperation

Meetings of the joint t-RFMOs have focused on several themes, one of which is bycatch (Joint Tuna RFMOs, 2010, 2011). The development of minimum bycatch data standards, including data fields to be collected across all RFMOs with a view to allowing interoperability, has been articulated as one of the priority issues (Joint Tuna RFMOs, 2011) and a workshop on purse seine observer data harmonization was held in 2012 (ISSF, 2012a). A similar initiative for longline observer data has been initiated under ICCAT leadership (Nicol, Bunce and Fitzsimmons, 2013).

Another forum for addressing bycatch issues is the International Scientific Committee (ISC), whose members have cooperated to produce two stock assessments for blue sharks based on data held by those members for the North Pacific (Kleiber *et al*., 2009; ISC 2013). Future shark stock assessments are planned by the ISC, but as yet no other bycatch taxa assessments are envisaged.

1.2.2 Other data

The various ecology and bycatch-related subcommittees of the t-RFMOs serve as regional fora for the dissemination and discussion of bycatch issues associated with pelagic longline fisheries. Those whose meeting papers are in the public domain provide a ready source of information on the latest research into population impacts and mitigation methods. Papers presented at t-RFMO fora that do not publish their supporting documents can sometimes be obtained from other sources such as national government websites or the academic literature, and were sourced in this way for this study as much as possible.

In some cases, national bycatch research and monitoring programme requirements exceed those of the t-RFMOs or provide information in the public domain not provided by the t-RFMOs. In such cases, national sources were used to supplement information available from the t-RFMOs. Descriptions of national data in this report may not be representative of the full extent of available national data as information that is available in English and easily located on the Internet was more likely to be reviewed for this study.

Several large-scale research programmes, both completed and ongoing, have made major contributions to the understanding and resolution of bycatch issues, although many of these have focused primarily on other gear types. For example, the four-year project (2008–2012) Mitigating Adverse Ecological Impacts of Open Ocean Fisheries (MADE) project (funded by the European Union [Member Organization]) was designed to develop measures to mitigate adverse impacts of both longline and purse seine gear targeting large pelagic fish, but mainly studied purse seine fisheries (MADE, 2013). This is also largely true of the bycatch research programme conducted by the International Seafood Sustainability Foundation (ISSF 2012b), although some of the research on handling and post-release mortality is relevant to both purse seine and

longline gear. A third major research programme is the Pelagic Fisheries Research Programme running since 1992 at the University of Hawaii. Studies produced by this programme which are relevant to bycatch issues have primarily addressed interactions between longline gear and sea turtles, and characterization of shark bycatch and post-release mortality from longline fisheries (PFRP 2012). In addition to collections of research papers from various sources, there are also several databases of information relevant to bycatch issues. This study has made extensive use of the Bycatch Mitigation Information System (BMIS) maintained by the Secretariat of the Pacific Community for the WCPFC. BMIS is a central repository of information on the mitigation and management of seabirds, sharks and sea turtles in the Western and Central Pacific, but much of the information is relevant to other oceanic fisheries around the world (BMIS 2013). Another bycatch database maintained by the Consortium for Wildlife Bycatch Reduction, which is primarily a bibliography of studies on bycatch reduction methods (CWBR 2013), was also used to identify information sources for this study. Finally, the results of an ICCAT feasibility study on developing a bycatch database were reviewed (Cotter 2010). This study, which was intended to compile both reports and dataset, gathered 372 publications but due to confidentiality restrictions obtained only one new dataset on bycatch (observer data for Mediterranean purse seines in 2003). Since the ICCAT bycatch database has not been made publicly available, its contents could not be utilized for this study.

GLOBAL TRENDS IN LONGLINE FISHERIES

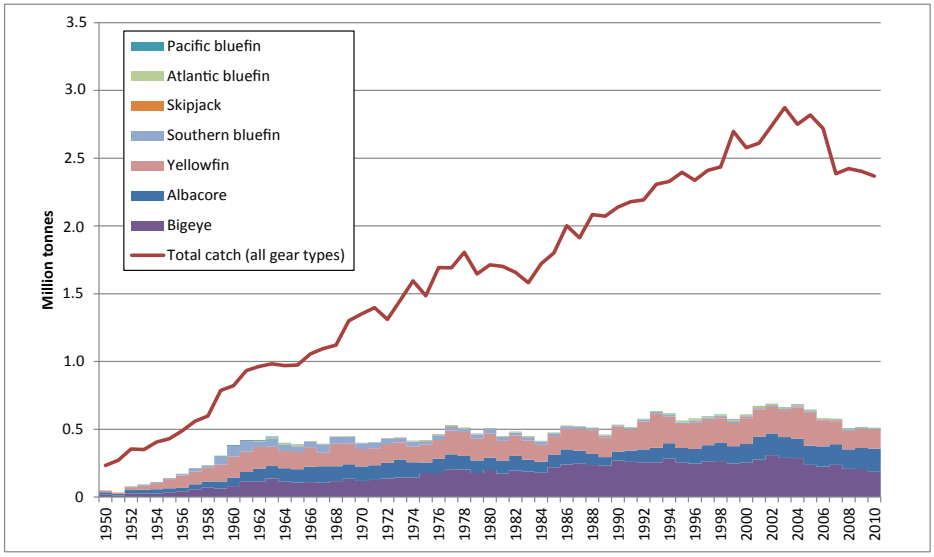
Catch

In this paper, the term "catch" refers to quantities, usually in weight, reported by fishers to management authorities. It should be noted that whether, and the degree to which, these reported catch quantities include individuals that were hooked and discarded (dead), released (alive, but may subsequently die), used onboard for food or bait (dead), escaped (alive but may subsequently die), depredated (presumably dead), or subject to other forms of cryptic mortality (Gilman *et al*., 2013) will vary by fishery, fleet and/or vessel. Therefore, the term "catch" should not be confused with mortality as some catch may survive and some mortalities may not be counted as catch. The term "landings" refers only to that portion of the catch that is brought to port or transshipped at sea. For taxa with no commercial value, the term "catch" is replaced by "interaction" wherever possible.

Total catches of the major market species of tunas by longline fleets worldwide have fallen continuously since 2004 (Figure 3, bars), coinciding with a decline in global tuna catches by all gear types since 2005 (Figure 3, line). In the early 1960s, the proportion of the global tuna catch taken by longline fleets was nearly one-half of the total, but by the mid-1970s this had declined to below one-third and is now about one-fifth.

The primary tuna species caught by longlines are bigeye (*Thunnus obesus*), albacore (*Thunnus alalunga*) and yellowfin (*Thunnus albacares*; Figure 3). These species' share of the total catch of major market tunas by longliners has remained above 95 percent since the mid-1980s. An increase in the proportion of bigeye tuna catches was observed in the mid-1970s as a result of a shift in targeting that occurred at that time (Miyake *et al*., 2010, see Section 1.3.4). In recent years, as total longline catches have fallen with declining catches of bigeye and yellowfin tuna, the proportion of longline-caught tuna that is albacore has risen (Figure 3).

FIGURE 3
Global catches of primary market species of tuna by longline fisheries and total catch of these species by all gear types, 1950–2010

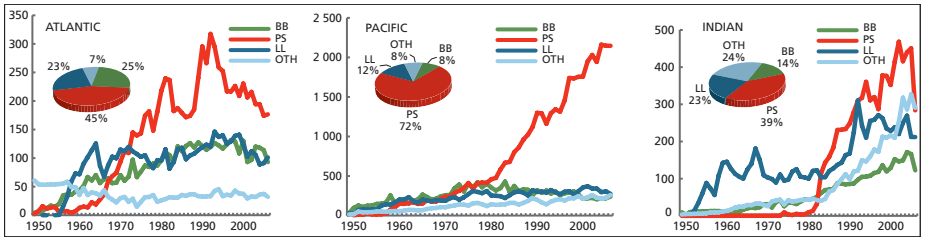


Note: Quantities of skipjack and Atlantic and Pacific bluefin are negligible and may not be visible in some years.

Source: FAO FIGIS database on global tuna catches by stock, 1950–2010

The decline in the share of the global tuna catch deriving from longline fisheries is primarily due to the rise in purse seine fisheries worldwide, particularly in the Pacific where almost 60 percent of the world's bigeye, yellowfin, albacore and skipjack has been taken since 2006 (WCPFC, 2012b; Miyake *et al*., 2010). Longline fishing contributes about 23 percent of the total catch of market tunas in the Indian and Atlantic Oceans, but only about half that proportion (about 12 percent) in the Pacific Ocean (Figure 4).

FIGURE 4
Tuna catch (thousand tonnes) in the Atlantic, Pacific and Indian Oceans by gear type 1950–2007 and percentage for 2007



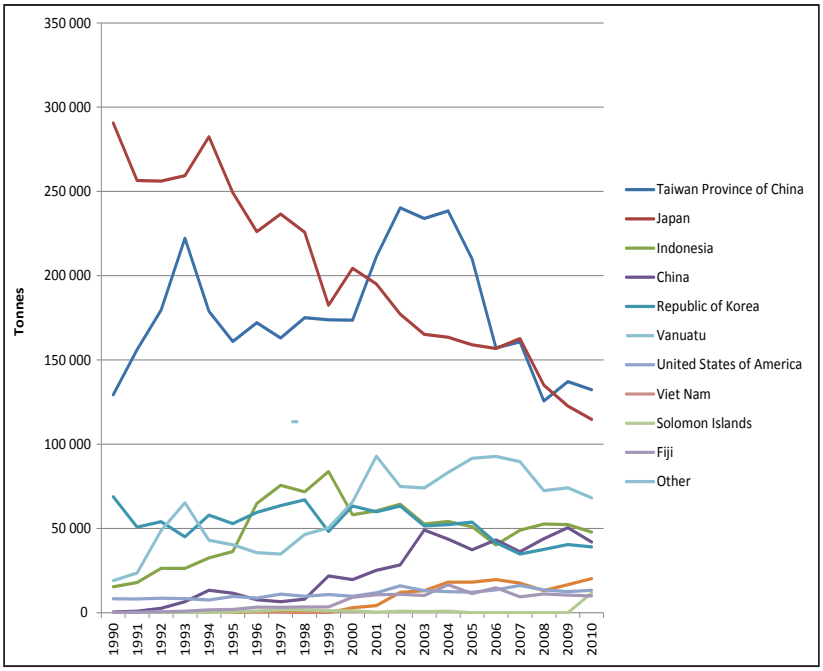
Note: BB = bait boat (pole and line); PS = purse seine; LL = longline; OTH = other.

Source: Miyake et *al.* (2010).

Taiwan Province of China and Japan have long dominated global longline fishing, but the trajectories of these fleets are quite different (Figure 5). While the catch of the major market tunas by the fleet of Taiwan Province of China varied between 125 000 and 340 000 tonnes in 1990–2010, that of the Japanese longline fleet gradually decreased from 290 000 to 115 000 tonnes. Viet Nam, which first reported longline catches of major market tunas in 2010 (about 12 000 tonnes), had the eighth-highest longline catch for that year.[1]

[1] Rather than being due to a sudden commencement of fishing, the appearance of Viet Nam in the tuna longline statistics for the first time in 2010 is probably due to recent improvements in fishery statistical systems.

FIGURE 5
Longline catches of major market tuna species by country, 1990–2010



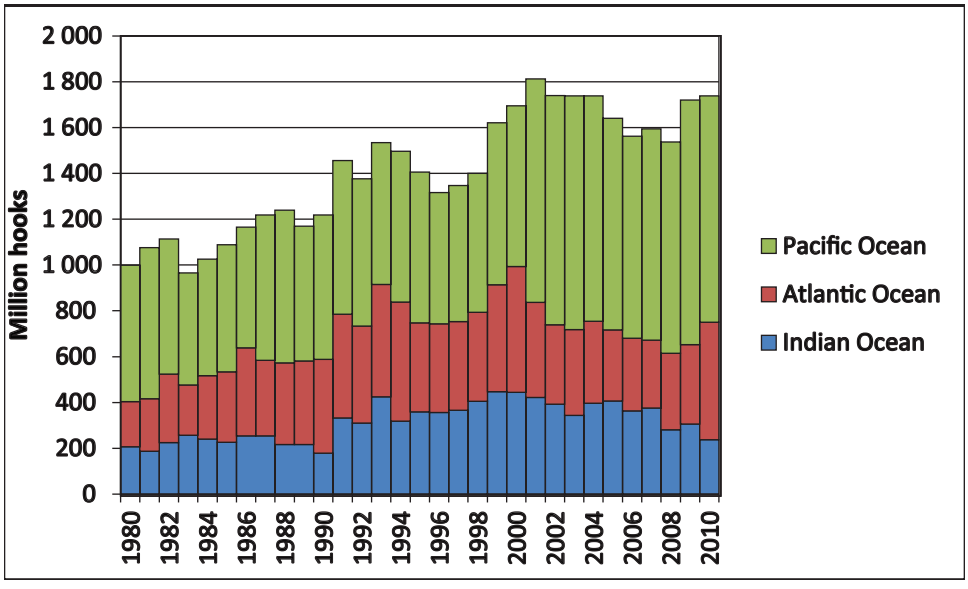
Note: Only the top ten countries in 2010 are shown in the legend; all other countries are aggregated into the "OTHER" category.Source: FAO FIGIS database.

Source: FAO FIGIS database

1.3.2 Effort

Although overall pelagic longline catches have decreased since 2004, global pelagic longline fishing effort as reported by the t-RFMOs followed this trend until 2008 and then increased. The longline fishery in the Pacific is the world's largest longline fishery in terms of effort (about 60 percent of the global total, Figure 6) despite being dwarfed in terms of its total tuna catch by the Pacific purse seine fishery (see Figure 4).

FIGURE 6
Global longline effort by ocean, 1980–2010



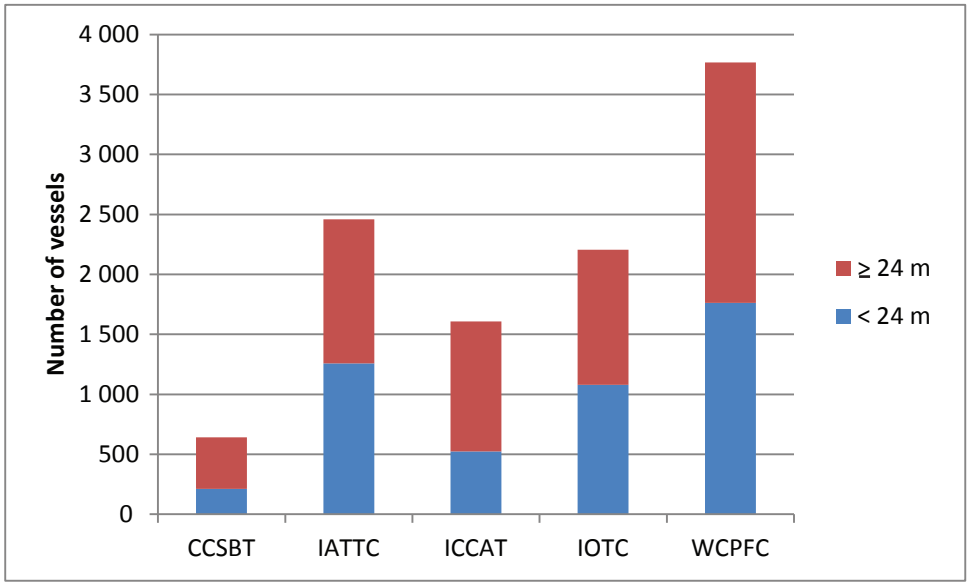
Note: Effort figures for the Pacific include both the Eastern and Western and Central Pacific.
Sources: ICCAT T2CE database (downloaded 13 March 2013; ICCAT, 2013); IOTC (2012b); M. Herrera (personal communication, March 2013); P. Williams (personal communication, March 2013).

Available data on the number of longline hooks fished is incomplete as some vessels and fleets do not report effort in number of hooks. As a result, data gathered from t-RFMOs may be based on estimates and/or be incomplete.

Statistics on pelagic longline vessels are available from an ongoing project of the Joint Tuna RFMOs aimed at producing a global list of authorized tuna fishing vessels (Joint Tuna RFMOs, 2013a). One of the difficulties in producing such a list is that any given vessel may be authorized to fish in the convention area of more than one t-RFMO and thus may be double-counted in a global tally. The project is working to assign unique vessel identifiers, and within the 10 677 'longline' and 'tuna longline' records (Figure 7) there are 7 645 unique vessel identifiers, 4 429 (58 percent) of which are for vessels of less than 24 m. Aside from the likelihood that duplicate records remain in the data set, there are two further shortcomings of using these data to represent the global pelagic longline fleet: some of the vessels on the list may not be active in tuna fisheries; small longliners (≤ 24 m) and those fishing exclusively in national waters that are not required to be reported to some of the t-RFMOs (see Table 1) may be under-represented. Nevertheless, this figure represents the current best estimate of the number of vessels in the global pelagic longline fleet.

FIGURE 7

Number of 'longliners' and 'tuna longliners' < 24 m and ≥ 24 m combined from lists of vessels authorized to fish in each of the five t-RFMOs' convention areas



Note: There have been initial attempts to remove vessels enumerated multiple times, but such double-counted vessels may remain.Source: Joint Tuna RFMOs (2013a).

1.3.3 Value

Although the catch quantity of longline tuna fisheries is considerably lower than purse seine fisheries, the unit value of longline caught tuna is higher. For example, in the Western and Central Pacific, longline catches are only one-seventh the size of purse seine catches, but the value of the longline fishery as a whole is nearly half that of the purse seine fishery (Table 2). Moreover, these statistics suggest that the unit value of longline tuna is more than double that of purse seine or pole and line caught tuna (Table 2)

TABLE 2
Total catch quantity and value of three tuna fisheries in the Western and Central Pacific Ocean for 2012

Fishery	Catch	Value	Price
	(tonnes)	*(US$ million)*	*(US$ per tonne)*
Purse seine	1 816 503	4 054	2 232
Pole & line	224 207	586	2 614
Longline	262 076	1 962	7 486

Source: Williams and Terawasi (2013).

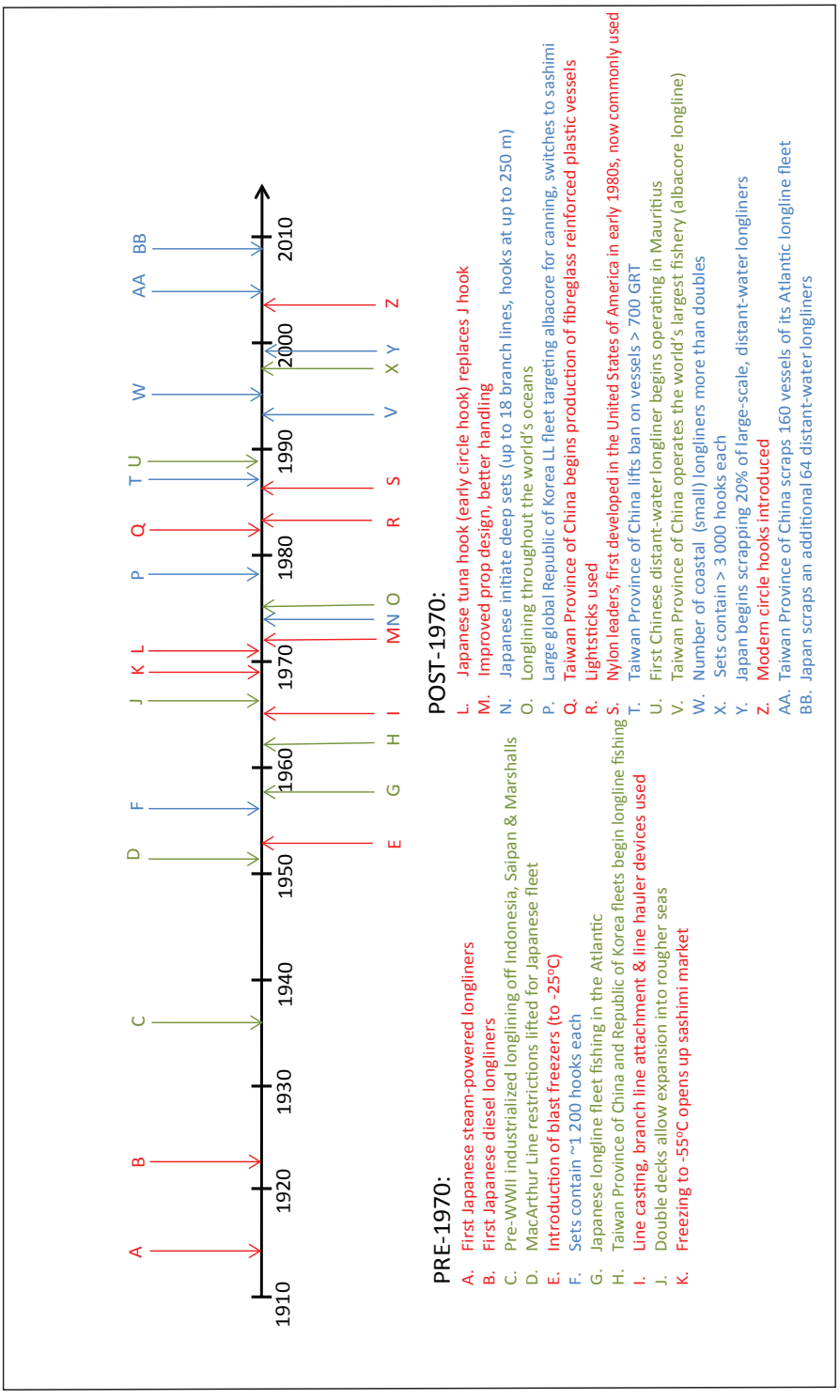
As Miyake *et al*. (2010) caution, it is problematic to compare market prices between various forms of tuna products owing to differences in packaging weights (e.g. cans versus loins), and yields (i.e. conversion ratios from whole to processed weights). Nevertheless, as most of the purse seine catch is canned and most of the longline catch, with the exception of albacore, is not, it appears valid to conclude that tuna for canning produced by the purse seine and pole and line fisheries are worth less per unit weight than tuna for the sashimi (and sushi) market caught by the longline fishery.

1.3.4 Technological and operational changes in longline fisheries

The early history of pelagic longline fishing was greatly influenced by the development of Pacific longline fleets, particularly the Japanese longline fleet, and the diffusion of its innovations to other fishing grounds and fleets. This section summarizes this history and introduces the new technologies, fishing grounds and operating practices that have most radically affected fishing efficiency and potential interactions with bycatch organisms (Figure 8; Miyake, 2005; Ward and Hindmarsh, 2007; Ward, 2008; Miyake *et al*., 2010; M. McCoy, personal communication, March 2013).

Internal combustion engine-powered vessels ("yakidama" or semi-diesel) existed in the Japanese longline fleet prior to the Second World War, but these vessels were still reliant on ice-based freezing technologies. Despite this limitation, the Japanese fleet had developed "industrialized" fishing operations, including the involvement of motherships, at fishing bases located in Indonesia, the Philippines, the Marshall Islands and Saipan (Miyake, 2005). These large-scale distant-water Japanese fishing activities collapsed at the end of the Second World War with implementation of the Potsdam Treaty of August 1945. This treaty initially proscribed all Japanese fishing activity but was quickly relaxed to allow fishing within 12 nautical miles of the Japanese coast. Progressive extension of the outer limit of the fishing grounds accessible to Japan, known as "MacArthur Lines", occurred over the following few years (1946–49) and Japan's far seas longline fishery was fully re-launched in 1952 when these restrictions were completely lifted (Okamoto, 2004; Gillett, 2007).

FIGURE 8
Major developments in longline fishing, 1910–2010



Note: Red = implementation of new technologies; green = expansion of fishing grounds; blue = operational characteristics of the fleets.

Sources: Suzuki, Warashina and Kishida (1977); Okamoto, 2004; Miyake, Miyabe and Nakano (2004); Miyake (2005); Miyake *et al*. (2010); Ward and Hindmarsh (2007); Ward (2008) and M. McCoy (personal communication, March 2013.

The introduction of blast freezers in 1953 signalled a new era in the longline fishery as it allowed for unprocessed product to be stored before being brought back to Japan, extending the one-week holding period allowed by ice-based technology. However, tuna thawed after being frozen in this manner had a brownish colour and thus most

of it was used for canning (Miyake, 2005). By the mid-1950s, Japanese longliners began fishing in the Atlantic, and Taiwan Province of China and Republic of Korea longline fleets began to develop and fish more widely in the Pacific in the 1960s. Better mechanisms for gear handling, such as line casting, branch line attachment and line hauler devices, were developed in the mid-1960s and resulted in higher efficiencies through faster hauling and less crew labour (Ward and Hindmarch, 2007; Miyake *et al*., 2010).

A major innovation occurred just prior to 1970 with the development of flash freezing (–55 °C) and super cold (minimum –40 °C) storage facilities, which allowed offshore or distant-water catches to be used for sashimi for the first time. This led many longliners to shift away from targeting albacore and yellowfin tunas for canning towards targeting bluefin, bigeye and yellowfin tunas for sashimi in subsequent years. Prior to this time, hooks were set shallower than 120 m, but using a new form of deep-setting, first developed by the Republic of Korea fleet, hooks were fished as deep as, or later deeper than, 250 m in order to better target bigeye tuna for the sashimi market (Suzuki, Warashina and Kishida, 1977; Ward and Hindmarsh, 2007). Developments in the early 1980s were mainly in the field of materials, including the introduction of lightsticks and nylon leaders, which led to increased catches of bigeye tuna and swordfish. Similarly, in the fleet of Taiwan Province of China of relatively small-scale longliners, the conversion of wooden vessels to fibreglass reinforced plastic facilitated the migration of these vessels to coastal waters throughout the world.

Materials science innovations had occurred and been disseminated by the 1990s to the point where not only branch lines but also mainlines could be made of nylon or other synthetic materials (e.g. kuralon), allowing for a wide variety of targeting strategies (Ward and Hindmarsh, 2007; Gilman *et al*., 2007a). By the late 1990s, the number of hooks per set had expanded to more than 3 000 from about 1 200 in the 1950s (Ward and Hindmarsh, 2007). At this point, acknowledging that fishing capacity had exceeded sustainable levels, Japan and later Taiwan Province of China began buying back licences and scrapping a portion of their distant-water longline fleets. Concurrent with this trend towards reduction of the large-scale longliner fleets, the contribution of small coastal longliners, particularly in the Pacific, to the global tuna catch substantially increased from about 15 percent in the early 1970s to about 35 percent in the late 2000s (Figure 9).

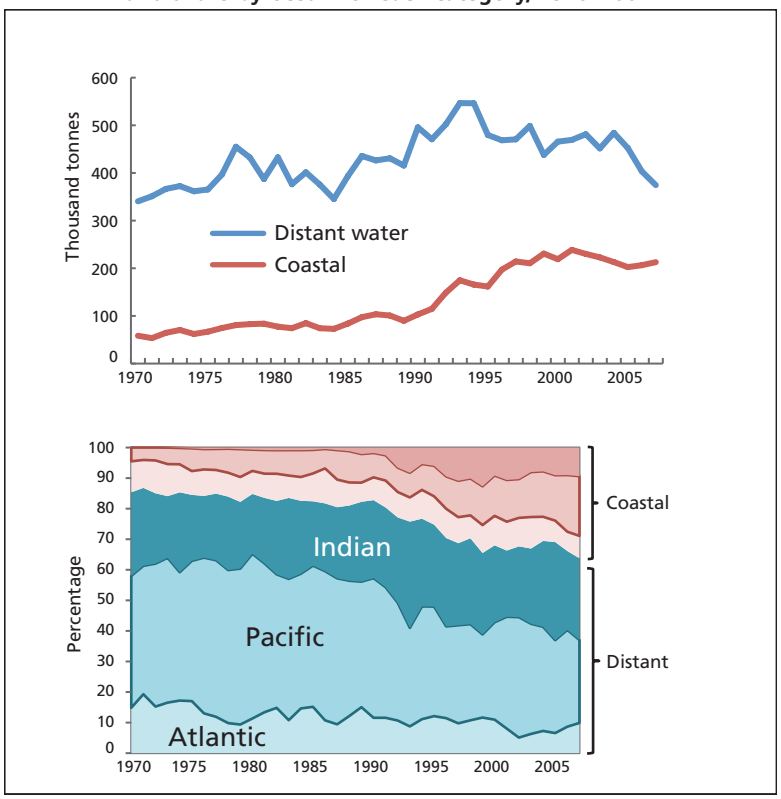
Miyake *et al*. (2010) characterized fishery development up to 2000 as being singularly focused on increasing fishing efficiency, and the subsequent period (i.e. post-2000) as one of looking beyond minimizing cost and maximizing production and coming to terms with ecosystem and sustainability considerations. A prime example of this is the emergence and popularization of the modern circle hook as a means of reducing bycatch of threatened species while maintaining catch rates of target species. This and other technologies and operating methods that have been developed in the past decade to address bycatch interactions with longline fisheries are the subject of the following chapters.

1.3.5 Other factors influencing trends in longline fishing

In addition to the technological and operational developments in longline fisheries described above, a number of other factors are now shaping the fishery. Although the influence of these policy, economic and socio-economic factors is less well understood, their effects may be significant.

The most important non-technical factor influencing longline fishing is arguably fishery management regulations. A comprehensive description of management measures applicable to global tuna fisheries is provided in Miyake *et al*. (2010), and in independent performance reviews that have been conducted for four of the five t-RFMOs (Joint Tuna RFMOs, 2013b; WCPFC, 2012c). To date, most of the

FIGURE 9
Coastal and distant-water longline catch of major tuna species worldwide and share by ocean for each category, 1970–2007



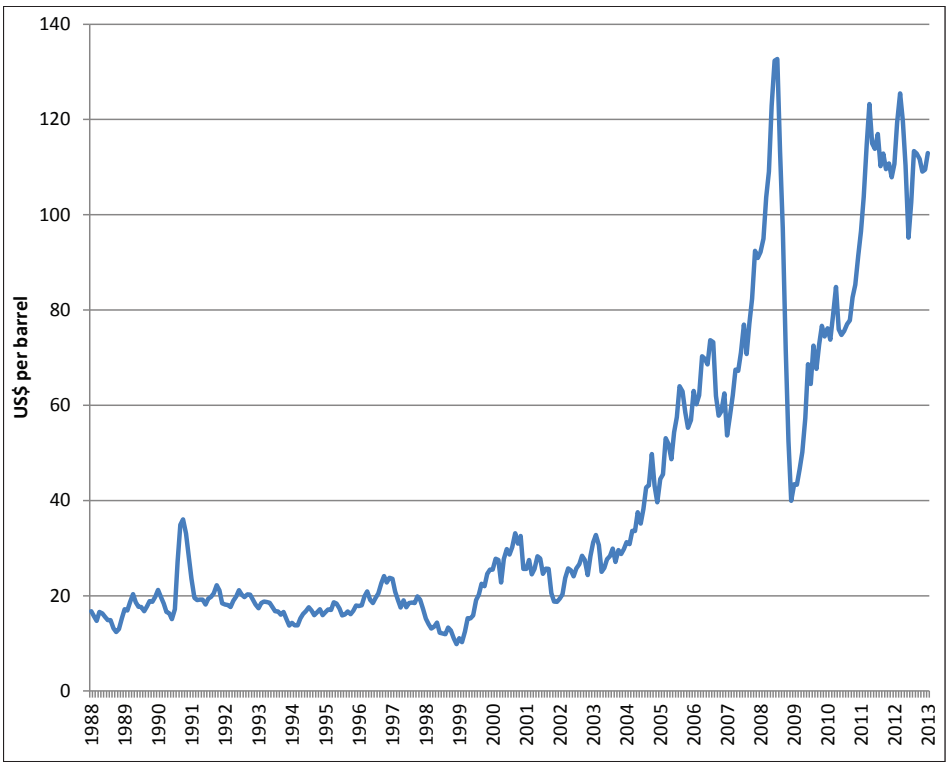
Source: Miyake *et al.* (2010)..

management measures have been designed to control target tuna catches (e.g. closures, effort or catch limits, size limits for target species) but they also cause changes in fishing behaviour that directly or indirectly affect bycatch interactions. Rather than discussing tuna management measures as a whole, this report discusses all management measures that affect bycatch interactions for each of the taxa in the following chapters (Chapters 2–6).

Another critical non-technical factor shaping longline fisheries is the price of oil (Figure 10). Fuel costs have been described as the most important influence on the tuna fishing industry in recent years (Miyake *et al.*, 2010). Because longline vessels are estimated to burn on average 1 070 litres of fuel per tonne of tuna landed versus purse seiners that burn on average 368 litres per tonne (Tyedmers and Parker, 2012),[2] and because this is only partially offset by the higher value per kilogram of the longline catch, the rise in oil prices in the last decade has contributed to the growth of the purse seine fleet at the expense of the longline fleet (Miyake *et al.*, 2010). A case study analysis showed that fuel costs for offshore longliners increased 160 percent from 2004 to 2006 and rose from 7 percent of total operating costs in 1994 to 23 percent of operating costs in 2006 (Miyake *et al.*, 2010). Current oil prices are about US$50 per barrel higher than they were in 2006. Such higher operating costs could prompt moves toward improved fuel efficiency through better maintenance and/or new energy-efficient equipment. However, unless these improvements resulted in major savings, further contraction of fishing grounds, greater use of transshipment, and the exit of unprofitable vessels from the fishery would be necessary to raise the overall fuel efficiency of the fleet.

[2] The authors note that these estimates do not account for fuel expenditure associated with the provision of bait.

FIGURE 10
Monthly spot prices for petroleum (Europe Brent Crude), January 1988 – January 2013



Source: U.S. Energy Information Administration (http://tonto.eia.gov/dnav/pet/hist/LeafHandler.ashx?n=PET&s=RBRTE&f=M)

The escalating cost of labour has also been a critical factor for some longline fleets. Despite the implementation of mechanized setting and retrieval devices, labour costs still comprise a major share of the operating expenses of longline fleets. In many East Asian longline fleets, labour costs have been trimmed by substituting foreign crew for nationals with concomitant changes in wage structures and policies (McCoy and Ishihara, 1999; Miyake *et al*., 2010; Stringer, Simmons and Coulston, 2011). These changes can in turn affect bycatch interactions through changes in utilization or handling. For example, one study documented that in some cases foreign and other lower-paid crew were allowed to retain revenues generated from shark fins as a bonus (McCoy and Ishihara, 1999). Other anecdotal examples of revenues from shark fins being used to supplement wages have also been noted (personal observation). Although this information was reported prior to implementation of various controls on shark finning, similar situations may still arise in areas where there are limited or no regulatory constraints, and where crew pay structures incentivize retention.

Another clear influence on the longline fishery is the raw material price of tuna. Although large-scale longline vessels focus on the most sought-after tuna species and have state-of-the-art freezer technology, their products do not always command enough of a price premium to compensate for their higher operating costs. As a result, the products of large-scale longline fishing are often out-competed by the products of other tuna fisheries with lower operating costs, particularly purse seine fisheries that produce tuna for canning. It has been predicted that the current trend of shrinkage in the world's large-scale longline fleet will continue unless market price trends for the different forms of tuna products reverse (Miyake *et al*., 2010). These market prices will in turn depend on socio-economic factors such as the global economy, dietary preferences and the pricing of substitutes (including farmed fish).

A final issue when considering the trends affecting pelagic longline fisheries is the use of bait. Although bait is not considered as bycatch in this study, some researchers consider it a form of discard associated with the fishery (e.g. Ardill, Itanoa and Gillett, 2013). This issue has recently received attention when contrasting purse seine fisheries, which do not use bait, and pole and line fisheries, which are estimated to catch 32 kg of skipjack tuna for every 1 kg of bait fish used (i.e. 0.03:1 bait-to-target ratio [Gillett, 2010]). Estimates for longline fisheries suggest that the amount of bait used amounts to about half of the weight of target species (i.e. 0.5:1 ratio of bait-to-target tuna catch weight) (Bach *et al*., 2012; Ardill, Itanoa and Gillett, 2013).[3] Concerns about indirect impacts on the ecosystem and energy inputs associated with bait usage appear, therefore, to be as, if not more, applicable to longline fisheries than to pole and line fisheries. In addition, as described in the following chapters, changes in the choice of bait species that may be motivated by bait price, as well as by fishers' preferences or regulations, may significantly influence interactions with bycatch species.

[3] This 0.5:1 bait-to-target ratio for longline fisheries compares favourably with tuna farming feed conversion ratios of 10–20 kg of feed for every 1 kg of tuna produced (10–20:1 [Ottolenghi, 2008]).

2. Elasmobranchs

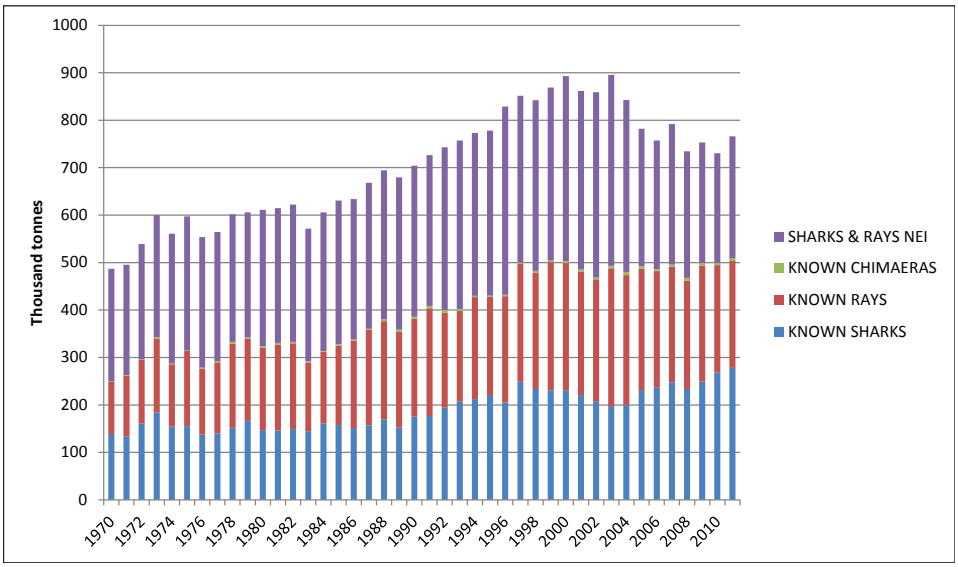
2.1 OVERVIEW OF ELASMOBRANCH INTERACTIONS WITH LONGLINE GEAR

2.1.1 Background, definitions and concepts

Elasmobranchs are a group of more than 1 000 species of sharks and rays that constitute a substantial proportion of the catch of pelagic longline fisheries. Chimaeras, also known as elephantfish or ratfish, are classified along with the elasmobranchs as chondrichthyan fishes but they inhabit very deep water and are rarely caught by pelagic longlines. While considerable energy could be expended debating whether elasmobranchs are target species, they will be discussed in this study regardless of whether they are intended or associated catch, and irrespective of whether they are retained, whole or in part, or released.

Despite the fact that, unlike most bycatch species discussed in this paper, sharks and rays may be commercially valuable, elasmobranch catches have long suffered from under-reporting and non-reporting owing to their relatively low value (compared with recognized target species) and lack of reporting requirements. As a result, although catch statistics do exist, there is a large degree of uncertainty associated with them. Reporting biases over time, in the form of a greater propensity to report shark catches or to report them to species, may also affect these data (see below). It is therefore important to examine global capture production figures compiled from national authorities by FAO with these caveats in mind. The FAO-compiled figures suggest that elasmobranch production reached a peak in 2003 at 896 000 tonnes and declined by 14 percent to 766 000 tonnes by 2011 (Figure 11). Although it is unlikely that these figures accurately represent an absolute measure of the actual biomass fished, they may be useful as a relative index of elasmobranch catches.

FIGURE 11
Shark, ray and chimaera capture production, 1970–2011



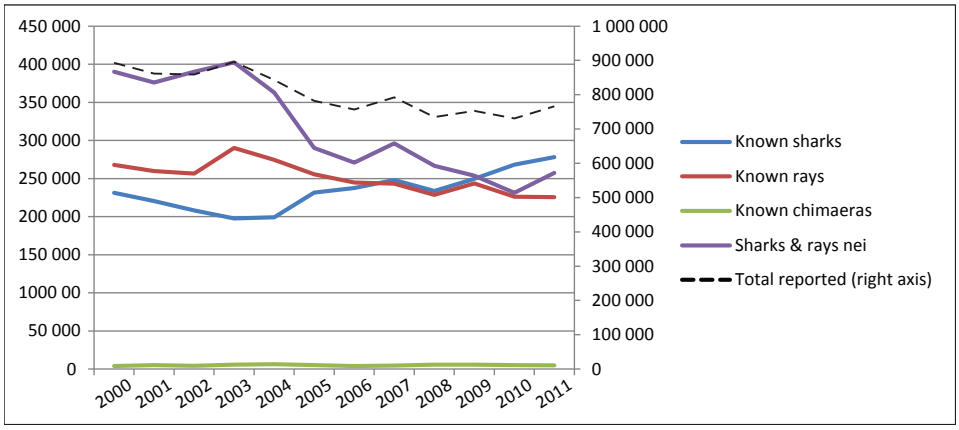
Note: "Sharks & rays nei" refers to elasmobranchs that were "not elsewhere indicated", i.e. not identified to a more specific taxonomic category.
Source: FAO FISHSTAT Capture Production Dataset (FAO, 2013a).

Several studies have attempted to quantify annual catches of sharks (only) and can be used to gauge the extent of potential under-reporting in the FAO database. Estimates of sharks utilized for the global shark fin trade were constructed through modelling of the size and weight of sharks associated with fins observed at auctions in China, Hong Kong SAR over an 18-month period spanning 1999–2001 (Clarke *et al*., 2006). These results found that the actual capture production biomass represented by the shark fin trade alone in 2000 was 1.7 million tonnes (range of 1.2–2.3 million tonnes). This study calculated that FAO capture production of species that could be used in the shark fin trade for 2000 was 0.4–0.6 million tonnes, and was therefore under-reported by at least a factor of three. Other estimates of shark catches for 2000 based on extrapolation of published catch and effort statistics and assumptions about illegal take, finning[4] and post-release mortality indicate that total mortality was of the order of 1.4 million tonnes (no confidence interval given; [Worm *et al*., 2013]). A third study based on catch reconstructions calculated that shark catches in the period 2000–09 were 0.57 million tonnes ± 0.11 million tonnes per year (Biery, 2012).These three studies all point to under-reporting of sharks in the FAO shark capture production data.

In addition to being conscious of potentially severe under-reporting biases in available elasmobranch data, it is also important to be aware of changes in the specificity of taxonomic reporting. In the FAO data set for the early 2000s, the tonnage reported in taxonomic categories that are identifiable as sharks was about half that of the tonnage unreported in unidentified elasmobranch categories. By 2009, these two quantities were about equal (Figure 12), although it should be noted that many of the reporting categories that can be identified as sharks are still not species-specific.

FIGURE 12

Reported capture production of known sharks, rays and chimaeras, and an unidentified group of sharks and rays, contrasted with the total of all identified and unidentified chondrichthyans, 2000–2011



Source: FAO (2013a).

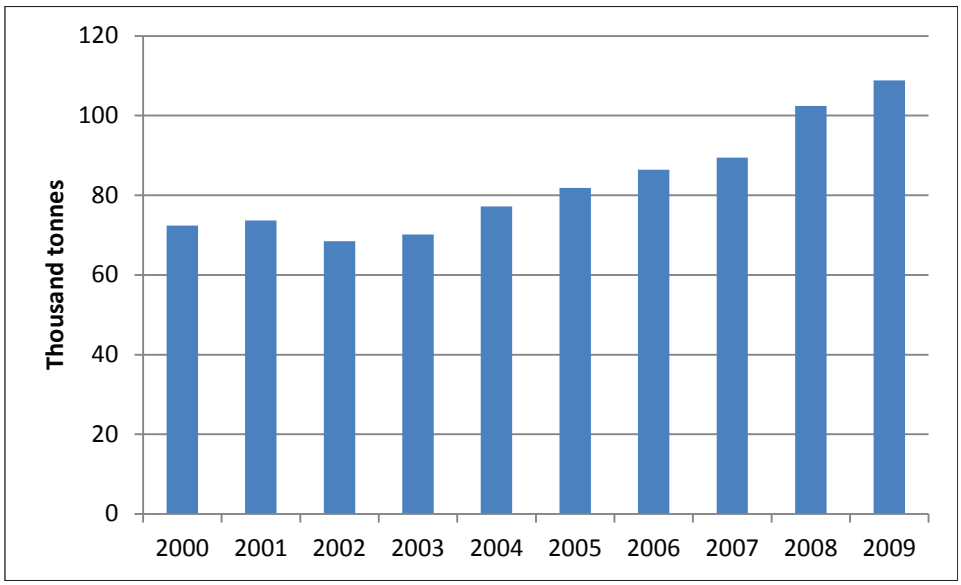
One interpretation of this trend assumes that taxonomic reporting practices have remained consistent and thus the increasing catches in identified shark species (versus "sharks and rays nei") reflect real increases in catches of sharks. A competing hypothesis is that the increase in reported shark catches is a result of a move toward greater taxonomic specificity such that the actual number of sharks caught has not increased, but simply that they are now being identified as sharks rather than "sharks

[4] In line with standard international usage, finning is defined in this paper as the practice of removing and retaining shark fins and discarding the remainder of the carcass at sea (IUCN, 2013b).

and rays nei". Consistent with this hypothesis, the reported catches of "sharks and rays nei" have declined, but so too have the reported catches of known rays. In any case, it is clear that the overall trend for the total of all categories has been downward since 2003.

Another factor that can have a major effect on shark catch statistics is whether or not full utilization is practised. Sharks, in particular, are more likely to be reported as catch when their carcasses are utilized for meat than when they are finned or discarded whole. An increasing trade in shark meat is suggested by global export trends for shark meat, which show growth of 50 percent since 2000 (Figure 13). This in turn suggests another possible explanation for the increase in shark capture production quantities shown in Figure 12: if there is a growing trade in shark meat, perhaps motivated by requirements for full retention (see Section 2.3.2), this is likely to be accompanied by an increase in recording of the full weight of the shark catch rather than merely the weight of its fins. The suggestion that the increase in traded quantities is being driven by an increasing value placed on shark meat is refuted by annual median market prices for shark meat ("quenlla") in Vigo, Spain, in the period January 2001 – January 2013,

FIGURE 13
Quantity of reported exports of shark meat products for all countries, 2000–09



Source: FAO (2013a).

which varied in a narrow range between EUR1.36 and 2.09/kg (Pesca de Galicia 2013).

Changes in consumer demand for shark fins can also exert an influence on shark catch statistics. In 2013, some international environmental organizations asserted, based on an observed decline in shark fin imports into China, Hong Kong SAR in 2012 of 70 percent (Table 3), that there had been a shift in consumer demand away from shark fins owing to conservation concerns. However, there are a number of other potential influences on these figures that must be considered. These include the implementation of new commodity codes in China, Hong Kong SAR and other countries in 2012 in response to changes by the World Customs Organization and the Government of China's ongoing campaign against conspicuous consumption (Tsui, 2013). It is probable that several of these influences, some of which may be temporary and not related to consumer demand, combined to depress shark fin imports into China, Hong Kong SAR in 2012. Whether these trade trends also reflect reduced shark catches in 2011–12 will only be seen once catch data become available.

TABLE 3
Imports of dried and frozen shark fins by China, Hong Kong SAR, 2008–2012

	Dried shark fins with cartilage (coded as 0305-5950 until 2012, then 0305-7111)	Adjusted frozen shark fins with cartilage (coded as 0305-5960 until 2012, then 0305-7112)	Shark fins NESOI (0305-7190, new code introduced in 2012)	Total adjusted shark fins with cartilage
		(tonnes)		
2008	4 131	1 405	–	5 536
2009	4 328	1 231	–	5 559
2010	4 522	1 237	–	5 759
2011	4 907	1 268	–	6 175
2012	3 117	47	0	3 164

Source: Hong Kong Government Census and Statistics Department (unpublished data).

Preservation and storage issues (see Section 1.3.4) may also be playing an important role in the quantities of sharks retained and recorded. The use of sharks for their meat is complicated by the need to bleed, wash and ice or freeze carcasses in order to avoid the high concentration of urea present in the blood and tissues from degrading into ammonia. Historically, this has led to a preference for smaller species to be used for meat as they generally have lower urea concentrations and are also easier to process (Vannuccini, 1999). However, there are now active markets for meat from a variety of species, including low-grade blue sharks, in countries such as Spain, Brazil and Japan. On some vessels, fishers take advantage of improved freezing technologies to quick

PLATE 1
Pelagic sharks caught by a longline fishery based in Su'Ao, Taiwan Province of China. These sharks are stored onboard using only ice-based preservation methods and are landed whole and fully utilized

Courtesy of E. Gilman.

freeze sharks, but in some fisheries brine wells, ice with plastic bags to separate sharks, and even no preservation are still used to bring shark meat to market (Plate 1).

Some vessels that freeze sharks, and are required by regulations to land sharks with their fins attached, cut fins prior to stacking sharks so that they can be folded onto the carcass and kept flush with plastic ties. More evidence for increasing refrigeration of shark catches is provided by imports into China, Hong Kong SAR of shark fins in "salted" (frozen) form, which doubled between 1999 and 2002, and as of 2006 weighed more than the imported weight of dried fins (not adjusting for water content). Whether

these changes in preservation and storage are part of the cause or the effect of the observed increase in shark meat exports (Figure 13) cannot be determined on the basis of available data.

2.1.2 Factors influencing elasmobranch interactions and mortality

Before examining pelagic longline fishery interactions with elasmobranchs in each ocean, a number of factors that can influence the extent of these interactions are identified. These factors can be classified into three phases of potential interaction discussed below – conditions in the fishing grounds, fishing gear and its deployment, and shark handling at the vessel – as well as species-specific factors. All of these factors influence catch rates, hooking mortality (also referred to as haulback mortality), retention practices and post-release mortality, which together determine the full extent of the impact of the fisheries on shark populations. As described in Section 2.1.3 below, few species of rays are known to interact with pelagic longline fisheries in substantial numbers. Therefore, the following discussion is based mainly on literature pertaining to sharks but is expected to also describe factors determining interactions with rays.

2.1.2.1 Conditions in the fishing grounds

A prerequisite for elasmobranch interactions is that these species be present in the fishing grounds. Global ocean mapping of the distribution of all shark species has shown that species richness peaks at mid-latitudes similar to patterns found for tunas and billfishes (Lucifora, García and Worm, 2011). In turn, the number of tuna and billfish species has been found to be highest not at specific latitudes but in areas with highly productive oceanographic fronts (Worm *et al*., 2005). With regard to abundance rather than species richness *per se*, pelagic sharks are known to congregate at the boundaries of thermal fronts (Bigelow, Boggs and He, 1999) and in areas of upwelling above seamounts (Gilman *et al*., 2012), although some species have also shown sex-specific segregation patterns elsewhere (Mucientes *et al*., 2009). Therefore, although sharks and tunas are generally expected to prefer similar habitats, infrequent high concentrations of sharks anecdotally reported by tuna fishers may be influenced by distributional patterns or preferences specific to sharks such as mating, pupping or nursery grounds.

2.1.2.2 Fishing gear and deployment

In addition to oceanographic features that influence whether sharks are present at a particular latitude, longitude and depth, longline gear and its deployment may also have a large effect on the number and lethality of shark interactions. For example, whether the hooks are set in shallow (< 100 m) or deep (> 100 m) water, or whether the set is made during the day or at night, can have consequences for the relative vulnerability of shark species with different habitat preferences (Gilman *et al*., 2008). It is possible to infer such preferences by species from a combination of fishery-dependent and independent studies. For example, oceanic whitetip shark (*Carcharhinus longimanus*) are often found in the epipelagic zone, blue shark (*Prionace glauca*) range widely throughout the epipelagic and mesopelagic zones, and thresher sharks (*Alopias* spp.) are found at depth during the day but forage near the surface at night (Boggs, 1992; Bonfil, Clarke and Nakano, 2008; Nakano, Matsunaga and Hiroaki, 2003; Ward and Myers, 2005a; Weng and Block, 2004). However, it may be unwise to use such generalized preferences to predict sharks interactions for shallow (e.g. swordfish-targeting) and deep (e.g. bigeye-tuna-targeting) set fishing. This is not only because the longline gear may not always fish at the intended depth and must pass through shallow habitats even when targeting deep ones (Boggs, 1992), but also because depth may be less important in determining shark vulnerability to longline gear than environmental variables such as thermocline and oxygen gradients (Bigelow and Maunder, 2007).

Other characteristics of longline gear and its deployment are important in determining whether a shark becomes hooked and remains alive in the water after it takes the hook. Most, if not all, bait types used to catch tunas and billfishes will also attract sharks to the longline gear. However, details of what factors cause sharks to pursue and strike baits, including the relative importance of chemical and visual cues, are not well understood (Jordan *et al*., 2013). Some longliners will attach short leaders to floats baited with fish meat (sometimes shark meat) specifically to catch sharks (Bromhead *et al*., 2012). It is not known whether these baits are more attractive to sharks or simply less attractive to other species. Once the shark bites the hook, the soak time and length of the branch line may affect mortality rates by influencing the amount of stress the shark experiences, particularly in terms of asphyxiation potential (Erickson and Berkeley, 2009; Campana, Joyce and Manning, 2009; Heberer *et al*., 2010; Carruthers, Neilson and Smith, 2011; Braccini, Van Rijn and Frick, 2012). Water temperature may also play a role in determining survival rates for hooked sharks as it has been suggested that lower temperatures might reduce locomotor activities and associated stress (Moyes *et al*., 2006). The size and shape of the hook may determine where it lodges in the shark and whether the shark suffers further trauma during haulback or can bite through the leader and escape (Ward *et al*., 2008; Carruthers, Schneider and Neilson, 2009; Afonso *et al*., 2011; Godin, Carlson and Burgener, 2012). As discussed further in Section 2.4.2, all of these factors may vary by species and/or size of the shark.

2.1.2.3 Handling practices

Arguably the most unequivocal determinant of mortality is how the shark is handled once it is brought to the vessel. In some situations, regulations require the release of sharks in an unharmed state, but these regulations apply to only a small proportion of total shark interactions and there is usually no specific handling guidance (see Section 2.3). Unless prohibited, across many fisheries sharks will be retained, whole or in part, for their meat or fins (Clarke *et al*., 2013). Mortality rates for these sharks would clearly be 100 percent, but the proportion of the shark catch that is retained or finned versus discarded is often not documented. For those sharks that are alive when brought to the vessel and then discarded without being finned, mortality rates are uncertain owing to the variety of handling techniques used. In a wide-ranging survey of shark handling techniques, sharks were found to be discarded by relatively benign methods, such as cutting the leader, and by relatively harsh methods such as cutting the hook from shark's mouth or pulling the hook out by force in order to retrieve the terminal tackle (Gilman *et al*., 2008). Other fisheries report that some crews will pull out hooks, occasionally removing the jaw, or body gaff sharks during gear retrieval (Campana *et al*., 2009a). Given these differences in handling practices, it is not surprising that studies of post-release mortality to sharks have generated estimates ranging from about 5 percent (Moyes *et al*., 2006, Musyl *et al*., 2009, Musyl *et al*., 2011) to 19 percent (Campana, Joyce and Manning, 2009, Campana *et al*., 2009b). While these studies provide some insights into the potential range of mortality rates for sharks hauled to the vessel and released alive, the proportion of sharks across all longline fisheries that are handled in ways that appear to maximize, or conversely minimize, their chances of survival is unknown.

2.1.2.4 Species-, sex- and size-specific survivorship

An overarching factor in the survivorship of sharks interacting with surface longline fisheries appears to be related to resilience characteristics that will vary from one shark to another. One study in the Pacific showed that hooking mortality (i.e. mortality prior to reaching the vessel) varied by species over a wide range – blue and oceanic whitetip sharks suffered mortality at rates of < 6 percent each whereas mortality rates for

crocodile and thresher sharks were 25–67 percent (Musyl *et al*., 2011). Another study of an Atlantic longline fishery that controlled for differences in fishing area and gear found that for four species of sharks mortality rates differed by size and sex such that females and larger specimens were more likely to survive (Coelho *et al*., 2012). Further analysis of Atlantic blue sharks in particular revealed that other than differences arising from annual variation, size was the most important factor determining whether the shark survived haulback (Diaz and Serafy, 2005; Coelho, Infante and Santos, 2013). A study of six species of Atlantic coastal sharks provides further confirmation of this effect in the form of a significantly higher at-vessel survival for older age classes in some species (Morgan and Burgess, 2007). However, the opposite result was found in a study of common thresher shark (*Alopias vulpinus*) in a recreational fishery where smaller individuals had higher survival rates, perhaps owing to the fact that they were hauled in more quickly (Heberer *et al*., 2010).

2.1.3 Species risk profiles

A number of elasmobranch species that interact with pelagic longline fisheries have been identified by international conventions and/or in scientific studies as being at risk. These species are briefly introduced below before discussing their status in each ocean in the following section.

Prior to the most recent Conference of the Parties (COP), the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) had listed a number of elasmobranch species on Appendices I or II,[5] but most of these species, i.e. basking shark (*Cetorhinus maximus*), whale shark (*Rhincodon typus*), great white shark (*Carcharodon carcharias*) and six species of sawfishes (Pristidae), do not frequently interact with pelagic longline fisheries. The remaining, previously listed species, porbeagle (*Lamna nasus*) and scalloped hammerhead (*Sphyrna lewini*) sharks, were included on Appendix III[6] in September 2012 by some countries of the European Union (Member Organization) and by Costa Rica, respectively. At the COP in March 2013, a number of species that are frequently caught by longline fisheries were listed on Appendix II, which applies international trade controls to all countries that are parties to CITES and do not take out a specific reservation to the listing. These recent listings include oceanic whitetip (*Carcharhinus longimanus*), scalloped hammerhead (*Sphyrna lewini* and lookalikes great [*S.* mokarran] and smooth [*S.* zygaena] hammerheads), porbeagle (*Lamna nasus*) sharks, the manta rays (*Manta birostris* and *M.* alfredi) and freshwater sawfish (*Pristis microdon*). All of these listings are scheduled to enter into force 18 months from the close of the COP, i.e. in mid-September 2014.

The Convention on the Conservation of Migratory Species (CMS) has also designated three elasmobranch species as either threatened with extinction (CMS Appendix I; basking shark, great white shark and manta ray) or able to benefit significantly from international cooperation (CMS Appendix II; basking shark, great white shark, whale shark, shortfin and longfin makos [*Isurus* spp.], porbeagle, northern hemisphere spiny dogfish [*Squalus acanthias*] and giant manta [*M.* birostris]). Of these eight, only the shortfin and longfin makos and the porbeagle, all of which are listed on Appendix II, interact frequently with longline fisheries.

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[5] CITES Appendix I species are those that are threatened with extinction and are or may be affected by trade. For these "black list" species, trade for "primarily commercial purposes" is prohibited. CITES Appendix II species are those that are not now necessarily threatened with extinction but may become so unless trade is strictly regulated. For these "grey list" species, trade is allowed subject to conditions (Reeve, 2002).

[6] CITES Appendix III species are those that are listed unilaterally by parties as being subject to regulation within their jurisdiction and for which international cooperation is needed to control trade. Exports originating in the listing country require certification of origin and legal provenance. Exports originating in a country other than the listing country require a certification of origin only (Reeve, 2002).

In July 2012, ten elasmobranch species were listed on Annex II of the Barcelona Convention, which sets protocols for specially protected areas and biological diversity in the Mediterranean. Of the listed species, shortfin mako, porbeagle, smooth, scalloped and great hammerhead, and tope (*Galeorhinus galeus*) sharks are known to interact frequently with longline fisheries. With this listing, these shark species can no longer be captured or sold, and plans for their recovery must be developed (ICCAT, 2012a). However, details of the implementation arrangements for these requirements are yet to be developed.

Elasmobranch species reported to interact with pelagic longline fisheries were matched with their classifications according to the 2012 IUCN Red List (ICCAT 2007a, Lack and Meere 2009, IUCN 2013a). This list of species is believed to be highly conservative since, for example, the likelihood of whale sharks and some of the rays interacting with longline gear is considered to be very low. The results indicate that six critically endangered or endangered species are believed to interact with longline gear at some level, as do another 23 species classified as vulnerable (Table 4). The remaining 60% of the species known to interact with longline gear are classified as not threatened with extinction (i.e. near threatened [21], least concern [14] or data deficient [15]).

TABLE 4
Species believed to interact with pelagic longline fisheries and their IUCN Red List classifications

Threatened categories		Non-threatened categories		Data-deficient category	
Angelshark (*Squatina squatina*)	CR	Blacktip reef shark (*Carcharhinus melanopterus*)	NT	Bignose shark (*Carcharhinus altimus*)	DD
Sawback angelshark (*Squatina aculeata*)	CR	Blacktip shark (*Carcharhinus limbatus*)	NT	Broadsnouted sevengill shark (*Notorynchus cepedianus*)	DD
Smoothback angelshark (*Squatina oculata*)	CR	Blue shark (*Prionace glauca*)	NT	Bull ray (*Pteromylaeus bovinus*)	DD
Devil ray (*Mobula mobular*)	EN	Bluntnose sixgill shark (*Hexanchus griseus*)	NT	Common eagle ray (*Myliobatis aquila*)	DD
Great hammerhead (*Sphyrna mokarran*)	EN	Bull shark (*Carcharhinus leucas*)	NT	Cuban dogfish (*Squalus cubensis*)	DD
Scalloped hammerhead (*Sphyrna lewini*)	EN	Caribbean reef shark (*Carcharhinus perezi*)	NT	Lesser devil ray (*Mobula hypostoma*)	DD
Basking shark (*Cetorhinus maximus*)	VU	Copper shark (*Carcharhinus brachyurus*)	NT	Longnose spurdog (*Squalus blainville*)	DD
Bigeye thresher (*Alopias superciliosus*)	VU	Crocodile shark (*Pseudocarcharias kamoharai*)	NT	Megamouth shark (*Megachasma pelagios*)	DD
Common thresher (*Alopias vulpinus*)	VU	Dusky smooth-hound (*Mustelus canis*)	NT	Narrowfin smooth-hound (*Mustelus norrisi*)	DD
Dusky shark (*Carcharhinus obscurus*)	VU	Galapagos shark (*Carcharhinus galapagensis*)	NT	Smalltail shark (*Carcharhinus porosus*)	DD
Great White shark (*Carcharodon carcharias*)	VU	Greenland shark (*Somniosus microcephalus*)	NT	Smalltooth sand tiger (*Odontaspis noronhai*)	DD
Greenback stingaree (*Urolophus viridis*)	VU	Grey reef shark (*Carcharhinus amblyrhynchos*)	NT	Spotted skate (*Raja straeleni*)	DD
Gulper shark (*Centrophorus granulosus*)	VU	Lemon shark (*Negaprion brevirostris*)	NT	Torpedo ray (*Torpedo nobiliana*)	DD
Longfin mako (*Isurus paucus*)	VU	Plunkets shark (*Proscymnodon plunketi*)	NT	Velvet dogfish (*Zameus squamulosus*)	DD
Manta ray (*Manta birostris*)	VU	Shagreen ray (*Leucoraja fullonica*)	NT	Whitenose shark (*Nasolamia velox*)	DD
Night shark (*Carcharhinus signatus*)	VU	Sharpnose sevengill shark (*Heptranchias perlo*)	NT		
Oceanic whitetip shark (*Carcharhinus longimanus*)	VU	Silky shark (*Carcharhinus falciformis*)	NT		
Pelagic thresher (*Alopias pelagicus*)	VU	Silvertip shark (*Carcharhinus albimarginatus*)	NT		
Porbeagle (*Lamna nasus*)	VU	Spinner shark (*Carcharhinus brevipinna*)	NT		
Sand tiger shark (*Carcharias taurus*)	VU	Whip stingray (*Dasyatis akajei*)	NT		
Sandbar shark (*Carcharhinus plumbeus*)	VU	Whitetip reef shark (*Triaenodon obesus*)	NT		
Shortfin mako (*Isurus oxyrinchus*)	VU	Atlantic sharpnose shark (*Rhizoprionodon terraenovae*)	LC		
Smalltooth sand shark (*Odontaspis ferox*)	VU	Australian blacktip (*Carcharhinus tilstoni*)	LC		

Threatened categories		Non-threatened categories		Data-deficient category
Smooth hammerhead (*Sphyrna zygaena*)	VU	Birdbeak dogfish (*Deania calcea*)	LC	
Smooth-hound (*Mustelus mustelus*)	VU	Caribbean sharpnose shark (*Rhizoprionodon porosus*)	LC	
Spurdog fish (*Squalus acanthias*)	VU	Carpet shark (*Cephaloscyllium isabellum*)	LC	
Tope shark (*Galeorhinus galeus*)	VU	Cookie cutter shark (*Isistius brasiliensis*)	LC	
Whale shark (*Rhincodon typus*)	VU	Longnose velvet dogfish (*Centroselachus crepidater*)	LC	
Zebra shark (*Stegostoma fasciatum*)	VU	Pelagic stingray (*Pteroplatytrygon violacea*)	LC	
		Roughtail stingray (*Dasyatis centroura*)	LC	
		Salmon shark (*Lamna ditropis*)	LC	
		Spined pygmy shark (*Squaliolus laticaudus*)	LC	
		Starry smooth-hound (*Mustelus asterias*)	LC	
		Velvet belly (*Etmopterus spinax*)	LC	
		Bullhead sharks (*Heterodontus* spp.)	LC/ DD	

Note: The IUCN Red List classifies the risk of extinction in six categories: data deficient (DD), least concern (LC), near threatened (NR), vulnerable (VU), endangered (EN) and critically endangered (CR).
Sources: ICCAT (2007a); Lack and Meere (2009); IUCN (2013a).

Ecological risk assessments of various types have been conducted for some of the major tuna fisheries and have provided combined productivity–susceptibility rankings for shark species. One study conducted for pelagic sharks in the Atlantic concluded that silky (*Carcharhinus falciformis*), shortfin mako (*Isurus oxyrinchus*), bigeye thresher (*Alopias superciliosus*), oceanic whitetip and longfin mako (*Isurus paucus*) were the five most vulnerable species (Cortés *et al*., 2010). Another study for a master list of all species known to interact with ICCAT longline fisheries (i.e. coastal and pelagic sharks as well as other fishes and turtles) ranked two pelagic species (shortfin mako and crocodile [*Pseudocarcharias kamoharai*] sharks), and seven coastal species (sharpnose [*Rhizoprionodon terraenovae*], sharpnose sevengill [*Heptranchias perlo*], bignose [*Carcharhinus altimus*], night [*C.* signatus], sandbar [*C.* plumbeus], spinner [*C.* brevipinna] and dusky [*C.* obscurus] sharks), as most at risk (Arrizabalaga *et al*., 2011). The most recent Atlantic ecological risk assessments for elasmobranch species were released by ICCAT in 2012 and concluded that bigeye thresher, longfin and shortfin mako, and night sharks were the most vulnerable, and that scalloped and smooth hammerheads and pelagic stingray were the least vulnerable species (ICCAT, 2012a). The IOTC has identified the shortfin mako, bigeye and pelagic (*A. pelagicus*) threshers, silky, oceanic whitetip, smooth hammerhead, porbeagle, longfin mako, great hammerhead and blue shark as the ten most vulnerable species to longline gear (IOTC 2012a). In the Western and Central Pacific, based on ecological risk assessment findings and other considerations, the WCPFC had, as of 2011, designated 13 shark species as key species including: blue; oceanic whitetip; silky; shortfin and longfin mako; common (*Alopias vulpinus*), pelagic and bigeye threshers; porbeagle; and winghead (*Eusphyra blochii*), scalloped, great and smooth hammerhead sharks (Kirby and Hobday, 2007;

Clarke and Harley, 2010; Rice and Harley 2012a). In 2012, the WCPFC also designated the whale shark as a key shark species but it is not known to interact with longline fisheries (WCPFC, 2013a; Harley, Rice and Williams, 2013).

Although the shark species identified as most vulnerable by the different t-RFMOs vary, particularly in terms of whether coastal as well pelagic species were assessed and ranked, there are a number of at-risk species common to all oceans. These include the makos and porbeagle, the threshers, the hammerheads and some pelagic requiem sharks such as the silky and oceanic whitetip sharks. Other species that have not consistently been included in these ecological risk assessments appear to deserve further consideration. The crocodile shark was not evaluated by Cortés *et al*. (2010) owing to a lack of biological information but was found to have a considerably higher mortality rate at haulback (66 percent) compared with other pelagic shark species (average of 14 percent; Musyl *et al*., 2011) and shows a large degree of sexual segregation and declining catch rates (Walsh, Bigelow and Sender, 2009). In addition, few skates or rays have been identified as being of concern perhaps due relatively high productivity and relatively low susceptibility (Cortés *et al*., 2010). Nevertheless, the recent decision by CITES to list manta rays, coupled with conflicting information on whether they form a non-negligible component of the longline catch (Lack and Meere, 2009; SPC-OFP, 2010) suggests that this species may warrant greater attention. The pelagic stingray (*Pteroplatytrygon violacea*), a wide ranging and common if not numerically dominant component of pelagic longline bycatch worldwide (Mollett, 2002; Lack and Meere, 2009), was assessed as being of low risk by Cortés *et al*. (2010) owing to low selectivity and post-capture mortality scores.

2.2 ELASMOBRANCH INTERACTIONS, MORTALITY RATES AND STOCK STATUS BY AREA

2.2.1 Atlantic Ocean

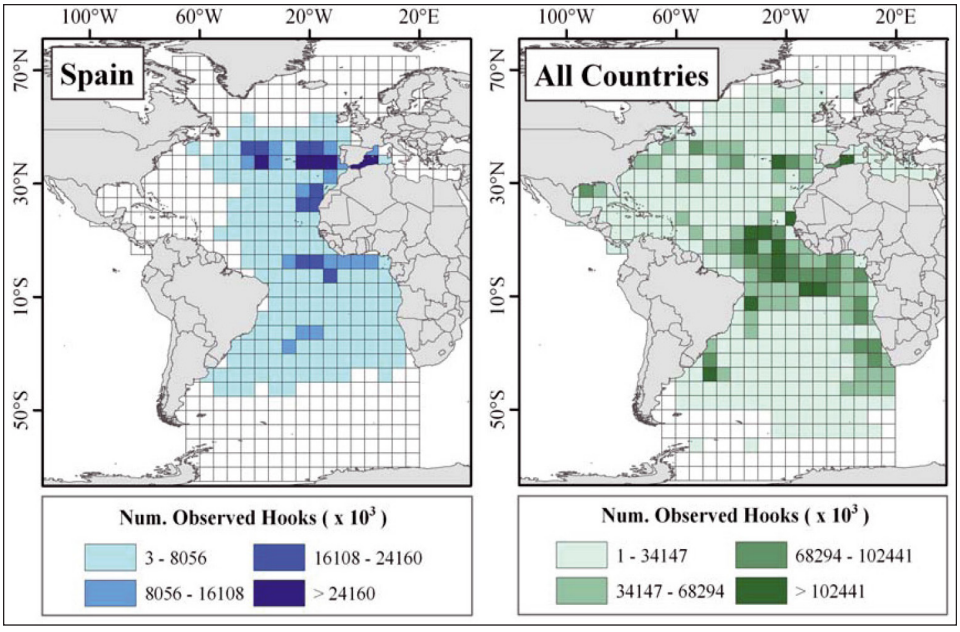
The first t-RFMO to attempt stock assessments for sharks was ICCAT. As such, it was the first t-RFMOs to formally catalogue, evaluate and attempt to improve its elasmobranch data holdings. The initial shark stock assessment exercise was limited to blue and shortfin mako sharks and faced considerable uncertainty associated with under-reporting of total shark removals and a lack of size, age and sex composition data. This situation resulted in an inconclusive stock status outcome, and prompted calls for better catch reporting by members, data mining to estimate missing historical data, and broader use of trade statistics (ICCAT, 2005). For example, catch estimates prepared by ICCAT that accounted for non-reporting but not under-reporting were about double, and catch estimates based on shark fin trade data were at least triple, the reported catches in the ICCAT database (Clarke, 2008).

The elasmobranch data holdings of ICCAT have improved considerably in the past decade such that catch data (i.e. Task I) for all but three of the species assessed in the 2012 ecological risk assessment (i.e. pelagic stingray, crocodile shark and giant manta) are now available for most major fleets for at least the last ten years. However, it is not clear to what extent these catch data include dead discards and/or live releases. Moreover, the availability and quality of catch and effort, and size, data (Task II) for most elasmobranch species remain poor in all but the most recent years (ICCAT, 2012b). Of the elasmobranch catches (in tonnes) reported to ICCAT, almost all consist of blue, shortfin mako and porbeagle sharks (> 99 percent in 2010) and blue shark alone constitute the vast majority (> 90 percent in 2010). Using shortfin mako catches to illustrate the most recent data submissions by fleet, only the EU-Spain (3 284 tonnes of shortfin mako) and EU-Portugal (1 315 tonnes of shortfin mako) fleets reported shortfin mako catches of more than 310 tonnes in 2010 (ICCAT 2012b). The total longline catch of shortfin mako reported to ICCAT for all fleets in 2010 was

6 029 tonnes (ICCAT 2012b) but this is lower than the shortfin mako catches for the Atlantic reported to the FAO (6 440 tonnes).[7] Questions arising from discrepancies between catch data reported to ICCAT and data compiled in EUROSTAT and FAO catch databases are the subject of ongoing discussions between these organizations (ICCAT, 2012a).

It might reasonably be hypothesized based on the locations of known markets for shark meat and fleet sizes, that the Atlantic's areas of highest longline fishery interactions with sharks would lie in the areas fished by the Spanish and Portuguese longline fleets. According to Cortés *et al*. (2010), information on the effort distribution for the Portuguese fleet is unavailable but can be approximated as being the same as the effort distribution for the Spanish fleet but at one-fifth the level. Assuming observed longline effort is representative, the Spanish fleet fishes intensively in fishing grounds around Spain and southward to Morocco, but also in the area of heaviest longline effort for all fleets combined, which runs westward toward the central Atlantic from the Gulf of Guinea (Figure 14). These primary fishing grounds are located within the distributional range of many of the most common pelagic shark species, but abundance of these species in these fishing grounds compared with other areas of lower fishing pressure is unknown. An ecological risk assessment based on these data assumed that the encounterability was 100 percent because there was always some degree of overlap between the depth distributions of each species and the longline gear (Cortés *et al*., 2012), regardless of whether the gear was fished shallow for swordfish or deeper for tuna.

FIGURE 14
Longline observer effort distribution for the Spanish longline fleet and all fleets reporting to ICCAT, 1950–2005



Source: Cortés *et al. (2010).*

As discussed in Section 1.2.1, there is as yet no ICCAT scientific observer programme and only one ICCAT contracting party reports data on bycatch interactions to the secretariat (Cotter, 2010). There is thus no comprehensive data source for determining what proportions of elasmobranch catches in the Atlantic are discarded (dead or alive)

[7] FAO capture production statistics by area and gear type are not available, but it is unlikely that substantial numbers of mako sharks are caught in purse seine fisheries.

and retained. However, with two of the world's largest known markets for shark meat in Spain and Brazil (Clarke, Francis and Griggs, 2013) supplied by domestic, and potentially by foreign, longline fleets working in nearby fishing grounds, it can be assumed that retention rates are high compared with the average for pelagic longline fleets worldwide.

As described above, ICCAT has over time improved its shark data holdings and used them to conduct stock assessments for three pelagic shark species that interact with pelagic longline fisheries in the Atlantic. Blue sharks were first assessed in 2004 and subsequently in 2008 for the North and South Atlantic, and it was concluded that neither biomass nor fishing mortality maximum sustainable yield reference points had been breached (ICCAT, 2005, 2008a). Shortfin mako assessments conducted by ICCAT at the same workshops failed to draw any conclusion about the North and South Atlantic stocks but considered that biomass and fishing mortality reference points may have been breached (ICCAT, 2005, 2008a). The shortfin mako stock assessment was re-visited in 2012 using improved data sets and it was concluded that current levels of catches may be considered sustainable. However, caution was expressed concerning inconsistencies between flat or increasing trends in catch rates in both the North and South Atlantic and estimated biomass trajectories (ICCAT, 2012b). Porbeagle sharks were assessed in 2009 but data for the South Atlantic were too limited to provide a robust indication of stock status. For the North Atlantic, biomass was estimated to be below maximum sustainable yield (MSY) levels, and fishing mortality was estimated to be at or above MSY levels in the northeast Atlantic and below MSY levels in the northwest Atlantic (ICCAT, 2012a).

Several other studies have examined catch rates for elasmobranchs interacting with longline fisheries in the Atlantic. Using United States longline fishery observer data and revisiting a previous analysis (Baum *et al*., 2003), abundance indices showed: major (76 percent) declines for hammerheads and large coastal sharks; moderate declines (53 percent) for blue and oceanic whitetip sharks; and no or increasing trends for mako, thresher and tiger sharks (Baum and Blanchard, 2010). A comparison of exploratory research survey data from the 1950s and 1990s in the Gulf of Mexico found that oceanic whitetip and silky sharks declined by 99 percent and 90 percent, respectively (Baum and Myers, 2004). Linking recent longline observer records with historical and recent longline survey data resulted in an estimated 30 percent decline in catch rates for blue shark for the western North Atlantic (Aires-da-Silva, Hoey and Gallucci, 2008).

2.2.2 Eastern Pacific Ocean

Owing to the focus of the fishery in the EPO on purse seine operations, there are relatively few data available on shark interactions with the longline fishery in this area (IATTC, 2012a). However, historical data from purse seine fleets, and from longline fleets based on the central Pacific, have raised concerns about the status of some shark species. For example, standardized catch rate analyses based on Eastern Pacific purse seine floating object set data for 1994–2004 demonstrated that catch rates of silky sharks had declined by 60–82 percent over the time series (Minami *et al*., 2007). For longline fisheries, standardized catch rate analyses for Japanese research and training vessels from the central North Pacific also showed a declining trend for this species since 1992 (Shono, 2008). A comparison of Japanese longline research and training vessel data in the central Pacific between the 1960s and the 1990s found no clear differences in abundance for blue, oceanic whitetip, silky and thresher sharks (Matsunaga and Nakano, 1999). In contrast, declining trends in biomass/abundance were found for all of these species in a comparison between historical scientific survey data and recent longline observer data for the eastern and central Pacific (Ward and Myers, 2005b).

The only longline fleet that reports its total catches of sharks to the IATTC is that of the United States of America. However, as part of an ongoing effort to assess the

status of silky shark stocks, estimates of catches have been constructed from catch rate and effort data from China, Japan, the Republic of Korea, Mexico and Taiwan Province of China. For many of the other coastal longline fleets catching sharks, species-specific records are available only for very recent years, if at all (IATTC, 2011b). The silky shark stock assessment exercise has also resulted in the compilation of size data for this species from a number of longline fleets including those from Japan, Mexico and Ecuador, but the temporal span of these databases is generally limited to six years or less (Aires-da-Silva, Lennert-Cody and Maunder, 2013). Size data holdings for other species from longline fisheries are likely to be even further constrained.

As a first step toward assessing stock status, a productivity–susceptibility analysis was conducted for the EPO purse seine fishery. Of the 26 species considered (i.e. including fishes, turtles and mammals), those with the greatest overall vulnerability to overfishing were shortfin mako, great hammerhead, bigeye thresher, and the giant manta ray (IATTC 2012b). Because prior analysis of purse seine data had identified silky and oceanic whitetip sharks as species of concern (Minami *et al*., 2007; IATTC, 2012a), the IATTC began working towards a stock assessment for silky sharks in 2009. Although this assessment is still in progress, preliminary results presented in May 2013 suggest that the stock declined from 1993 to 2005 but was rebuilding up to 2010. It was noted that this rebuilding trend coincides with recent fishery closures associated with IATTC tuna conservation measures, finning restrictions in national waters, and a decline in pelagic longline fishing effort related to increased fuel prices. An additional finding from the preliminary assessment is that high seas purse seine and longline fisheries targeting tunas take a minor component of the total silky shark catch compared with other coastal and non-coastal longline fisheries targeting a mix of sharks, billfishes and tunas (Aires-da-Silva, Lennert-Cody and Maunder, 2013). As a caveat to the indications of rebuilding, standardized purse seine "catch per unit of effort" estimates for 2011–12 suggest that abundance is declining (IATTC, 2013a).

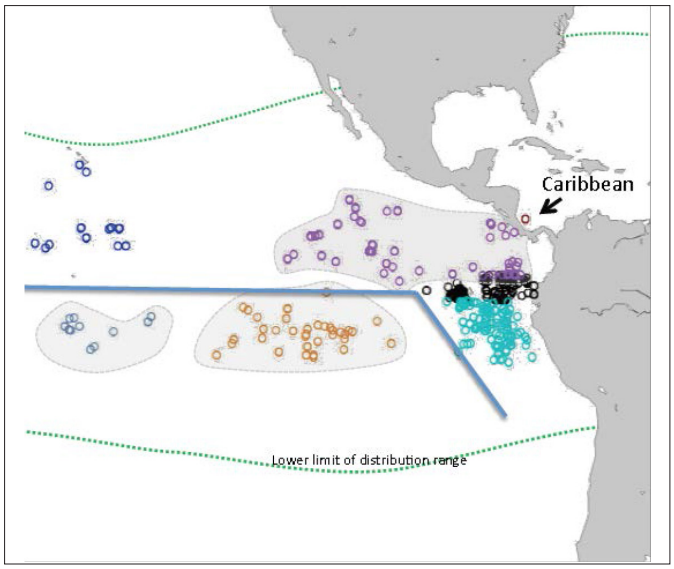
The preliminary IATTC assessment was undertaken mainly on the basis of what is believed to be a northern EPO silky shark stock that is primarily distributed north of the equator but extends southwards along the coast of South America (IATTC, 2013a; Figure 15). A study of spatial distribution and catch rates of silky shark by the purse seine fishery in the EPO indicated considerable spatial aggregation of small-sized silky sharks such that only 6 percent were found south of the equator, 33 percent between the equator and 5°N and 57 percent north of 5°N (Watson *et al*., 2009). This kind of spatial aggregation pattern was also found in the southeast Pacific in a study of shortfin mako sharks caught by Spanish longliners. It showed a distinct division between aggregations of mainly adult males in the west and mainly juvenile females in the east (Mucientes *et al*., 2009).

2.2.3 Western and Central Pacific Ocean

The WCPFC was the first t-RFMO to establish a formal shark research plan covering stock assessment, research coordination and fishery statistics improvement (Clarke and Harley, 2010). With the implementation of this plan in December 2010, the WCPFC also agreed that annual catch estimates and operational-level catch and effort data for designated key shark species would become part of the scientific data to be provided by members each year to the commission (WCPFC, 2013b).[8] Data holdings as of April 2013 (WCPFC, 2013c) show that of the 33 longline fleets that may interact with the key shark species, the majority are providing some catch and catch/effort data on blue and mako sharks, but fewer are providing data on silky, oceanic whitetip and thresher

[8] The whale shark was added to the list of key shark species in December 2012, and so data reporting requirements for this species did not begin in 2011 as for the other 13 species. Members of the WCPFC with domestic legal constraints are allowed to provide aggregated (rather than operational) catch and effort data.

FIGURE 15
Location of capture of silky shark genetic samples in the Eastern Pacific Ocean and a hypothesized boundary between northern and southern stocks



Source: Aires-da-Silva, Lennert-Cody and Maunder (2013).

sharks (Table 5). Data submission for porbeagle sharks is only required for catches south of 20°S, and this, as expected, leads to fewer records. It is not possible to assess from the data holdings catalogue the degree to which the data provided are complete (e.g. whether each fleet is actually recording all catches of each species). In addition, the values in Table 5 represent any reporting by a fleet for a key shark species for any period; therefore, some of the fleets shown in the table have reported consistently for many years whereas others have not.

TABLE 5
Number of longline fleets (of a total of 33) reporting each type of data to the WCPFC for the key shark species shown

	Blue	Mako	Silky	Oceanic whitetip	Thresher	Porbeagle
Catch estimates	22	20	14	15	13	5
Aggregate catch and effort data	20	22	11	14	11	0
Operational catch and effort data	14	16	9	12	10	4
Length data	23	22	19	20	17	3

Notes: "mako" = *Isurus* spp.; "thresher" = *Alopias* spp.). Data holdings for hammerhead and whale sharks have not yet been catalogued.
Source: WCPFC (2013c).

Although catch, effort and size reporting for sharks is improving, the best method for estimating total catches of shark species is still to use the WCPFC's Regional Observer Programme data to statistically extrapolate from observed longline sets to the Western and Central Pacific Ocean (WCPO) longline fishery as a whole. Using this method, catch estimates for the five original key shark species from 1992–2009 indicated that blue sharks made up more than 80 percent of the longline catch, with silky and oceanic whitetip sharks representing 6 percent each and mako and thresher sharks 3 percent each (Table 6). Earlier alternative catch estimates based on shark fin trade data suggested that, rather than catches declining since 1998 as shown in the observer data extrapolations, shark catches may have continued to increase for several more years to a level two to three times higher than the highest estimates based on

observer data (Clarke, 2009). As both observer and shark fin trade estimates may be biased both by the underlying data and by the methodologies applied in the estimates, considerable uncertainty about the true values of WCPO shark catches remains.

TABLE 6
Estimates of longline shark catches in the WCPFC Statistical Area east of 130°E

	Blue	Mako	Oceanic whitetip	Silky	Thresher	Total
			(thousands of sharks)			
1992	1 351	86	39	0	58	1 534
1993	1 333	71	85	0	64	1 553
1994	1 662	75	184	16	70	2 007
1995	2 350	73	236	161	75	2 895
1996	3 050	72	196	140	68	3 526
1997	3 587	76	186	135	57	4 041
1998	4 049	90	249	165	62	4 615
1999	3 683	100	223	167	74	4 247
2000	2 124	91	186	163	70	2 634
2001	1 033	84	122	149	71	1 459
2002	627	79	110	142	80	1 038
2003	574	74	88	97	76	909
2004	639	65	100	103	75	982
2005	671	55	74	114	71	985
2006	642	47	46	133	64	932
2007	672	44	51	167	72	1 006
2008	588	47	55	185	71	946
2009	358	53	53	189	61	714

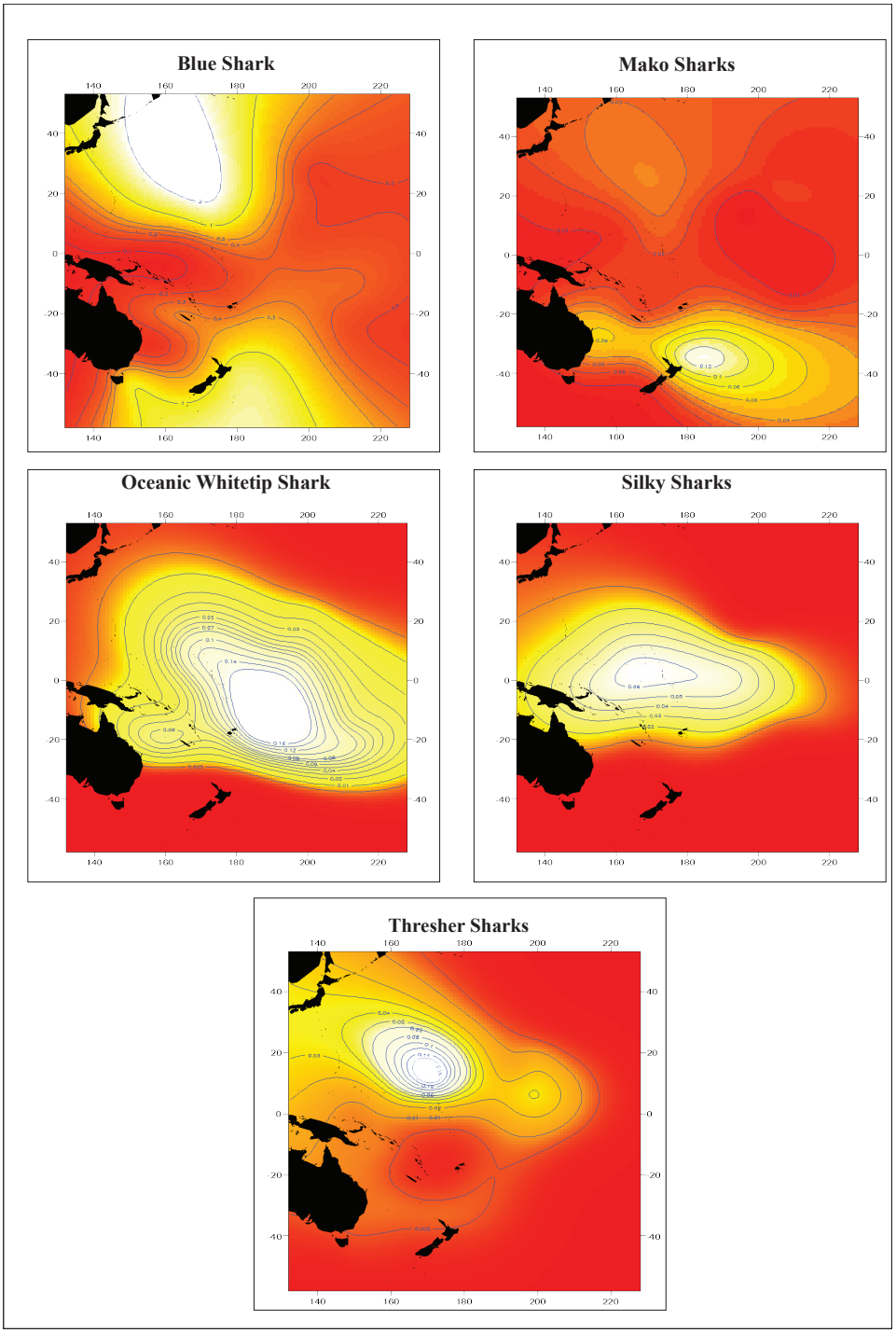
Source: Lawson (2011).

In addition to providing quantitative estimates of species-specific catches, Lawson (2011) also used the observer data set to model and predict catch rates for each species across the WCPO (Figure 16). For the tropical sharks, model-predicted catch rates were highest in the central Pacific near the equator, with the centre of the silky shark distribution slightly to the west of the centre of the oceanic whitetip distribution. In subtropical and temperate waters, catch rates for blue sharks appear to be high in both northern and southern hemispheres, but it should be noted that this result derives from extrapolation of large observed catches during a small number of trips by Hawaii longliners into the otherwise data-poor areas of the northwest Pacific. Catch rates for mako sharks were found to be higher in the southern hemisphere, while catch rates for threshers were higher in the northern hemisphere around 15°N (Lawson, 2011).

Using the same WCPO observer data set, an indicator of population status in the form of catch rate, and an indicator of biological response to fishing pressure in the form of median size, were estimated for blue, mako, oceanic whitetip and silky sharks (Clarke *et al*., 2013). Standardized longline catch rates were found to have declined significantly for blue sharks in the North Pacific (5 percent per year; confidence interval 2–8 percent), mako sharks in the North Pacific (7 percent per year; confidence interval 3–11 percent), and oceanic whitetip sharks in tropical waters (17 percent per year, confidence interval 14–20 percent). Within their core habitat in tropical waters, both silky and oceanic whitetip sharks' median lengths declined significantly, suggesting that these populations are subject to considerable fishing pressure. Confirmatory evidence for heavy exploitation of North Pacific blue and mako sharks can be found

FIGURE 16

Model-predicted catch rates for key shark species (sharks per hundred hooks) showing the effects of latitude and longitude



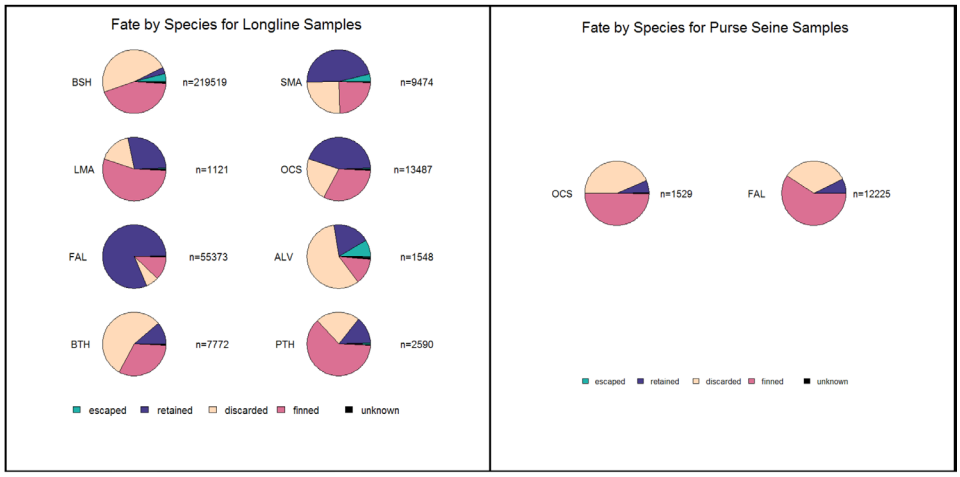
Note: Red, yellow and white indicate low, intermediate and high catch rates, respectively.
Source: Lawson (2011).

in an analysis of Japanese research and training vessel and commercial longline data for 1992–2008 (Clarke *et al*., 2011) and, for blue shark only, by an analysis of catch rates from deep-set Hawaii longline fishing for 1996–2006 (Polovina *et al*., 2009). Similar trends in declining catch rates for oceanic whitetip sharks were found since 1995 in an independent analysis of the Hawaii longline fishery (Walsh and Clarke, 2011).

Insights into shark utilization in the WCPO are the best informed of any of the t-RFMOs because the WCPFC has the longest-running and most comprehensive longline fishery observer programme. An analysis of these data indicated that, despite the adoption of finning regulations by the WCPFC in 2007, there is little evidence of a reduction of finning rate (Figure 17). Moreover, species-specific patterns in utilization showed that silky and oceanic whitetip sharks are more likely to be retained than finned and, thus, that mortality to these species is not reduced by the presence of a finning ban (Clarke *et al.*, 2013).

Observer data from the WCPFC also allow for an analysis of the proportion of

FIGURE 17
Fate of sharks by species as recorded by observers on longline vessels reporting to the WCPFC, 1995–2009



Note: BSH = blue shark; SMA = shortfin mako shark; LMA = longfin mako shark; OCS = oceanic whitetip shark; FAL = silky shark; ALV = common thresher shark; BTH = bigeye thresher shark; PTH = pelagic thresher shark.

Source: Clarke (2011).

sharks suffering mortality owing to either haulback or handling (i.e. either rough handling before discarding, or being finned or retained). Calculation of the proportion of sharks of each species that were: (i) initially recorded by observers as alive or alive but injured, (ii) cut free, escaped or discarded, and (iii) not recorded at the last sighting as dead or dying, showed that survival ranged as high as 36–56 percent for common thresher, blue shark and bigeye thresher, but as low as 4–10 percent for silky, pelagic thresher and longfin mako sharks (Clarke, 2011). This analysis was based on the entire time frame of the data set and thus was not designed to reflect changes in national regulations regarding finning or retention of particular species.

Several stock assessments have been conducted for pelagic shark species in the WCPO that interact with pelagic longline fisheries. The first of these focused on blue sharks and used commercial logbook data from Japan and Hawaii up to 2002 (Kleiber *et al.*, 2009). It concluded that the population biomass appeared close to its MSY reference point but that fishing mortality may be approaching the MSY reference point. Using the biomass MSY reference point from a previous version of this stock assessment and an estimate of the North Pacific catch of blue shark based on shark fin trade data, it was estimated that harvest levels were close to or possibly exceeding MSY levels (Clarke *et al.*, 2006). Recent North Pacific blue shark stock assessments have shown mixed results based on which stock abundance index was applied. Using an index based on longline fishing in the western North Pacific by a fleet based in

northeastern Japan, catch rate trends increase in recent years and suggest that stock size is increasing (ISC, 2013; Rice *et al*., 2013). It should be noted, however, that this fleet is known to be targeting blue sharks (Clarke *et al*., 2011) and proper account must be taken of this factor when standardizing catch rates. In contrast, when an abundance index based on the Hawaii-based longline fishery was applied, the stock was found to be overfished and overfishing was occurring (ISC, 2013; Rice *et al*., 2013). As a result of this situation, the WCPFC Scientific Committee has called for both North Pacific blue shark assessments to be revisited in 2014.

While the majority of stock assessment work in the North Pacific has focused on the blue shark, three population studies of shortfin mako and thresher sharks have been completed using data from longline fisheries of Taiwan Province of China. For the shortfin mako, a Northwest Pacific stock assessment based on virtual population analysis found a downward trend in spawning potential ratio and concluded that the stock "might have been overexploited", recommending a reduction in current fishing effort of 32 percent (Chang and Liu, 2009). Using a spawning per recruit analysis, pelagic thresher sharks were found to be slightly overexploited and a reduction in fishing effort was recommended (Liu *et al*., 2006). An update to this study concluded that the stock is overexploited and recommended nursery closures and/or size limit management (Tsai, Liu and Joung, 2010).

Most recently, shark stock assessments in the tropical WCPO conducted under the WCPFC's Shark Research Plan have focused on oceanic whitetip and silky sharks. Both species have been assessed primarily on the basis of WCPFC regional observer programme data. For oceanic whitetip sharks, the assessment noted that the species has very low fecundity and that estimated spawning biomass, total biomass and recruitment all decline consistently throughout the period of the model. All plausible model runs indicated that overfishing is occurring: estimated current fishing mortality was more than six times the MSY fishing mortality. The stock was also found to be in an overfished state relative to MSY-based (currently only 15 percent of MSY spawning stock biomass) and depletion-based reference points (currently only 6 percent of pre-fishing spawning stock biomass) (Rice and Harley, 2012b; WCPFC, 2012d). The initial assessment of silky sharks revealed conflicting catch rate trends for longline fleets, which indicated overfishing is occurring, and for purse seine fleets, which indicated no sustainability concerns (WCPFC, 2012d). Revisiting this assessment in 2013, it was concluded that the current rate of fishing mortality is 4.5 times higher than the MSY fishing mortality, i.e. overfishing is occurring, and that spawning biomass has declined to 70 percent of MSY spawning biomass, i.e. the stock is overfished (Rice and Harley, 2013).

2.2.4 Indian Ocean

Data submission requirements for parties to the IOTC have included catch reporting for blue, shortfin mako, porbeagle and "other" sharks (i.e. not identified to species) since 2008. In addition, since 2013, catch reporting has been required for hammerhead sharks, and encouraged for thresher, crocodile, oceanic whitetip, tiger, great white, pelagic stingray, mantas and other rays (IOTC, 2012a). Under Resolution 10/02, IOTC members are also required to report size data for "at least one fish per ton caught by species and fishery," or for longline fleets to provide size data through an observer programme that has at least 5 percent coverage of all fishing operations.

Current data holdings for sharks have been evaluated in terms of what percentage of the fleets that now report nominal catches of IOTC species also report the required shark data (i.e. percentages are not based on all fleets that should report, only on those that do report their catches of other species). More than 75 percent of the fleets report blue, mako and "other" shark catches; 30–75 percent of the fleets report porbeagle, hammerhead, thresher, oceanic whitetip and silky shark catches; and less

than 30 percent of the fleets report any catches of the other species. In most cases, these reporting rates have only been achieved from the early 2000s onward (IOTC, 2012b).

As the shark data reporting requirements have been in place only since 2008, the IOTC has not yet undertaken extensive evaluation of the extent to which submitted data is likely to under-represent actual catches. However, for most species, the IOTC has noted that it appears that substantial catch quantities have gone unrecorded in some countries, and that available data often do not account for discards, and reflect dressed weights rather than live weights. Similarly, for most species, finning is believed to be regularly occurring. Post-release mortality rates for shark discarded without being finned are unknown but probably high (IOTC, 2012b). Despite these uncertainties, catch estimates for the most recent years for seven species of sharks, as well as an aggregated catch estimate for all sharks not reported to species, were produced (Table 7). These estimates indicate that 74 percent of the longline shark catch consists of blue shark.

TABLE 7
Longline catch estimates for sharks reported to the IOTC, 2009–2011

	2009	2010	2011	Number of countries reporting catches in 2011	Percentage of reported longline shark catch	Standardized catch rate trend available?
		(tonnes)			*(%)*	
Blue shark	9 687	9 829	9 540	12	74	Yes (n=2)
Oceanic whitetip	245	761	388	4	0.6	Yes (n=2)
Scalloped hammerhead	21	15	120	1	–	No
Shortfin mako	896	1 246	1 361	9	12	Yes (n=1)
Silky	655	1 836	3 353	5	1.5	No
Bigeye thresher	5	2	330	2	–	No
Pelagic thresher	1	1	10	2	–	No
Sharks not reported to species	65 380	64 387	55 135	–	–	No
Total	76 890	78 077	70 237			

Source: IOTC (2012b).

No catch rate data are held by the secretariat; therefore, to date, the only available abundance indices have been produced by national scientists from Japan, EU-Portugal and EU-Spain. In general, these indices are associated with considerable uncertainty and/or show variable or no trends (IOTC, 2012b). Owing to the lack of data, broadscale spatial or temporal patterns of interactions between shark species and the longline fishery in the Indian Ocean are unknown.

The IOTC's Working Party on Ecosystems and Bycatch has noted that as data accumulates over time the potential for conducting simple stock assessments will increase. For the time being, however, assessments have been limited to productivity–susceptibility analyses (PSAs). The 2012 PSA identified the ten most vulnerable shark species to longline gear as shortfin mako, bigeye thresher, pelagic thresher, silky shark, oceanic whitetip, smooth hammerhead, porbeagle, longfin mako, great hammerhead, and blue shark (IOTC, 2012b).

2.2.5 Summary of shark stock status

Table 8 provides a summary of the latest stock status of shark species as assessed by the t-RFMOs.

TABLE 8
Shark stock status based on assessments conducted by t-RFMOs

t-RFMO	Species	Assessment released	Overfished?	Overfishing?	Reference
ICCAT	Blue (North Atlantic)	2008	No	No	ICCAT (2008a)
ICCAT	Blue (South Atlantic)	2008	No	No	ICCAT (2008a)
ICCAT	Shortfin Mako (North Atlantic)	2012	No, but uncertainty noted	No, but uncertainty noted	ICCAT (2012b)
ICCAT	Shortfin Mako (South Atlantic)	2012	No, but uncertainty noted	No, but uncertainty noted	ICCAT (2012b)
ICCAT	Porbeagle (North Atlantic)	2009	Yes	Yes (Northeast Atlantic); No (Northwest Atlantic)	ICCAT (2012a)
ICCAT	Porbeagle (South Atlantic)	2009	Inconclusive	Inconclusive	ICCAT (2012a)
IATTC	Silky	2013	Yes ("rebuilding")	No ("rebuilding")	Aires-da-Silva, Lennert-Cody and Maunder (2013)
WCPFC	Oceanic whitetip	2012	Yes	Yes	Rice and Harley (2012b); WCPFC (2012d)
WCPFC	Silky	2013	Yes	Yes	Rice and Harley 2013
WCPFC	Blue (North Pacific)	2013	Inconclusive	Inconclusive	Rice *et al.* (2013); WCPFC (2013d)
ISC	Blue (North Pacific)	2013	Inconclusive	Inconclusive	ISC (2013); WCPFC (2013d)

2.3 MANAGEMENT MEASURES AND THEIR EFFECTIVENESS

Management measures for sharks in pelagic longline fisheries are either national regulations applied to certain fleets or certain areas (e.g. national waters), or are international measures adopted by the t-RFMOs and implemented by their members for vessels they flag. This section describes existing shark management measures in the form of finning controls, catch controls, operational and gear controls, closures, trade controls and shark-specific management plans.

2.3.1 Finning controls

Starting with ICCAT in 2004, and followed by the IATTC and the IOTC in 2005, the WCPFC in 2006 and the CCSBT in 2008,[9] all of the t-RFMOs have adopted controls on shark finning (Clarke, 2013). Most of these measures have similar provisions relating to the mitigation of fishing impacts on sharks including:

- an intent to minimize waste and discards through full utilization;
- encouragement of live release in fisheries that are not directed at sharks and in which sharks are not used for food or subsistence;
- prohibition of retention on board, transshipment, landing or trading shark fins that total more than 5 percent of retained shark carcasses.

[9] The CCSBT measure is a non-binding call for members to comply with all current binding and recommendatory measures aimed at the protection of ecologically related species, including seabirds, sea turtles and sharks when fishing in the ICCAT, IOTC and WCPFC convention areas (CCSBT, 2008).

Several problems have arisen with regard to interpretation of the 5 percent fin-to-carcass ratio (Fowler and Séret, 2010; Biery and Pauly, 2012; Santana-Garcon, Fordham and Fowler, 2012). First, while provision is made in the measures for the ratio to be reviewed and modified, it is now well understood that the actual ratio of fin to body weight will vary by species, the number of fins utilized from each shark, the type of cut used to remove the fins from the carcass, and even the size class of sharks of the same species. Nevertheless, none of the ratios has been amended since the measures were adopted. Second, the measures do not make clear whether the ratio applies to fresh or dried fins, and to what form of the carcass (i.e. whole weight, dressed or partially dressed carcass) the fins are to be compared. These interpretation issues, along with the difficulties of weighing fins and carcasses in an enforcement setting, have led some countries to replace fin-to-carcass ratios with national requirements for fins to remain attached to the carcasses until landing (IUCN, 2013b). While similar measures have been discussed within t-RFMO fora, to date no t-RFMO has adopted a fins-attached policy. Summaries of national shark finning regulations that may apply to pelagic longline fisheries can be found in Camhi *et al*. (2009) and Fischer *et al*. (2012).

As discussed in Section 2.2.3, the WCPFC is the only t-RFMO to have formally evaluated compliance with its finning controls in longline fisheries, in part because it is the only t-RFMO to have a comprehensive regional longline observer programme. As of 2010, there was little evidence from the longline fishery that finning rates had decreased (Clarke *et al*., 2013). In that year, only 11 of the 32 WCPFC members required to report to the commission on their implementation of the shark measure (Conservation and Management Measure [CMM] 2010-07) provided specific confirmation of either implementation of the 5 percent rule or an alternative measure in national waters (e.g. requiring fins to be attached, banning shark fishing or fin trade, or controlling shark mortality under a quota management system). Beyond this lack of confirmation of implementation, the degree of compliance is also often not reported. Therefore, for the WCPFC, as well as for the other t-RFMOs, the extent of shark finning that continues in their longline fisheries is difficult to quantify (Clarke, 2013). There is also very little information on the levels of monitoring and enforcement, and the compliance rates, with national shark finning regulations beyond anecdotal accounts in the media. Compounding these issues, even full compliance with the t-RFMO finning regulations will not necessarily reduce shark mortality as these regulations allow whole sharks to be retained. In fact, evaluation of the WCPFC finning controls implemented since 2008 appears to show only a negligible decrease in shark mortality (Clarke, 2013).

2.3.2 Catch controls

Although global shark catches remain for the most part unmanaged, several countries have set catch limits or quotas for some species. In addition, in some fisheries retention of certain species has been banned such that any inadvertently caught sharks of those species must be released promptly and without further harm (i.e. even if already dead).

In the WCPO, trip limits for retained sharks have been implemented by Australia for the Eastern Tuna and Billfish Fishery; a quota management system for 11 species comprising 85 percent of all shark catches, and including the common longline-caught species blue, shortfin mako and porbeagle sharks, has been implemented by New Zealand; and Papua New Guinea, Samoa and Tonga have set shark catch limits[10] (Clarke, 2011; Fischer *et al*., 2012; Clarke, 2013). The United States of America also applies quotas in its Atlantic shark fishery for sandbar sharks, non-sandbar large coastal sharks, small coastal sharks, and pelagic sharks (Fischer *et al*., 2012). Other national catch prohibitions or limits for elasmobranchs imposed by countries ranked in the top 26 for elasmobranch capture production include *inter alia* (Fischer *et al*., 2012):

[10] Fiji may soon set such shark catch limits (Clarke, 2013).

- Argentina – for rays and narrownose smooth-hound (*Mustelus schmitti*);
- Australia – for a number of shark and ray species, which varies by state;
- Canada – prohibition on catching white, basking and bluntnose sixgill sharks (*Hexanchus griseus*);
- European Union (Member Organization) – a quota system exists and recent quotas of zero have been set for dogfish (spurdog, *Squalus acanthias*), porbeagle and all deep sea sharks.
- Republic of Korea – prohibition on catching CITES-listed species;
- Malaysia – prohibition on catching CITES-listed species;
- Mexico – no retention of 12 species including great white, basking and whale sharks as well as sawfishes and rays;
- New Zealand – prohibition on catching basking shark, great white shark, whale shark, deepwater nurse shark (*Odontapsis ferox*), manta ray, spinetail devil ray (*Mobula japonica*), and oceanic whitetip shark;
- Thailand – prohibition on catching whale sharks;
- United Kingdom – prohibition on catching basking, white, angel and school (tope) sharks.

While this list illustrates a range of catch limits (including zero catch) for the major elasmobranch capture production countries, it should be noted that not all of these species would be expected to interact frequently with pelagic longline fisheries.

In addition to national catch limit and catch prohibition regulations, the t-RFMOs also have "no-retention" measures for particular species considered to be threatened by the fisheries they manage. These measures prohibit retaining any part or whole carcass of the designated species and have been adopted as follows: by the IATTC for the oceanic whitetip shark; by the ICCAT for bigeye thresher, oceanic whitetip, hammerhead (except *S.* tiburo), and silky sharks; by the IOTC for all thresher sharks and the oceanic whitetip shark; and by the WCPFC for the oceanic whitetip shark.[11] Most of the measures also have language calling for the sharks to be released promptly and unharmed. It is not possible given existing information to evaluate the effectiveness of these no-retention measures. This is in large part because t-RFMO members are not consistent in their reporting of discarded sharks and, thus, in cases of zero reported catches it is not possible to determine whether these species are now being discarded or are no longer being caught at all. For example, China reported to the WCPFC that, in response to the no-retention measure for oceanic whitetip, it notified fishers of the prohibition on landing, and catches of oceanic whitetip of 532 tonnes in 2010 dropped to zero for 2011 (WCPFC, 2012d). Recently, the ICCAT has implemented two measures that require improvements in shark data submissions and are designed to support compliance reviews beginning later in 2013 (Recommendations 11-15 and 12-05).

2.3.3 Operational and gear controls

Several types of longline gear configurations are likely to result in higher catches of elasmobranches, and are used either to target sharks intentionally or for other, unrelated reasons. In contrast, other gear configurations may serve to reduce shark interactions without sacrificing target catch. Fisheries that wish to reduce elasmobranch catches have in some cases implemented regulations that either ban or require particular gear components or fishing practices.

The most common gear control to reduce shark catches is a ban on wire (or steel) leaders. This measure is reported to be implemented by Australia (for its Eastern Tuna and Billfish Fishery), the Cook Islands, Fiji (domestic longline fishery), the Marshall

[11] The specific measures are: IATTC Resolution C-11-10; ICCAT Recommendations 09-07, 10-07, 10-8 and 11-08; IOTC Resolutions 12/09 and 13/06; and WCPFC CMM 2011-04.

Islands, Palau, Samoa and South Africa (Clarke, 2011; Fiji Fisheries Department and Forum Fisheries Agency, 2013; Gilman *et al*., 2007a; RMI, 2011; Clarke, 2013). It should be noted, however, that the presence of wire leaders may not necessarily indicate shark targeting. Some fisheries, such as the deep-set Hawaii longline fishery targeting bigeye tuna, prefer to use wire leaders in 90 percent of the sets. Almost all of the sharks caught on this gear (92 percent) are released alive (WCPFC, 2013a). Part of the reason wire leaders are used in this fishery is to facilitate branch line weighting to avoid seabird interactions. There are concerns that banning wire leaders might contribute to less use of line weighting as attaching weights close to hooks on nylon leaders may result in the weight snapping back toward the crew if the branch line breaks, causing serious injury or death (Gilman, 2011). While bans on wire leaders have been discussed in some t-RFMO fora, no t-RFMO has yet adopted this form of gear control for shark management purposes.

Other forms of gear modification that are expected to result in increased shark catches include shark lines and shark baits as described in Section 2.1.2 (see Bromhead *et al*., 2012). There are no known national or regional bans on this type of gear. However, the presence of such gear (and perhaps also the presence of wire leaders) may be considered evidence of shark targeting, which is not allowed under certain national regulations and/or licence conditions (e.g. in the Federated States of Micronesia [McCoy, 2006]). Although the use of circle hooks may mitigate interactions between sharks and longline fisheries and/or reduce mortality,[12] circle hooks are not known to be required as a shark mitigation measure *per se* in any pelagic longline fishery. Some research has explored whether limiting the length of time longline gear remains in the water between the end of setting and the beginning of hauling can reduce shark catches (Carruthers, Neilson and Smith, 2011), but to date there are no known examples of shark management using soak time rules.

2.3.4 Closures

As applied in this section, the term "closures" refers to management measures that prohibit fishing for or catching any species of shark, either permanently or for a fixed period (e.g. seasonal closures on a recurring annual basis). It should be noted that this definition differs from the standard usage of the term closure to indicate that the area as a whole is closed to fishing. Species-specific management measures such as no-retention and setting of catch limits, including zero catch, are discussed in Section 2.3.2.

Permanent shark closures are often referred to as "shark sanctuaries" although it should be noted that many such designations do not offer full protection for sharks, e.g. subsistence catches are not controlled. In 2005, Palau was the first country to designate the whole of its national waters as closed to commercial shark fishing (Clarke, 2013). More recently, shark sanctuary designations have been reported for the Marshall Islands, Tokelau, Maldives, Honduras, the Bahamas, French Polynesia, the Cook Islands and New Caledonia (RMI, 2011; Eilperin, 2012; Agence France-Press 2013). A number of other countries, states and territories can be found in various media and non-governmental organization (NGO) reports of shark "sanctuaries", fishing moratoria, or other forms of marine protected areas, but it is not possible to verify what degree of management is being achieved in these zones.

Pelagic longline fisheries operate within some of the designated shark "sanctuaries" and the specific provisions of each designation are presumably enforced through licensing conditions. To date, none of the t-RFMOs uses closures as shark management measures for longline fisheries.

[12] See Section 2.4.2 for further discussion of which types of circle hooks (e.g. hook shape and width parameters) and which combinations of circumstances (e.g. other mitigation measures, operational characteristics, species/sex/size of sharks) would be most likely to achieve lower interaction or mortality rates.

2.3.5 Trade controls

The primary instruments for controlling trade in sharks as a means of conserving and managing populations is CITES (discussed in Section 2.1.3). All CITES-listed species that frequently interact with longline fisheries (i.e. oceanic whitetip, hammerhead and porbeagle sharks and manta rays) are listed on Appendix II of the convention, which requires that export permits only be issued if the specimen was legally obtained and if the export will not be detrimental to the survival of the species (i.e. non-detriment finding). All countries that are parties to the convention, and have pelagic longline fisheries catching these species, are expected to comply with these requirements unless they take a reservation to the CITES listing.

In addition to the restrictions on trade for CITES-listed species, some countries, states or territories have banned the trade or possession of shark fins regardless of their origin (Clarke, 2011). Unlike the CITES trade controls, which presumably will reference existing fisheries management systems as a basis for non-detriment findings, these shark fin possession bans appear to prohibit any trade or consumption of shark fins even if the fins were obtained in compliance with all applicable fishery regulations.

2.3.6 National plans of action

The birth of international shark conservation arguably occurred with the adoption in 1999 of the International Plan of Action (IPOA) – Sharks under the FAO Code of Conduct for Responsible Fisheries (FAO, 1999a). The IPOA-Sharks was designed to encourage States to develop national plans of action (NPOAs) for sharks (including skates, rays and chimaeras) caught by all types of fishing gear and to manage and conserve these species. A recent review of the status of NPOA development and implementation surveyed 143 countries/entities that report shark catches to FAO and found that one-third had adopted their own NPOA-Sharks. However, if only the top 26 shark-fishing nations are considered, two-thirds have NPOAs (Fischer *et al*., 2012). Implementation of adopted NPOA-Sharks was noted as a further challenge. The most pressing issues limiting implementation were cited as institutional weaknesses, lack of trained personnel and budget shortfalls for research and monitoring (Fischer *et al*., 2012).

With regard to the role of NPOAs-Sharks in managing pelagic longline fishery shark catches, individual NPOAs would presumably package and provide national context for specific management measures such as finning controls that have been adopted by the t-RFMOs of which each country is a member, as well as supplement these with any country-specific controls. Fischer *et al*. (2012) state that three of the t-RFMOs require that their members adopt NPOA-Sharks. In fact, while the actual requirements placed on members may differ based on the legal framework of each convention, it is unlikely that any t-RFMO can require NPOAs, and none of them formally evaluates a member's compliance based on whether an NPOA has been implemented. In the case of the WCPFC, the shark measure (CMM 2010-07) refers to NPOAs in a non-binding, preambulatory section of the measure. In addition, all of the binding elements of the measure are subject to a clause that allows coastal States to apply alternative measures for "exploring, exploiting, conserving and managing" sharks in national waters.

2.4 REVIEW OF MITIGATION METHODS

This section describes mitigation methods that have been proposed, and to some extent tested, to reduce interactions between pelagic longline gear and elasmobranchs, but have not been incorporated into formal management measures. These mitigation methods are discussed in categories relating to modification of fishing behaviour, modification of fishing gear, repellents/deterrents and handling practices.

An inherent difficulty with literature reviews of this type is that they require comparisons across studies for which not all experimental factors were held constant.

For example, even trials of the same mitigation measure in the same area by the same vessel may be affected by differences in environmental conditions such as temperature and wind speed. Contrasts across studies multiply the opportunities for extraneous factors to vary and create inconsistent results. Therefore, in the following sections, potential reasons for differences between study results are presented where possible, but the absence of such discussion should not be taken to mean that the studies are otherwise directly comparable. Even with perfect information, optimal mitigation strategies will vary by fishery; thus, the studies reviewed below will at best serve as a starting point for further fishery-specific research.

2.4.1 Modifying fishing behaviour

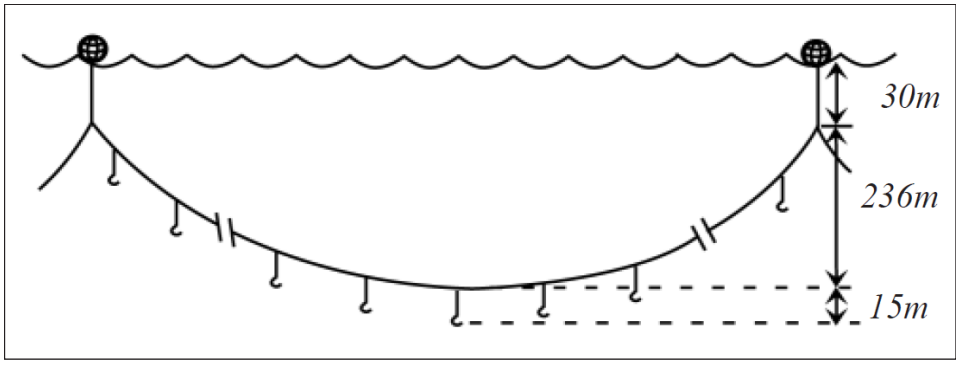
Fishers earn their living by understanding where, when and how to catch target species and avoid wasting time and gear catching unwanted species. As this behaviour is part of normal fishing operations, it can be difficult to identify specific practices as mitigation measures. Nevertheless, a comprehensive set of interviews with longline fishers from 12 fleets around the world revealed that several fleets will either shift fishing grounds if shark interactions are high (n=7) or avoid areas of high shark interactions based on past experience or communication with other vessels (n=5). Only one fleet, the Eastern Australian Tuna and Billfish Fishery, reported using other fishing behaviour modifications such as setting deeper, avoiding specific areas of sea surface temperature (SST), setting during daytime, or minimizing soak time, to reduce shark interactions (Gilman *et al*., 2007a). In some of these fleets, it is likely that shark interactions are not sufficient disincentive to continuing normal fishing operations either because sharks are desirable catch or because interactions are rare.

In addition to being difficult to identify as mitigation measures *per se*, particular changes in fishing behaviour may be more or less effective in different fisheries. For example, although it is well known that higher shark catch rates for some species occur on the shallower hooks located along the catenary curve formed by the main line between floats (Rey and Muñoz-Chapuli, 1991), and the shallow-set sector of the Hawaii longline fishery is known to have high shark catch rates (Walsh, Bigelow and Sender, 2009), an experiment in Hawaii to remove branch lines fishing shallower than 100 m did not demonstrate any significant reduction in shark catch rates (Beverly *et al*., 2009; Figure 18). In contrast, a study of the Spanish swordfish fleet in the South Pacific found that increasing wind velocity up to a speed of 12 knots led to higher shark catch rates probably because it caused the gear to fish at shallower depths (Vega and Licandeo, 2009). Variation in the effectiveness of deep setting as a mitigation measure may also be attributed to the mix of shark species in the fishing grounds. For example, shark catch rates in the central Pacific were found to be determined by a number of factors including SST, set time, moon phase and hook depth, and most species (i.e. blue, silky, oceanic whitetip and pelagic and bigeye threshers) were found to have at least one catch rate factor or relationship apparently unique to that species (Bromhead *et al*., 2012). Similarly, species-specific depth and temperature ranges for pelagic sharks have recently been elucidated by Musyl *et al*. (2011) further suggesting that a shallow–deep approach to mitigating shark catches is overly simplistic.

Compared with the complexities of understanding pelagic shark habitats and the variations in performance of fishing gear, soak time is one element of fishing behaviour that can be straightforward to control.[13] However, its effects on shark catches are complicated by issues related to whether the shark remains on the hook and, if so,

[13] As noted by Carruthers, Neilson and Smith (2011). it is important to carefully calculate actual (effective) soak time, particularly as haulback time (and thus effective soak time) will increase as a function of catch.

FIGURE 18
Diagram showing how the length of float lines can be adjusted to maintain hooks below a certain depth



Source: Beverly, Chapman and Sokimi (2003).

the survival rate for hooked sharks. In particular, sharks that do not survive on the hook are more likely to be depredated or removed by scavengers and never recorded as catch, thus low survivorship may negatively bias catch rates. Survival rates vary by species, and catch rates have been shown to increase with soak time for those species, such as blue shark, that have high on-hook survival (Ward, Myers and Blanchard, 2004; Campana, Joyce and Manning, 2009; Campana *et al.*, 2009b[14]). Other factors believed to influence on-hook survival are shark size, with many studies finding that larger sharks are more resilient (Diaz and Serafy, 2005), and the ability to swim forward once hooked, likely to be a function of branch line length, to ensure adequate ram-ventilation over the gills (Heberer *et al.*, 2010). In contrast to higher survival leading to higher catch rates, it is also possible that higher survival could lead to a greater probability of severing the branch line and escaping (particularly in the case of nylon branch lines), thus leading to lower catch rates. In fisheries where depredation of target species is an issue (Gilman *et al.*, 2007a), it is likely that fishers have already selected soak times to maximize the catch of target species and minimize opportunities for depredation. Therefore, opportunities to reduce soak times in these fisheries may be more limited.

2.4.2 Modification of fishing gear

Other than changing fishing behaviour through shifting fishing grounds, setting deeper, or soaking gear for a different duration, fishers can also influence shark interaction rates through modifying their hooks, branch line materials or bait types. Studies examining the effects of these gear modifications are often complicated, either internally or across studies, by synergies between materials making it difficult to isolate the effects of any one material change.

There is a voluminous body of literature on the subject of circle hooks,[15] and their testing with various leader and bait types. However, an important contribution to understanding the effectiveness of circle hooks was made by a recent symposium's compilation and synthesis of studies (Serafy *et al.*, 2012). One of the most important results of the symposium was a proposed definition of a circle hook as a hook with (Figure 19):

[14] However, see Carruthers, Neilson and Smith (2011), who found no increase in blue shark catch rates with increasing soak time.

[15] A comprehensive catalogue of hook types is available at: www.iattc.org/downloads/hooks-anzuelos-catalogue.pdf

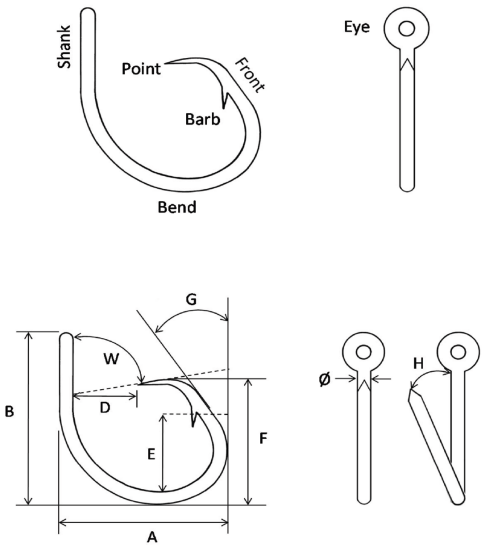
- a point angle (Figure 19, W) of at least 90 degrees; and
- a front angle (Figure 19, G) of at least 20 degrees; and
- a front length (Figure 19, F) of 70–80 percent of the total length of the hook.

A defining distinction between a circle hook and the traditional J hook is thus that for the J hook the point is parallel to the hook shaft (i.e. the point angle is 0 degrees). "Japanese tuna hooks" are intermediary to the circle and J hooks with the point angle somewhere between 0 and 90 degrees (FAO, 2010).

Use of this standardized definition by researchers is expected to facilitate future interpretation of study findings, although it was acknowledged that continuing differences in hook sizing methodologies, as well as other differences among hooks meeting this definition, may still lead to variable results (Serafy *et al*., 2012).

The symposium also found that it is critical to distinguish whether circle hooks result in lower hooking rates for sharks (i.e. sharks are less frequently engaged by circle hooks), lower hooking mortality for sharks (i.e. those sharks that are engaged by circle hooks are less likely to suffer mortality on the hook or during haulback), or both. In examining shark hooking rates *per se*, a meta-analysis of 18 studies showed no

FIGURE 19
Basic components and measurements of a circle hook



Note: A = width; B = length; D = gape; E = throat; F = front length; W = point angle; G = front angle; H = offset angle; Ø = wire diameter.

Source: Serafy et al. (2012).

significant differences between circle hooks and standard/traditional J-shaped hooks (Godin, Carlson and Burgener, 2012). Recent studies not included in the meta-analysis both confirm and refute these findings. One study conducted in the Brazilian longline fishery found no difference in shark catch rates between circle and J hooks (Afonso *et al*., 2012), whereas another study of the Hawaii longline fishery found that wider circle hooks had higher catch rates of blue and oceanic whitetip sharks, and lower catch rates of target species, than did J hooks (Gilman *et al*., 2012). The latter study cites two other studies from the Azores with similar findings of higher shark catch rates on circle hooks. A further study in Uruguayan longline fisheries also found higher catch rates for shortfin makos on circle hooks but only on one of the two types of longline gear used in the study (Domingo *et al*., 2012). It appears from the meta-analysis that which hook type has higher catch rates may vary by species (e.g. catches of pelagic stingray in

particular are higher on J hooks) although the results were not statistically significant (Godin, Carlson and Burgener, 2012).

Turning from the issue of hooking rates to hooking mortality adds another layer of complexity to the discussion. This is because hooking mortality is clearly related, *inter alia* (e.g. see soak time discussion above), to trauma suffered during haulback, which is in turn probably related to where in the shark the hook engages. In the meta-analysis, of eight studies that addressed this issue, six of them found that sharks captured on circle hooks were more frequently hooked in the mouth or jaw rather than internally in the oesophagus or gut (Godin, Carlson and Burgener, 2012; Serafy *et al*., 2012). This effect, which is known from experiments with other species as well, is believed to be due to the fact that a circle hook with little or no offset does not engage, even if swallowed, and as the animal pulls away from the leader and turns, the hook is pulled outward and rotates, catching in the corner of the jaw (Epperly *et al*., 2012; Afonso *et al*., 2011). While it can thus be assumed that circle hooks have a higher probability of jaw hooking, this may be either beneficial or detrimental to survivorship. Jaw hooking is likely to cause less trauma to the shark during haulback than would gut hooking, which frequently results in evisceration. However, jaw hooking may lower the probability that the shark can bite through the leader since its teeth will be less likely to come into contact with the leader material. If the leader is made of wire, the shark is unlikely to bite through it in any case, but if the leader is made of nylon (monofilament), the shark may be better able to "bite off" if it is gut hooked on a J hook. This scenario was noted in the Brazilian longline fishery where 97 percent of the "bite offs" occurred on nylon leaders, and significant differences in shark catch rates between wire and nylon leaders were observed only for J hooks (Afonso *et al*., 2012). Potential interactions between hook type and leader types are thus critical considerations when designing and evaluating mitigation measures. For example, it would be important to assess whether the banning of wire leaders in conjunction with switching from J hooks to circle hooks would result in higher or lower shark hooking and at-haulback mortality rates. Moreover, mortalities that occur after sharks have bitten through the leader, but retain the hook (or hooks) in their jaw, oesophagus or gut with whatever effects that may entail, remain largely unstudied and cannot be quantified (Ward *et al*., 2008; Gilman *et al*., 2008; Godin, Carlson and Burgener, 2012).

A further complicating factor is the selection of bait type. This issue has mainly been explored in longline fisheries through comparison of mackerel and squid baits. Both the meta-analysis and an earlier survey of longline fishing practices around the world concluded that the use of squid baits would result in higher shark catch rates (Gilman *et al*., 2008; Godin, Carlson and Burgener, 2012), whereas another study found higher blue shark catch rates with mackerel bait (Coelho, Santos and Amorim, 2012b). One study also suggests an interaction between bait type and hooking position: in two of three species of sharks examined (blue and porbeagle), the probability of gut hooking increased when mackerel baits were used (Epperly *et al*., 2012). These results indicate that bait type may have a similar confounding role as leader material when evaluating the mitigation effects of hook type. In particular, if nylon leaders are expected to be effective in allowing sharks to "bite off" and escape, the degree to which a higher probability of jaw hooking through the use of circle hooks and squid baits would offset this benefit should be considered. In addition to the traditional squid and mackerel baits, it may become possible in the future to use artificial baits designed to repel sharks or other unwanted bycatch (e.g. using necromones, see Section 2.4.3), while at the same time enhancing selectivity for target species and reducing demand for bait fish (Bach *et al*., 2012).

One final consideration was highlighted from a paper testing similarly sized circle hooks and J hooks in the Gulf of Mexico (Hannan *et al*., 2013). This study found that circle hooks caught relatively smaller fish and showed a higher catch rate

than J hooks for Atlantic sharpnose (*Rhizoprionodon terraenovae*) and blacknose (*Carcharhinus acronotus*) sharks. These results suggest that while the circle hooks had a narrower minimum width,[16] for smaller sharks catch rates could increase with circle hooks. Therefore, species and size-specific vulnerabilities should be considered when designing mitigation strategies. Further studies of differences between species in mouth shape, approach and bait ingestion have been suggested as a means of designing new hooks or bait attachment strategies (Jordan *et al*., 2013).

2.4.3 Repellents and other deterrents

Shark catch mitigation options that do not change catches of target species nor entail major changes to fishing operations will have an automatic advantage in being accepted by fishers. For this reason, mitigation methods that exploit the differences in sensory systems between elasmobranch and bony fishes have gained considerable attention in recent years but have yet to show sufficiently widespread effectiveness and practicality. Shark repellents or deterrents tested thus far include ferrite magnets, rare earth electropositive metals, rare earth electropositive metal / magnet combinations, electrical currents and chemical surfactants.

Most electrical current and chemical studies have not shown these methods to be clearly effective as shark deterrents (Sisneros and Nelson, 2001; Marcotte and Lowe, 2010). However, recent studies testing shark necromones (chemicals extracted from decaying shark tissues) demonstrated that aerosol exposure in the field caused all feeding activities to stop in blacknose and Caribbean reef sharks (*Carcharhinus perezi*, Stroud *et al*., 2013). While shark necromones may thus warrant further research, Jordan *et al*. (2013) point out that, as sharks are known to feed on other sharks, necromones may in some cases be more attractive than repulsive. Moreover, it should be noted that all of the electrical and chemical methods face challenges for longline fishery application in terms of method of delivery, as well as assessment of ancillary environmental effects.

Rare earth electropositive metals (of which the lanthanides – elements 57–71 in the periodic table – are a commonly utilized group), ferrite magnets and combinations of the two have been developed and tested as a means of deterring sharks from taking baited longline hooks. The principle behind their application is that elasmobranchs are sensitive to and repulsed by magnetic fields through their ampullae of Lorenzini, a structure through which elasmobranchs sense and perhaps orient to the earth's geomagnetic field (Kaimmer and Stoner, 2008; O'Connell, Stroud and He, 2012). Because bony fishes do not possess these jelly-filled canals for electroreception, they are believed to be insensitive to weak magnetic fields, but scientific confirmation of a lack of adverse effects on bony fishes is yet to be provided (O'Connell, Stroud and He, 2012). The difference between the two materials is that ferrite materials are permanently magnetized and create a weaker magnetic field, whereas electropositive materials react with seawater to create a stronger magnetic field but corrode as a result and often become unusable in a matter of days (O'Connell *et al*., 2011; O'Connell, Stroud and He, 2012). Rare earth materials trialled thus far include cerium, lanthanum, neodymium and praseodymium, mixtures of which are sometimes referred to as "mischmetals" (Kaimmer and Stoner, 2008; Tallack and Mandelman, 2009; Brill *et al*., 2009; Robbins, Peddemors and Kennelly, 2011; Hutchinson *et al*., 2012).

Testing of shark deterrent magnets and electropositive metals has produced mixed results in different locations with different species. In some cases, even studies involving the same species in different locations have been inconsistent. For example, studies of sandbar sharks in Virginia and Hawaii (Brill *et al*., 2009; Hutchinson *et al*., 2012), studies of spiny dogfish in the Pacific and Atlantic (Kaimmer and Stoner, 2008; Tallack and

[16] See Figure 27 for illustration of a circle hook's minimum width, a parameter not described by Serafy *et al*. (2012) in Figure 19.

Mandelman, 2009), and studies of Galapagos sharks in Hawaii and Australia (Wang, McNaughton and Swimmer, 2008; Robbins, Peddemors and Kennelly, 2011) showed contrasting significant and non-significant reductions, respectively, in catch rates when using lanthanide metals. Potential explanations for these intraspecific differences include differences in hunger levels, shark density/competition, size, or plasticity in feeding strategies under different environmental conditions (e.g. visibility, salinity) (Hutchinson *et al*., 2012; O'Connell, Stroud and He, 2012). Interspecific differences may be related to varying sensory capabilities, e.g. hammerheads rely heavily on their electrosensory nerves, whereas makos preferentially use vision, and blue sharks use olfaction (Hutchinson *et al*., 2012).

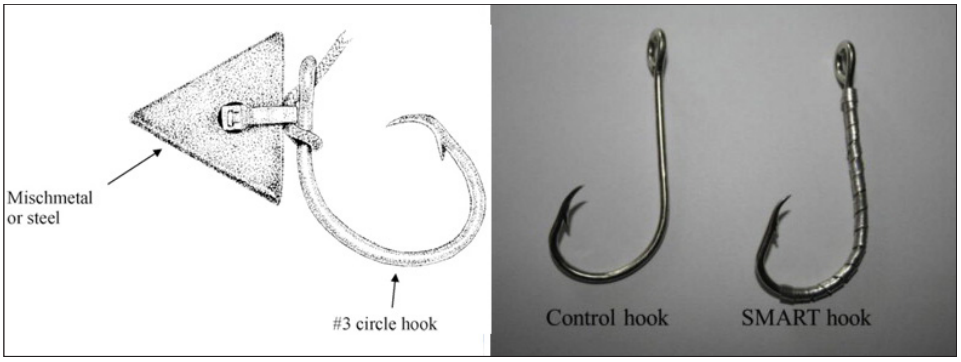
The deployment of magnets and rare earth metals in conjunction with longline gear has been trialled both with direct incorporation of these materials into the hooks (O'Connell *et al*., 2012) and through attachment of discs, weights or plates at varying positions proximal to the hook (most other studies; Figure 20). Although a primary attraction of these materials is their differential effect on elasmobranchs, there are some suggestions that the physical structure of the magnets may affect the behaviour of the branch line in ways that can reduce the catch of target species (Cosandey Godin *et al*., 2013). In addition, since as the rare earth metals create their magnetic fields through chemically reacting with seawater, the need to place these materials in, on or near every hook, in combination with dissolution time frames as short as two days, and a cost of US$20 per kg of material (Stoner and Kaimmer, 2008), is likely to be a significant obstacle to broad-scale implementation.

2.4.4 Handling practices for hooked sharks

Thus far, this discussion has focused on mitigation measures that either reduce elasmobranch hooking rates or reduce the probability of mortality in hooked sharks prior to and during haulback. However, as highlighted by Kaplan, Cox and Kitchell (2007), it is unrealistic to predict reductions in elasmobranch mortality rates without also considering how sharks are handled during haulback and their likelihood of post-release survival. As discussed above, several of the t-RFMOs have adopted mitigation

FIGURE 20

Various methods for incorporating shark repellents/deterrents into longline fishing gear



Notes: Attachment of a rare earth electropositive metal plate to a circle hook (left; Kaimmer and Stoner, 2008); and standard J hook and the same hook modified to incorporate magnetic and electropositive features (right; O'Connell et al., 2012).

measures for certain elasmobranch species that require no-retention and prompt release unharmed, but none of these measures has yet been evaluated for its effectiveness in reducing mortality rates. Recently published studies on post-release mortality can provide some insight but have not always explicitly addressed the issue of handling.

The most important factor to consider when evaluating handling practices for hooked sharks is whether the shark is discarded or retained (whole or in part). It is obvious that a shark that is retained in any form will not survive, and that a discarded shark will survive with some probability depending on the stress and injury to which it has been subjected prior to release. Quantifying this stress and injury (or more directly, the actual survivorship of the animal through tagging) has been explored for several different types of fishing operations.

Two North American fisheries provide contrasting views of the degree to which post-haulback handling increases mortality. Application of pop-up satellite tags to blue sharks released in the Hawaii longline fishery predicted that 95–100 percent of those released in an apparently healthy condition would survive (Moyes *et al*., 2006). In contrast, in the Canadian longline fishery, post-release survival was found to be somewhat lower (about 81 percent; Campana, Joyce and Mannin, 2009). The authors of the latter study attributed this, in part, to more severe handling techniques such as ripping the hook out of the fish (which occasionally removed the jaw) and body-gaffing practised by some vessels in the Canadian fishery. Similar high-impact hook retrieval procedures were also reported for several longline fisheries surveyed by Gilman *et al*. (2008). While the authors of the Hawaii longline post-release mortality study agreed that there are important differences in expected mortality based on handling practices, as well as hook type and soak time, they maintained that their study results reflect actual practices in their fishery (Musyl *et al*., 2009). Which fishery, and which estimated post-release mortality rates, best reflect commercial longline fisheries as a whole remains a topic of discussion (Campana *et al*., 2009).

In order to resolve this issue it will be necessary to obtain better information on the handling practices employed in various longline fisheries, and to explore how these can be improved, taking into consideration the desire to retrieve the gear and maintain crew safety. This is probably best accomplished through improving longline observer coverage and standardizing the recording of information on shark condition when brought to the vessel and, if discarded, at the point of release. Key data types would include: the type and size of hook; the branch line material; the hooking position (e.g. jaw, oesophagus, gut, foul-hooked); the condition at haulback and at release (e.g. alive-uninjured, alive-injured, alive-moribund or dead); and the method of handling between these two observations (e.g. whether hauled on deck, whether hook is retrieved, time the shark is out of the water, whether the shark is struck, stepped on, cut, etc.; see WWF Canada [2012] for examples). Quantification of this information as coded categories would then facilitate analysis, leading to a better understanding of post-release survival rates. Electronic monitoring could also help to provide some of this information. Additional studies involving attaching tags to released sharks should also be pursued to further quantify post-release mortality.

2.5 CONCLUSIONS REGARDING ELASMOBRANCH INTERACTIONS

Management and mitigation of elasmobranch interactions in pelagic longline fisheries presents special problems because of these species' unique position somewhere along the spectrum between explicit target species and undesirable bycatch. Sharks have for decades been under-reported in catch records, owing both to the retention of fins only and to the lack of any regulatory or economic reason to identify them to species. As a result, the only database that provides any sense of global trends suggests that worldwide elasmobranch catches have fallen 14 percent since their peak in 2003. Given the extent of under-reporting in this database, the actual number of sharks caught,

both currently or in the past, remains a matter of some debate. However, as more and more shark carcasses are being landed and traded, catches identified as sharks (rather than unidentified elasmobranchs) appear to be increasing even as overall elasmobranch catches fall.

Sharks, like tuna, have habitat preferences that often place them at the edges of oceanographic fronts. However, unlike tunas, sharks also display sex-specific distributional patterns owing to mating and pupping cycles. Recent research has begun to elucidate species-specific habitat and feeding preferences, but in general this information is not yet sufficient to predict or mitigate interactions with longline fisheries. Several characteristics of longline operations, including bait type, soak time, hook shape, leader length and material, depth at which the hook is fished, and whether special gear is deployed to target sharks, all may determine shark catch rates but with different effects by species and area. Once hooked, various shark species and sizes have been shown to have varying resilience to haulback and handling processes, and this resilience in combination with fleet- or vessel-specific handling practices will determine mortality rates.

In March 2013, CITES listed 8 new shark and ray species, bringing the total number of listed elasmobranchs to 17, several of which frequently interact with longline fisheries. Sharks and rays are also listed on the CMS and the Barcelona Convention, and 29 species that are threatened with extinction according to the IUCN Red List are known to interact with longline fisheries at some level. Those species considered to be most at risk from longline fisheries worldwide include the makos and porbeagle, the threshers, the hammerheads, and some pelagic requiem sharks such as the silky and oceanic whitetip sharks.

The status of these species varies by region, and while some t-RFMOs have already conducted a number of status assessments for sharks, others are still in the early stages of analysis. In the Atlantic, ICCAT's first two blue and shortfin mako stock assessments either showed the stocks were in good condition or were inconclusive, but the most recent assessments concluded that current levels of catches may be considered sustainable. For porbeagle sharks, however, the assessment for the South Atlantic was inconclusive, and for the North Atlantic it showed an overfished stock with overfishing occurring in some areas. In the EPO, the IATTC has yet to finalize its silky shark stock assessment, but the most recent results suggest that the stock is currently rebuilding. Assessment of the same species, and of the oceanic whitetip, in the Western Pacific concluded that both stocks were overfished with overfishing occurring. A recent blue shark assessment for the North Pacific shows conflicting results based on whether a Japanese commercial catch rate index or a Hawaii-based fishery catch rate index is applied. The IOTC has conducted an ecological risk assessment for sharks interacting with longline fisheries but has yet to undertake any formal shark stock assessments.

All of the t-RFMOs data holdings for sharks suffer from under-reporting and an insufficiency of species-specific records, but new catch reporting requirements are expected to gradually improve the situation. All t-RFMOs implement controls on shark finning by imposing a 5 percent fins-to-carcass weight ratio up to the point of landing. With regard to catch-based controls, no-retention measures have been adopted for oceanic whitetip, bigeye thresher, hammerhead and silky sharks by one or more of the t-RFMOs. A number of national measures complement the t-RFMO measures including NPOAs-Sharks by about one-third of all countries worldwide, at least nine shark "sanctuaries" that prohibit shark catches by some or all vessels in national waters, and quota controls for sharks in a small number of countries. In most cases, for the t-RFMO and the national measures there is, at present, little information documenting to what degree these measures are implemented and to what degree they may be acting to reduce shark mortality.

For those situations in which fishers wish to mitigate impacts on sharks (i.e. either avoid catching them or, if caught, to release them unharmed), there are a number of operational and technology-based approaches available. Operational approaches, such as shortening soak times, fishing hooks deeper, or changing set locations, can serve to reduce shark catches or mortality under some situations, but fishers will probably resist any operational changes that reduce target species catches. Moreover, for those fleets and vessels that do not wish to catch sharks, it is probable that some of these techniques are already in use. With regard to new materials and technologies, numerous trials of circle hooks have produced a confusing array of results that suggest that there is no apparent reduction of shark catches with their use, or that circle hooks actually increase shark catches. Circle hooks, leader type (i.e. wire versus nylon) and bait type may also interact and cause changes in hooking rates by shark species or shark size. Repellents or deterrents attached to hooks have, in some cases, been shown to be effective in trials, but face challenges associated with cost and practicality when expanding to broad-scale implementation. Finally, hooking rates are only one factor in determining whether a shark's interaction with a longline fishery will result in mortality. Until there is an improved understanding of retention rates and discard handling practices across fleets and vessels, considerable uncertainty in estimating mortality will remain.

3. Sea turtles

3.1 OVERVIEW OF SEA TURTLE INTERACTIONS WITH LONGLINE GEAR

3.1.1 Taxonomy and impact characterization

There are seven extant species of sea turtles, all of which are known to interact with pelagic longline fisheries through their distribution. These species are classified into two taxonomic families: Cheloniidae (turtles with shells covered by scutes [horny plates]) and Dermochelyidae (turtles with leathery skin). The Cheloniidae include the green (*Chelonia mydas*), hawksbill (*Eretmochelys imbricata*), flatback (*Natator depressus*), loggerhead (*Caretta caretta*), Kemp's ridley (*Lepidochelys kempii*), and olive ridley (*Lepidochelys olivacea*) sea turtles, whereas the leatherback sea turtle (*Dermochelys coriacea*) is the only species in the Dermochelyidae family. Most sea turtles hatch in nests on sandy beaches, grow in offshore waters, return to coastal waters to mature and mate, and then return to shore (females only) to nest and lay eggs thereby completing the life cycle (Figure 21). Two species that do not follow this general pattern are leatherback turtles, which generally remain in pelagic habitats, and flatback turtles, which remain in coastal habitats throughout their lives (FAO, 2010).

Sea turtles have been utilized for their eggs (all species), meat (primarily green), shell (primarily hawksbill but also green), leather (primarily olive ridley but also green), and in some cases oil (e.g. leatherback) for many centuries (IUCN, 2013a). Available records suggest that exploitation rates peaked from the 1950s to early 1970s with almost half the catch occurring in Mexico (Mancini and Koch, 2009). At that time, recognition of severe declines in several populations worldwide resulted in implementation of national and international protection measures (see Section 3.1.3). Nevertheless, exploitation for consumption continues either as part of traditional cultures that are exempted from the established conservation measures (IUCN, 2013a) or by poachers supplying the illegal trade (Mancini and Koch, 2009).

In addition to anthropogenic take of adults and eggs, sea turtle populations are threatened by a number of other factors. These include non-human predation of eggs, nesting habitat disturbance and degradation, oceanic climate change and sea-level rise, marine pollution (both direct debris effects as well as pollution-related viruses), boat collisions and unintentional capture in fishing gear (FAO, 2010). Given the number of factors that affect global populations, in combination with a scarcity of data on many of these impacts, it is difficult to quantify the importance of the various threats facing sea turtle species in various regions. Nevertheless, it is widely recognized that interaction with fishing gear is one of the most serious (FAO, 2010; Wallace *et al*., 2011, 2013). A recent expert-judgement-based study ranked fisheries bycatch as the most important threat to global sea turtle populations, followed by climate change (where it could be assessed), human take of meat and eggs for consumption, coastal development and pollutions/pathogens (Wallace *et al*., 2011). Among fisheries, a global comparison of calculated impact scores between three classes of gear types (longlines, nets and trawls), longlines were found to have similar interaction rates and to affect the same size of sea turtles as the other gear types, but had a significantly lower mortality rate and thus had a significantly lower overall impact score (Wallace *et al*., 2013; see Section 3.2 for more details). Nevertheless, it was noted that interaction rates are highly variable among gear and regions (Lewison *et al*., 2013). A number of national and international management

systems have designated mitigation of impacts on sea turtles across all gear types, as well as across the entire range of factors threatening their populations, as an urgent priority (see Section 3.1.3).

Although sea turtles differ from other bycatch organisms in that they have lower taxonomic diversity (i.e. only seven species), and unlike sharks are not secondary targets of most longline fishing operations, it appears that data gaps for sea turtle bycatch are no less severe than for other taxa. An estimate of global bycatch from pelagic longline fisheries in 2000 suggested that more than 200 000 loggerhead sea turtles and 50 000 leatherback sea turtles were caught and "tens of thousands" were expected to have died as a result of their encounters (Lewison, Freeman and Crowder, 2004). However, as with any such study, these estimates are based on available data that are skewed toward fishing fleets that have relatively better management and data reporting systems (and/or more transparency). Moreover, assumptions necessary to extrapolate the available data to a global total have the potential to increase bias in the results. In the case of the 2004 study, there were interaction rates available for less than 30 percent of the longline fishing effort potentially affecting both species (Lewison, Freeman and Crowder, 2004) and some of the necessary extrapolations may have overestimated the interaction rates in the Pacific (FAO, 2010). Another problem compounding mortality estimates is that existing information on post-release mortality in sea turtles remains scarce and is not likely to be representative of species and fisheries worldwide (Swimmer *et al*., 2006; Swimmer and Gilman, 2012).

A more recent study tallied the number of known global sea turtle interactions (for all seven species) with longline gear for 1990–2008 at just under 56 000. While noting that its tallies were based on observer coverage, which is generally less than 5 percent, this study did not attempt to extrapolate the tallies to a global estimate nor did it provide an estimate of mortalities (Wallace *et al*., 2010a).

As these two studies illustrate, while it is useful to monitor and reduce interaction rates, it is essential to distinguish between interaction and mortality rates. This is particularly important for fisheries that routinely and successfully deploy post-capture mitigation measures. Simultaneously, it is necessary to continue efforts to acquire both reliable interaction and mortality data for a broader range of pelagic longline fisheries worldwide in order to better understand total impacts on each species across its range. This kind of approach has recently begun to be used to estimate risk at the population level as a product of interaction rates, mortality rates, productivity and vulnerability, rather than to simply present numbers of individuals effected (Wallace *et al*., 2013, Lewison *et al*., 2013).

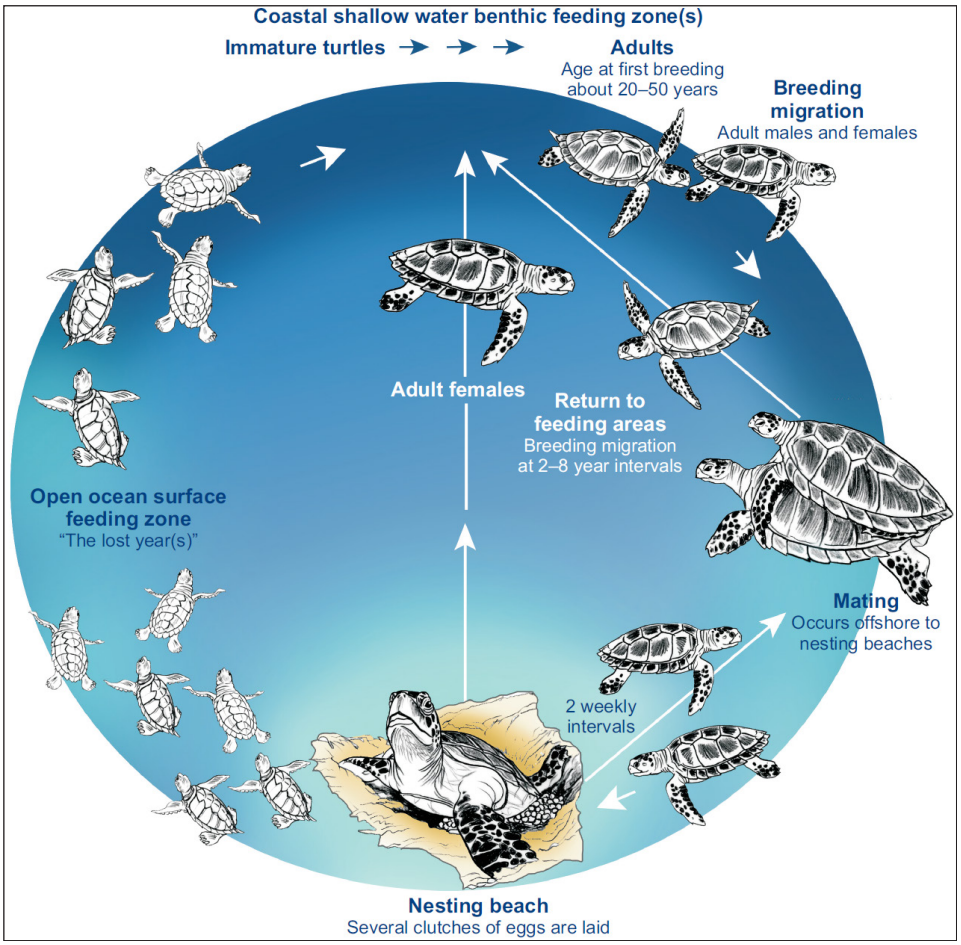
3.1.2 Factors influencing sea turtle interactions and mortality

Before considering the results of studies on specific sea turtle bycatch mitigation techniques and trials, this section presents an overview of broad factors that influence interaction and mortality rates, including species-specific biological features, oceanographic conditions, gear characteristics, and types of sea turtle handling. While some generalizations will not hold for all fisheries worldwide, they are intended to provide biological, oceanographic and fishing technology context for specific discussions to follow.

Sea turtle morphology, distribution and behaviour are all important factors underlying species-specific differences in interaction rates with longline fisheries. For example, studies in the Northwest Atlantic in the early 2000s found that while both loggerheads and leatherbacks can be snagged in longline gear, leatherbacks are 3 times more likely to become entangled and almost 40 times more likely to become foul-hooked (Watson *et al*., 2005). This is likely to be due to a combination of species-specific feeding behaviours as well as lack of a protective shell. The primary issue arising from entanglement in longline fisheries, as in other fishing gear premised on

entanglement (i.e. gillnets), is that the turtle cannot reach the sea surface to breathe. Entanglement may also lead to strangulation or soft tissue trauma during longline haulback or if the turtle is discarded without disentangling it from the branch line

FIGURE 21
Life cycles and main habitats of sea turtles



Source: FAO (2010).

(Balazs, Pooley and Murakawa, 1995; FAO, 2010). Species other than leatherbacks are less likely to be entangled and more likely to be injured by the hook itself (see below). Another biologically based factor that might explain different interactions with longline gear by species is foraging strategy. Loggerheads are observed to have higher variability in interaction rates than leatherbacks and this may be due to the loggerhead's more flexible and opportunistic feeding behaviour (Kot, Boustany and Halpin, 2010).

A number of studies, mainly focused on United States waters in the Atlantic and Pacific, have examined whether interactions with sea turtles can be predicted, and thus avoided, based on knowledge of oceanographic conditions. Most of these studies have found a strong relationship between sea turtle abundance and SST, although the other oceanographic variables examined, and found to be significant, have often varied between studies. Species-specific differences in preferred temperature ranges have also been documented. For example, the leatherback is the only sea turtle able to maintain its body temperature considerably above ambient and is thus less limited in its geographic range than the other sea turtle species (Braun-McNeill *et al.*, 2008; Gardner

et al., 2008). In the Pacific, loggerhead sea turtles were found in association with fronts, eddies and geostrophic currents, often in waters of 15–25 °C, whereas olive ridley turtles occupied warmer waters of 23–28 °C and only some subpopulations were found in association with fronts (Polovina *et al*., 2004). A study of the relationship between sea turtle abundance and temperature in the North Atlantic attempted to define thermoclines of risk for all species and suggested that a conservative approach would be to require mitigation measures when 25 percent of each 0.5 degree latitudinal zone was warmer than 11 °C (Braun-McNeill *et al*., 2008). However, one weakness to this approach noted by that study is that some individuals, particularly larger loggerheads, may be able to tolerate colder waters compared with the other hard-shelled turtles owing to their enhanced thermoregulatory ability (Braun-McNeill *et al*., 2008). Another potential weakness is that, if sea turtles and target species both prefer similar oceanographic conditions, it may be difficult for fishers to operate in areas inhabited only by target species.

Factors other than temperature have also been explored to determine whether they affect sea turtle distribution and interaction rates. Studies of sea turtle occurrences on seamounts have shown either higher abundance (Gilman *et al*., 2012) or no significant effect of the seamount feature (Morato *et al*., 2008). A study of the United States Atlantic longline fishery examined both seasonal and lunar periodicity (note that temperature is expected to often, but not always, vary with season), and found that seasonal patterns were stronger than lunar cycles. Although higher interactions were observed during a full moon it was postulated that this could be due to improved visibility of the bait in brighter conditions (Kot, Boustany and Halpin, 2010). Other studies have found that bottom depth, chlorophyll-*a* concentration and magnetic forces can be significant predictors of habitat usage by loggerhead turtles (Gardner *et al*., 2008; Kobayashi *et al*., 2008). While some of these studies have tried to identify particular areas of high interactions, all of them have concluded that the factors governing sea turtle habitat use are complex and variable in time and space, and therefore require further study. Some of the studies suggest that real-time monitoring of oceanographic conditions may be a necessary, ongoing component of mitigation strategies (Braun-McNeill *et al*., 2008, Kot, Boustany and Halpin, 2010; see also Section 3.4.4).

While the performance of specific types of mitigation devices is discussed in Section 3.3, it is useful to review the potential for interaction with longline gear based on what is known about sea turtle behaviour. As mentioned above, sea turtles are visual predators and as such the potential for interactions should increase when the bait becomes more visible either through brighter ambient light (Kot, Boustany and Halpin, 2010), attachment of attractive light sticks (Wang *et al*., 2007; Crognale *et al*., 2008) or ambient colour contrasts or turtle colour preferences (Cocking *et al*., 2008; Southwood *et al*., 2008; Piovano, Farcomeni and Giacoma, 2012a). However, conflicting results of studies concerning whether minimizing daytime or night-time soak duration or hauling would reduce sea turtle interactions suggest species-specific differences in diurnal behaviour patterns (FAO, 2010).

Another important factor is the depth at which the longline hook is set relative to the habitat utilization of sea turtle species (Wallace *et al*., 2008). In general, deep-set hooks catch fewer sea turtles but those that are hooked are more likely to drown because they cannot reach the surface to breathe. Depth preferences vary by species: loggerheads in the Pacific were found to spend 40 percent of their time at the surface and only rarely (10 percent) were found below 40 m (Polovina *et al*., 2004); olive ridley turtles prefer deeper habitats than loggerheads but are still usually found above 40 m (Polovina *et al*., 2004); and the average depth of a leatherback dive is 62 m (FAO, 2010). However, depth preferences appear to vary by season as well as behavioural phase (e.g. during migration) (B. Wallace, personal communication, August 2013). Therefore, while it may be possible to reduce interactions through hook depth, depending on the

species of concern, some sources recommend fishing all hooks below 40 m whereas others recommend 100 m as a minimum depth (Beverly *et al*., 2009; FAO, 2010; Gilman, 2011).

Although diurnal and depth usage patterns thus appear to vary by species, the feeding processes through which sea turtles take bait from longline hooks are thought to be remarkably consistent. There appear to be two distinct types of bait consumption: sea turtles are known to take small bites of fish bait but swallow rubbery squid bait whole (FAO, 2010; Gilman, 2011). When the texture or threading of the bait on the hook allows the sea turtle to strip it easily, the risk of hooking is reduced. This may be because the turtle can sense and avoid the hook point. Even if the turtle attempts to swallow the bait whole, if the hook is large enough relative to the size of the turtle's mouth and/or the point of the hook is shielded, the risk of hooking will also be reduced (Stokes *et al*., 2011).

Finally, whether or not turtles survive their interactions with longline fisheries depends, as it does for other bycatch organisms, on the degree of injury and stress sustained during haulback and release. However, some special characteristics of turtles are particularly important to take into account. Turtles that appear lifeless after haulback may recover if water is allowed to drain from their lungs and they are kept shaded and damp for up to 24 hours (FAO, 2010). It is not easy to generalize regarding the least and most damaging sea turtle body parts for a hook to lodge in as this depends on whether sensitive structures are damaged in any location. It is also important to consider the probability that the hook can be successfully removed (Plate 2) as the best option may be to leave the hook in place. In such cases, the branch line itself is potentially more dangerous to the health of the turtle than the hook itself. If the line is not cut as close to the turtle as possible, it may tangle around and amputate an appendage or be swallowed and cause a slow and painful death (Parga, 2012).

PLATE 2

The importance of assessing the location and severity of hooking before attempting to remove the hook

Courtesy of NMFS.

3.1.3 Species risk profiles and international conservation initiatives

The threatened status of sea turtles was widely recognized as early as the mid-1970s when the hawksbill became one of the first species to be listed on CITES Appendix I and all international trade was banned. By 1981, all sea turtle species were listed on CITES Appendix I and subsequent efforts to re-open international trade based either on wild population harvests or on sea turtle farming operations have all been unsuccessful (Oceanic Society, 2011).

In the mid-1990s, negotiations began on three intergovernmental treaties designed to promote cooperation on sea turtle conservation among signatories. The first of these, the Memorandum of Understanding Concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa, entered into force in July 1999. A comprehensive management plan, the Nairobi Declaration, was agreed between signatories in May 2002. The second intergovernmental treaty to enter into force was the Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC), which became effective in May 2001 and currently has 15 signatories ranging from the United States of America in the north to Argentina and Chile in the south. This treaty is the only binding international agreement exclusively for the protection of sea turtles (NOAA, 2013a). The third treaty was the Memorandum of Understanding on the Conservation and Management of Marine Turtles and their Habitats of the Indian Ocean and South-East Asia in September 2001. The latter two treaties were concluded under the auspices of the Convention on the Conservation of Migratory Species of Wild Animals of 1979 (CMS or the Bonn Convention). All of the sea turtle species except for the flatback are included on both Appendices I and II of the CMS Convention (CMS, 2012).

Because sea turtles face a variety of threats in various phases of their life cycle, integrated monitoring and assessment is complicated and the relative contribution of fishing impacts is difficult to quantify. In the mid-2000s, FAO led expert consultations and reviews aimed at assessing and prioritizing sea turtle–fishery interactions with a view towards impact mitigation. Pelagic longline fisheries were identified as significant sources of impacts on North and South Pacific loggerheads and Eastern Pacific leatherbacks, both of which have seen nesting population declines of more than 80 percent in the past two decades, as well as to Mediterranean populations of green and olive ridley turtles (FAO, 2010).

Assessing the seven extant species of sea turtles against IUCN Red List criteria has mainly been accomplished on the basis of surveys of reproductive activity at nesting beaches (IUCN 2013a). The results have listed two of the species as critically endangered (hawksbill and Kemp's ridley), two as endangered (green and loggerhead), and two as vulnerable (leatherback and olive ridley). The threat category of the flatback sea turtle could not be classified because it is data deficient (Table 9).

Although the number of sea turtle species is small, and most have been listed in one of the "threatened" categories[17] since 1986, several species have not been re-assessed for more than a decade (Table 9). This situation reflects the difficulties of discriminating stocks or population units as well as ongoing debate within the IUCN Marine Turtle Specialist Group regarding the usefulness of assigning each species to an IUCN Red List category on the basis of its global status, particularly when the status of regional populations may vary widely (Godfrey and Godley, 2008; Wallace *et al*., 2013). While the updating of IUCN Red List assessments of sea turtles appears to have stalled (with the exception of leatherbacks), recently published research has made great strides in assessing and mapping regional populations and ranking conservation priorities

[17] The IUCN Red List threatened species are those in the categories of "critically endangered", "endangered" and "vulnerable".

(e.g. Wallace *et al*., 2010a, 2010b, 2011, 2013). These studies first delineated regional management units (RMUs), i.e. spatially defined population units, on the basis of nesting sites as well as marine distribution (Wallace *et al*., 2010b), and then combined them with risk scores based on, *inter alia*, the population's size, trajectory and vulnerability (Wallace *et al*., 2011) to identify the top ten healthiest and top ten most threatened sea turtle populations (Figure 22).

TABLE 9
IUCN Red List categories, date of last assessment and distribution for the seven extant species of marine turtles

Species	Common name	IUCN Red List category	Last assessed	Distribution
Dermochelys coriacea	Leatherback	Vulnerable*	2013	Circumglobal, tropical to temperate; the species' stronghold is the Atlantic
Eretmochelys imbricata	Hawksbill	Critically endangered	2008	Circumglobal; the most tropical of the marine turtles; found throughout Central America and the Indo-Pacific
Lepidochelys kempii	Kemp's ridley	Critically endangered*	1996	Atlantic coastal waters of the United States of America and Mexico; some records from the coastal eastern Atlantic and Mediterranean
Caretta caretta	Loggerhead	Endangered*	1996	Circumglobal, tropical to temperate; widely distributed in the Atlantic and Pacific, more limited to coastal regions in the Indian Ocean
Chelonia mydas	Green	Endangered	2004	Circumglobal; widely distributed in tropical and subtropical waters
Lepidochelys olivacea	Olive ridley	Vulnerable	2008	Circumglobal; tropical, and to a lesser extent, subtropical Atlantic, Indian and Pacific Oceans
Natator depressus	Flatback	Data deficient*	1996	Coastal waters of northern Australia, extending to Indonesia and Papua New Guinea

Note: An asterisk indicates those listings that are annotated by IUCN as "needs updating".

Sources: FAO (2010), Wallace *et al*. (2010b), IUCN (2013a), Wallace *et al*. (2013).

These RMU-specific risk scores were then assessed against "bycatch impact scores" calculated on the basis of interaction rates, mortality rates and body sizes for longline fisheries in each RMU (Wallace *et al*., 2013). By integrating all of these results, conservation and monitoring priorities were grouped into four categories of high/low risk and high/low impact (Figure 23). The four RMU–gear combinations that were found to be both high risk and high impact were the leatherbacks in the Southwest Indian and Southwest Atlantic Oceans, the loggerheads in the South Pacific, and the green sea turtles in the Mediterranean (Wallace *et al*., 2013).

3.2 SEA TURTLE INTERACTIONS BY AREA

A primary source of data on interactions between sea turtles and pelagic longline fisheries in each ocean should be the t-RFMO operating there. However, the sea turtle assessment and management programmes of the t-RFMOs are for the most part not yet well developed, and in many cases they are plagued by a lack of data submission from members. The discussion under each of the ocean basins below begins with a review of existing information on interaction rates, mortality and stock status from the scientific

literature, and then supplements this, where possible, with additional information produced by the t-RFMOs.

3.2.1 Atlantic

The Atlantic is home to all of the sea turtle species except for the flatback turtle. Interaction and mortality rates were compiled by Wallace *et al.* (2013) for 71 combinations of species and areas worldwide, and of these combinations 19 interacted with longline fisheries in the Atlantic. Loggerhead sea turtles showed by far the highest interaction rates (bycatch per unit effort [BPUE]) with longline fisheries of any species in this or any ocean (Table 10). Although it is not known why this is the case, it is noted that loggerheads in the Atlantic do not fall within the high risk – high interaction conservation category (Figure 23). In contrast, leatherback and green turtles, which have slightly lower interaction rates, are placed within both high risk and high impact[18] categories (Figure 23).

TABLE 10
Median interaction and mortality rates (not including post-release mortality) by species and area for longline fisheries in the Atlantic

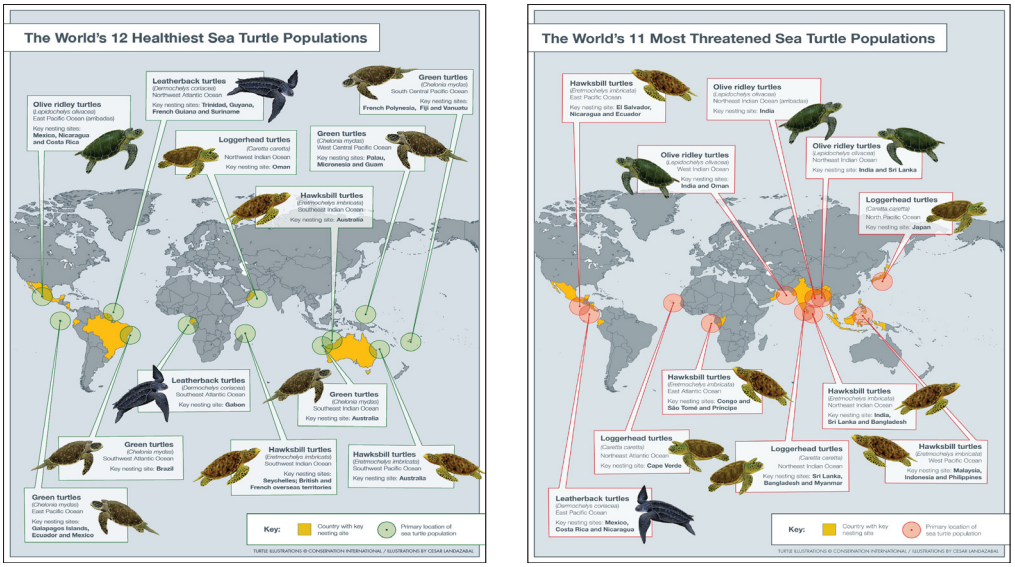
Species	RMU	N	BPUE	MR
Green	E Atlantic	1	ND	ND
	NW Atlantic	1	ND	ND
	Central Atlantic	18	0.139	ND
	Mediterranean	7	0.103	0
	SW Atlantic	30	0.071	0
	S Caribbean	29	0.006	0.02
Hawksbill	E Atlantic	1	ND	ND
	SW Atlantic	1	ND	ND
	W Atlantic	22	0.003	0
Kemp's ridley	NW Atlantic	1	ND	ND
Leatherback	SE Atlantic	31	0.171	0
	SW Atlantic	31	0.171	0
	NW Atlantic	77	0.062	0
Loggerhead	NE Atlantic	23	0.871	0.04
	Mediterranean	70	0.409	0
	SW Atlantic	48	0.407	0.04
	NW Atlantic	144	0.274	0.01
Olive ridley	W Atlantic	12	0.022	0
	E Atlantic	8	0.014	0

Notes: RMU = regional management unit; N = sample size; BPUE = bycatch per unit effort (1 000 hooks); MR = mortality rate (scale of 0 to 1). Longline data presented by Wallace *et al.* (2013) represent a combination of pelagic, bottom, surface/drifting and "other" longline gear; however, no statistically significant differences were found between these gear types in BPUE or MR values overall.
Source: According to Wallace *et al.* (2013).

The best estimates of interaction and mortality rates produced by Wallace *et al.* (2013) represent an integration of data from 1990 to mid-2011 (i.e. spanning a period in which some mitigation measures were introduced). The authors describe this approach as representing impacts on sea turtles over their most recent generation. A different approach was adopted in a paper summarizing interaction and mortality rates in United States fisheries both before and after mitigation measures were implemented

[18] A function of bycatch interaction rates, mortality rates, and body size as a proxy for reproductive value (productivity).

FIGURE 22
Sea turtle population risk rankings based on Wallace *et al.* (2011)



Source: Conservation International, reprinted with permission.

(Finkbeiner *et al.*, 2011). For United States Atlantic and Gulf of Mexico pelagic longline fisheries, the mean annual number of interactions with sea turtle species of any kind was estimated to be 1 600 (maximum of 3 553) prior to regulation and 1 400 (maximum of 2 143) after regulation. Although these interaction figures do not show a major change (i.e. a decline of only 12 percent), post-mitigation annual mortalities were reduced from 100 (maximum of 726) to 20 (maximum of 50), a decline of 80 percent. The main regulatory changes between the two periods for this fishery were described as spatial and temporal closures and mandated use of circle hooks (Finkbeiner *et al.*, 2011). The species that appear to have benefited most from the mitigation measures are loggerhead (both interactions and mortality) and the Kemp's ridley sea turtles (Figure 24). This apparently positive result for the loggerhead sea turtle stands in contrast to recent findings of an overall decline in abundance for the Atlantic loggerhead population, based on nest surveys at Florida beaches, for 1989–2006 (Witherington *et al.*, 2009).

Besides the United States of America, other members of ICCAT have recently produced analyses of sea turtle interaction rates in their fisheries (ICCAT 2012c). Interaction rates based on Brazilian and Uruguayan fleets in the Southwest Atlantic were estimated at 0.9613 loggerheads and 0.1437 leatherbacks per 1 000 hooks. No clear trend in loggerhead interactions between 1998 and 2010 was found when the data were standardized. In contrast to the Brazilian and Uruguayan fleets' interaction rates, estimates for the longline fleets of Taiwan Province of China and Japan for the Atlantic showed a maximum of 0.0311 turtles per 1 000 hooks (mainly leatherbacks) for the fleet of Taiwan Province of China, and maxima of 0.0021 turtles per 1 000 hooks for leatherbacks, and 0.0003 turtles per 1 000 hooks for other turtles, in the Japanese fleet. (Further interpretation of the interaction rates for the fleet of Taiwan Province of China is provided by Huang [2011], who states that the rates were 0.0004–0.0027 per 1 000 hooks at high latitudes as compared with 0.0145 per 1 000 hooks in tropical regions in 2007). ICCAT's Subcommittee on Ecosystems noted that the estimates from the Asian fleets were two to three orders of magnitude lower than those for other fleets and may be biased downward by fishing effort in areas that are not inhabited by turtles.

FIGURE 23

Regional management unit (RMU) risk scores plotted against bycatch impact scores for longline fisheries and used to identify the highest conservation and monitoring priorities

Source: After Wallace *et al.* (2013).

These estimates represent the beginning of efforts by ICCAT to produce an assessment by 2013 of the impact of the incidental catch of sea turtles resulting from ICCAT fisheries as required under the current ICCAT sea turtle management measure (ICCAT Recommendation 2010-09). This measure mandates that members annually report sea turtle interaction data by species starting no later than in 2012, and thus should start to form the basis for a review of whether existing mitigation measures are effective or need to be strengthened. However, as ICCAT's Subcommittee on Ecosystems' initial discussions highlighted, there are several issues that will impede this assessment. First, not all of the members have submitted the required data and many of the submissions, including those described above, did not report on the mortalities associated with the reported interactions. Second, most members will rely on their observer programmes for interaction and mortality data; therefore, low or unrepresentative observer coverage may bias these estimates (see Section 1.2.1). Third, the subcommittee has yet to conduct a thorough evaluation of whether the available interaction rate data can be used to estimate the total sea turtle take by ICCAT longline fisheries. It is perhaps for this reason that the subcommittee is currently planning to give greatest priority to comparing the impact of ICCAT fisheries with non-ICCAT fisheries, and to use an ecological risk assessment approach (PSA) to identify whether there are significant risks arising from fisheries that are not well described (ICCAT, 2012c). Moreover, ICCAT further noted that the assessments would be conducted for the fishery rather than for each sea turtle species and the range of threats it faces throughout its life cycle (ICCAT, 2012a).

3.2.2 Eastern Pacific

The habitats of the EPO are used by leatherback, green, hawksbill and olive ridley sea turtles for both feeding and nesting. Loggerhead turtles do not nest in the EPO but

are widely distributed throughout the area when feeding (IAC 2012). Wallace *et al.* (2013) found seven distinct species–area interactions in the EPO from among which olive ridley (0.127) and hawksbill (0.118) turtles had the highest interaction rates with longline fisheries, and loggerhead and leatherback turtles had the lowest (0.01–0.02). These interaction rates are lower than those calculated for loggerheads in the Atlantic and similar to those for other species. Mortality rates were considered to be negligible for all species–area combinations in the EPO (Table 11).

TABLE 11
Median interaction and mortality rates (not including post-release mortality) by species and area for longline fisheries in the Eastern Pacific

Species	RMU	N	BPUE	MR
Green	E Pacific	30	0.098	0
Hawksbill	E Pacific	9	0.118	0
Leatherback	E Pacific	9	0.016	0
Loggerhead	S Pacific*	23	0.02	0
	N Pacific*	36	0.011	0
Olive ridley	E Pacific	50	0.127	0

* = including N and S Pacific.

Notes: RMU = regional management unit; N = sample size; BPUE = bycatch per unit effort (1 000 hooks); MR = mortality rate (scale of 0 to 1). Data presented by Wallace *et al.* (2013) represent a combination of pelagic, bottom, surface/drifting and "other" longline gear; however, no statistically significant differences were found between these gear types in BPUE or MR values overall.

Source: According to Wallace *et al.* (2013).

Information compiled by the IATTC on sea turtle interactions with pelagic longline fisheries is very limited owing to the lack of an IATTC longline observer programme and a requirement for members to implement longline observer coverage of 5 percent only as of January 2013. As such, only ad hoc reports of interactions and mortalities are available. Japan reported that its longline fleet in 2000, a year in which they reported longline effort at 79.3 million hooks caught 166 leatherbacks, of which 25 died, and 6 000 other turtles, of which half died. This would equate to about 0.08 sea turtle interactions and a mortality rate of 0.038 per 1 000 hooks. Taiwan Province of China has reported interaction rates of 0.015 per 1 000 hooks from the central Pacific for 2002–06 (Huang, 2011).

The Spanish longline fleet targeting swordfish reported an average of 65 interactions per million hooks (0.065 interactions per 1 000 hooks) and 8 mortalities (mortality rate of 0.008 per 1 000 hooks) from 1990 to 2005 (IATTC, 2012a). Fleets like this one that set at shallower depths than fleets such as Japan's, which are targeting bigeye, would be expected to have higher interaction rates with sea turtles (Gaos *et al.*, 2012). The IATTC reports that about 10 percent of its distant-water longline vessels use shallow-set longlines (IATTC, 2012a). Other artisanal longline fleets in the region described in Gillett (2011) sometimes fish for tuna and interact with sea turtles at rates that can be similar to rates in industrial-scale fisheries (Peckham *et al.*, 2007). To mitigate potential impacts on sea turtle populations from longline fisheries in the region, the IATTC has participated in a programme exchanging J hooks for C hooks in several Latin American countries since 2003 (IATTC, 2012a; see Section 3.4.1 below for more details).

3.2.3 Western and Central Pacific

All of the sea turtle species occur in the WCPO except for the Kemp's ridley. The population of greatest concern in this region is the South Pacific loggerhead, although a number of other loggerhead, leatherback and hawksbill populations are considered to be at high risk but have more limited interactions with longline fisheries (Figure 23). Interaction and mortality rates calculated by Wallace *et al.* (2013) covered a total of

16 species–area combinations in the WCPO (Table 12). Interactions in this region are low relative to other areas with rates of generally ≤ 0.01 per 1 000 hooks. The only exception to this is the South Pacific loggerhead with a rate of 0.02 per 1 000 hooks. Mortality rates are also generally low other than in the case of South Central Pacific green turtle where the median mortality rate was 100 percent.

TABLE 12
Median interaction and mortality rates (not including post-release mortality) by species and area for longline fisheries in the Western and Central Pacific

Species	RMU	N	BPUE	MR
Flatback	SW Pacific	ND	ND	ND
Green	NW Pacific	1	ND	ND
	S Central Pacific	6	0.008	1
	W Central Pacific	5	0.003	0.05
	N Central Pacific	13	0.001	0
	SW Pacific	6	0.001	0
	W Pacific	3	0	ND
Hawksbill	SW Pacific	3	ND	ND
	N Central Pacific	4	0.002	0
	W Central Pacific	5	0.002	0.05
	S Central Pacific	3	0	0
	W Pacific	4	0	ND
Leatherback	W Pacific	22	0.005	0
Loggerhead	S Pacific*	23	0.02	0
	N Pacific*	36	0.011	0
Olive ridley	W Pacific	49	0.007	0

* = including N Pacific and S Pacific.

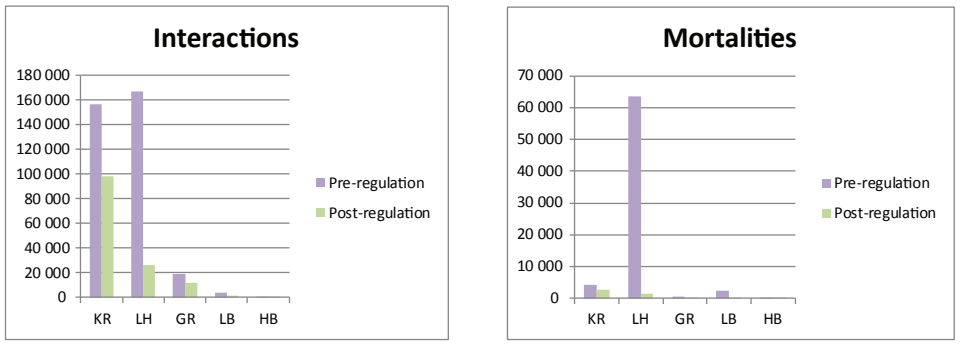
Notes: RMU = regional management unit; N = sample size; BPUE = bycatch per unit effort (1 000 hooks); MR = mortality rate (scale of 0 to 1). Longline data presented by Wallace *et al*. (2013) represent a combination of pelagic, bottom, surface/drifting and "other" longline gear; however, no statistically significant differences were found between these gear types in BPUE or MR values overall.
Source: According to Wallace *et al*. (2013).

Finkbeiner *et al*. (2011) reported interaction and mortality rates that were orders of magnitude lower for United States pelagic longlines in the Pacific[19] versus the Atlantic (Figures 24 and 25). Loggerheads showed the highest interaction and mortality rates both before and after regulations, but for all species both interactions and mortalities were reduced to below 50 individuals per year after mitigation.

According to the sea turtle conservation and management measure adopted by the WCPFC in 2008 (CMM 2008-03), members should report their interactions with sea turtles, as well as their progress with implementation of the "FAO Guidelines to Reduce Sea Turtle Mortality in Fishing Operations", to the Commission in their Annual Reports-Part 2. However, as these reports are not in the public domain, and the WCPFC's Technical and Compliance Committee's Work Plan does not call for a review of compliance with the sea turtle measure until 2015 (WCPFC, 2012d), there is currently no publicly available compilation or evaluation of the interaction or mortality rates of the WCPFC members' longline fisheries on sea turtles. The WCPFC completed an ecological risk assessment in 2007 that found most sea turtle species at high risk from longline fisheries relative to all species captured, but only at medium

[19] Finkbeiner *et al*. (2011) data for the Pacific include the Hawaii longline fishery and the California deep-set longline fishery. Its results are discussed under the WCPO because the Hawaii longline fishery is a larger fishery; and because it has both deep and shallow sets, it has a potentially greater impact on sea turtle populations, than the California deep-set longline fishery.

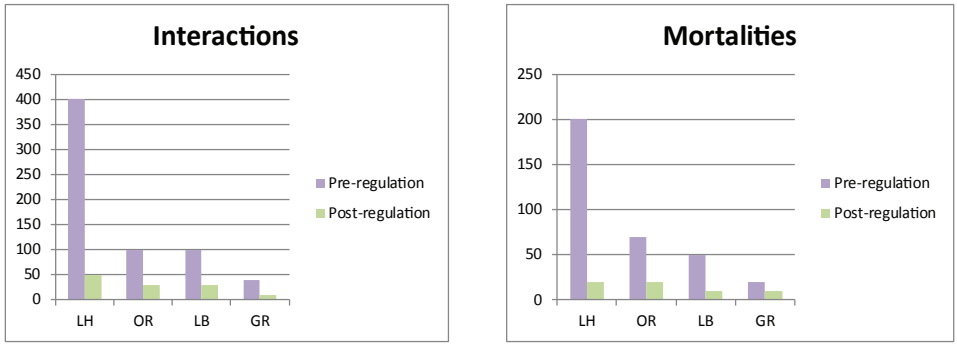
FIGURE 24
Mean annual pre- and post-regulation interactions and mortalities by species in United States pelagic longline fisheries in the Atlantic, 1990–2007



Note: KR = Kemp's ridley; LH = loggerhead; GR = green; LB = leatherback; HB = hawksbill.
Source: Finkbeiner *et al.* (2011).

risk relative to species of special interest (i.e. seabirds, mammals, turtles and sharks). The leatherback turtle was the exception to this and was ranked as medium and low risk, respectively. This was attributed to its deeper dwelling habits, its lower age at maturity and its propensity to survive interactions (Kirby and Hobday, 2007).

FIGURE 25
Mean annual pre- and post-regulation interactions and mortalities by species in United States pelagic longline fisheries in the Pacific, 1990–2007



Note: KR = Kemp's ridley; LH = loggerhead; GR = green; LB = leatherback; HB = hawksbill.
Source: Finkbeiner *et al.* (2011).

Some WCPFC members follow guidance implemented in 2011 regarding the format of the Annual Reports-Part 1 with regard to reporting sea turtle interactions. These reports, which are in the public domain,[20] are to provide "observed annual estimated catches of species of special interest (seabird, turtle and marine mammals) by gear for the [National fleet], in the WCPFC Convention Area, for years [x–5] to [x–1] to the extent available". For example, Australia reported 13 interactions and two mortalities for 2011 (interaction rate of 0.0311 per 1 000 hooks) in its Pacific longline fisheries with most reported interactions involving green turtles. New Zealand reported a low rate of interactions for all turtles (0.006 turtles per 1 000 hooks) since 2001 with four interactions (three leatherback and one olive ridley) and no mortalities in 2011. Taiwan

[20] See https://wcpfc.int/meetings/8th-regular-session-scientific-committee for 2011 Annual Reports and www.wcpfc.int/meetings/9th-regular-session-scientific-committee for 2012 Annual Reports.

Province of China observers recorded 12 sea turtles caught in 2010 and 32 caught in 2011 (mortality data not provided). The United States of America reported 90 sea turtle interactions for 2011 (mortality data not provided), mostly leatherback and olive ridley, whereas Canada reported none. The annual reports from Japan, China, the Republic of Korea and Indonesia did not address sea turtle issues. As this sample of WCPFC Annual Reports-Part 1 illustrates, a number of members appear not to be following guidance on data reporting formats, and some of those that do provide data do not provide sufficient information to calculate species-specific interaction or mortality rates.

3.2.4 Indian Ocean

Of the world's most threatened sea turtle populations, 45 percent are found in the northern Indian Ocean (IUCN, 2011). Species–area combinations interacting with longline fisheries in the Indian Ocean tally to 17 according to Wallace *et al*. (2013), and represent all species except Kemp's ridley (Table 13). The highest interaction rates for the Indian Ocean (e.g. 0.04 interactions per 1 000 hooks for the loggerhead in the southwest) are low compared with other oceans, but the mortality rates are some of the highest recorded worldwide (Table 13).

TABLE 13
Interaction and mortality rates (not including post-release mortality) by species and area for longline fisheries in the Indian Ocean

Species	RMU	N	BPUE	MR
Flatback	SE Indian	ND	ND	ND
Green	NW Indian	2	ND	ND
	SE Indian	2	ND	ND
	NE Indian	4	0.037	0.03
	SW Indian	23	0.03	0.16
Hawksbill	NE Indian	ND	ND	ND
	NW Indian	2	ND	ND
	SW Indian	22	0.034	0.16
	SE Indian	5	0.004	0.03
Leatherback	NE Indian	ND	ND	ND
	SW Indian	8	0.014	0
Loggerhead	NW Indian	1	ND	ND
	SW Indian	25	0.04	0.16
	SE Indian	6	0.023	0
	NE Indian	4	0.009	0.29
Olive ridley	NE Indian	2	ND	ND
	W Indian	5	ND	0.05

Notes: RMU = regional management unit; N = sample size; BPUE = bycatch per unit effort (1 000 hooks); MR = mortality rate (scale of 0 to 1). Longline data presented by Wallace *et al*. (2013) represent a combination of pelagic, bottom, surface/drifting and "other" longline gear; however, no statistically significant differences were found between these gear types in BPUE or MR values overall.
Source: According to Wallace *et al*. (2013).

Some ad hoc information on sea turtle interactions for Indian Ocean longliners is available for recent observer coverage of Japanese and Spanish fishing effort and from older studies of South African domestic longline fleets (Ardill, Itano and Gillett, 2013). According to this source, Japanese longline fisheries observers in the latter half of 2012 recorded 14 sea turtle interactions (12 of which were with olive ridley turtles). No information is provided on the number of observed hooks; therefore, it is not possible to calculate an interaction rate. A second fishery's interaction data for the Indian Ocean is drawn from an experimental Spanish longline fishery that caught 25 turtles

in 532 000 hooks (0.05 turtles per 1 000 hooks). A South African domestic longline fleet was reported by observers as also having a turtle interaction rate of 0.05 turtles per 1 000 hooks from 2000 to 2003 with 36 percent leatherback, 31 percent loggerhead and the remainder comprised of green and olive ridley turtles (Ardill, Itano and Gillett, 2013). Taiwan Province of China reported interaction rates of zero per 1 000 hooks for southern temperate areas and 0.0112 per 1 000 hooks for tropical areas in 2004–08 (Huang, 2011).

A more comprehensive review of the situation in the IOTC with regard to sea turtle data was recently undertaken under the Indian Ocean South-east Asian Marine Turtle Memorandum of Understanding (IOSEA Secretariat, 2013). This review noted a recent decrease in longline effort by some fleets, which cited piracy and escalating costs as the reason for the decline. However, as a moderate increase in longline effort was reported by other fleets, there was not enough information to judge whether there would be a positive or negative impact on sea turtle populations. The review also assessed the sea turtle data submitted by IOTC members and found that several members were not implementing their observer programmes, or do not appear to include sea turtle monitoring as part of those programmes. Moreover, in many cases, observer coverage was found to be either low, or unrepresentative (or both) and thus meaningful extrapolation to the entire fishery would be problematic. As it was difficult to determine in some cases whether low reported interactions were a result of effective mitigation or merely poor implementation of monitoring protocols, the review concluded that "in general, the levels of marine turtle bycatch recorded in [member] reports should be considered with great caution" (IOSEA, 2013).

3.3 MANAGEMENT MEASURES AND THEIR EFFECTIVENESS

The t-RFMOs have had sea turtle conservation and management measures in place for up to ten years (e.g. ICCAT's first sea turtle measure was adopted in 2003) but few of these measures require specific actions by members' vessels to mitigate sea turtle interactions. Instead, most measures call for implementation of the FAO Guidelines to Reduce Sea Turtle Mortality in Fishing Operations, best practice with regard to safe handling and release, continued research into mitigation techniques, and provision of educational materials to fishers (Table 14). Most current measures also call for reporting of interaction data, but in some cases this data provision remains voluntary (or is not complied with), is not species-specific, does not require mortality rates to be reported, is not in the public domain, and/or has only recently been implemented. This situation limits the ability of the t-RFMOs to conduct impact assessments, and to consider what further mitigation measures may be required to protect threatened sea turtle populations.

Only one of the current t-RFMO management measures, WCPFC's CMM 2008-03, specifies requirements for longline fisheries that go beyond safe release equipment and techniques. This measure requires that shallow set longline fisheries (provisionally defined as those for which the majority of the hooks are set shallower than 100 m) use one of the following three mitigation measures: (i) large circle hooks with an offset of no more than 10 degrees; (ii) whole finfish bait; or (iii) another measure, mitigation plan or activity approved by the WCPFC Scientific Committee. While this measure is the most prescriptive of all the existing t-RFMO measures currently in effect, it has attracted criticism for the following reasons (Gilman, 2011):

TABLE 14
Currently active t-RFMO conservation and management measures pertaining to sea turtles

t-RFMO	CMM	Major provisions relevant to longline fisheries	A	B	C	D	E	F	G
CCSBT*	Oct 2011	Implement FAO Guidelines; comply with all ICCAT, IOTC and WCPFC measures; report data on interactions to the Commission which is authorized to exchange it with other t-RFMOs	X	X					
IATTC	Resolution 04-05 (Rev 2)	Prompt release unharmed; voluntarily provide bycatch data; training and equipment for safe release		X	X		X		
	Resolution 04-07	Voluntarily provide data on interactions; bycatch mitigation research; informational materials for fishers		X		X	X		
	Resolution 07-03	Implement the FAO Guidelines; require vessels to carry and use safe release equipment; continue mitigation research; Secretariat to further consider mitigation measures	X		X	X		X	
ICCAT	Recommendation 03-11	Encourage provision of data on interactions; encourage safe handling and release		X	X				
	Recommendation 05-08	Undertake trials of circle hooks; exchange information on safe release techniques			X	X			
	Recommendation 10-09	Starting in 2012 report annually on interactions; carry and use safe handling equipment; ICCAT to conduct impact assessment by 2013 and consider additional mitigation measures; members to report on implementation including FAO guidelines annually	X	X	X		X	X	
IOTC	Resolution 12/04	Implement the FAO Guidelines; report interactions and mortalities annually; carry and use safe handling equipment; identification cards for fishers; encourage use of finfish bait; report all interactions in logbooks; continue bycatch mitigation research; Commission to consider additional mitigation measures	X	X	X	X	X	X	X
WCPFC	CMM 2008-03	Implement the FAO Guidelines; report interactions annually; safe handling and release guidelines for fishers; carry and use safe handling equipment; shallow-set longlines must either (i) use large circle hooks, (ii) use whole finfish bait, or (iii) employ another measure approved by the Scientific Committee; all interactions to be recorded in logbooks and reported to Commission; continue mitigation research; Commission to consider additional mitigation measures.	X	X	X	X	X	X	X

* The CCSBT's convention area overlaps with those of ICCAT, IOTC and WCPFC in the higher latitudes of the Southern Hemisphere, where it has been suggested that sea turtle interactions are less problematic than in other areas (FAO, 2010).

Notes: A = implement FAO Guidelines; B = reporting of interactions; C = safe handling and release; D = conduct bycatch mitigation research; E = information for fishers; F = impact assessment and consideration of further mitigation; G = reference to specific mitigation measures.

- The measure only applies to shallow set longline fisheries even though interactions may occur in other longline fisheries.
- Each member may apply its own definition of what constitutes a shallow set longline fishery and a large circle hook.
- Requiring both large circle hooks and whole finfish bait would have a higher probability of mitigating sea turtle interactions than would either technique alone.
- There is no empirical basis for requiring circle hooks to have an offset of less than 10 degrees.

In addition, there is an exemption within the measure for shallow set longline fisheries that can demonstrate that there have been "minimal" interactions with sea turtles in the past three years and that observer coverage exceeds 10 percent in each of those three years. Minimal interactions have tentatively been defined as a mean of fewer than 0.019 sea turtle interactions (all species combined) per 1 000 hooks over the preceding three consecutive years (WCPFC, 2009). The WCPFC has yet to undertake any evaluation of whether members are complying with the measure or whether it is working effectively.

In contrast to the loosely prescriptive approach to mitigation adopted by the WCPFC in 2008, both ICCAT and the IOTC have committed to undertaking sea turtle assessments before considering what further mitigation measures may be required. However, as discussed above for both t-RFMOs, even with explicit data provision requirements for members in place, the data quantity and quality is likely to create considerable uncertainties in the assessments. Nevertheless, both ICCAT and the IOTC have mandates to discuss further measures to mitigate sea turtle interactions in 2013.

Another approach has been adopted by the IATTC. This t-RFMO has actively participated in a long-standing, cooperative, "bottom-up" programme supported by the World Wildlife Fund and the United States government that involves the voluntary testing of circle hook performance aboard fishing vessels during regular fishing operations. A focus of the programme has been to compare and demonstrate interaction and mortality rates for circle and J hooks and to allow fishers to confront, and often solve for themselves, many of the practical obstacles to circle hook deployment (Serafy *et al*., 2012). The programme began in 2003 and by 2008 was active in Colombia, Costa Rica, Ecuador, El Salvador, Guatemala, Mexico, Nicaragua, Panama and Peru, and under development in Chile (IATTC, 2012a). Trials of circle hooks in the IATTC convention area have also been reported by Japan, the Republic of Korea, the United States of America, Spain and Taiwan Province of China (FAO, 2010). A summary of this research by the consortium supporting it concludes that circle hooks reduced sea turtle hooking rates in most, but not all, of the fisheries and countries analysed and thus advocates fishery/region-specific management approaches (Andraka *et al*., 2013).

Members of t-RFMOs and other States are free to enact their own sea turtle mitigation requirements for national waters and nationally flagged vessels. In some cases, these national requirements may go beyond what is required by the t-RFMOs. For example, in the United States of America, the Hawaii longline swordfish fishery regulations implemented in 2001 in response to high interaction rates with leatherback and loggerhead sea turtles in the shallow set swordfish led to a closure of this fishery for more than three years (Walsh, Bigelow and Sender, 2009). (Coincidentally, the California-based longline fishery was also closed just as the Hawaii fishery re-opened, prompting many California-based longline vessels to relocate to Hawaii [NOAA, 2012a]). The Hawaii shallow-set longline fishery is now required to use large circle hooks (size 18/0) and fish bait, is restricted in its annual effort, has an annual cap on turtle interactions, and requires 100 percent observer coverage (Gilman *et al*., 2006a). It was reported in 2007 that after the fishery re-opened in 2004 that target catch rates

increased by 16 percent while leatherback and loggerhead turtle interactions decreased by 83 percent and 90 percent, respectively (Gilman *et al*., 2007b; Curran and Bigelow, 2011). In 2006, the fishery was closed from March onward owing to loggerhead interactions, remained open for all of 2007–2010, and was closed again for the year in November 2011 owing to leatherback interactions (NOAA, 2012a).

At about the same time, an area of the North Atlantic that included the Grand Banks was partially closed to the United States pelagic longline fleet in 2000, and fully closed from 2001–04 (Gilman *et al*., 2006a). Subsequently, the United States of America implemented mandatory sea turtle bycatch reduction measures for the United States Atlantic pelagic longline fishery as a whole, including attendance at sea turtle release and disentanglement workshops, bait specifications, use of circle hooks (size of hook depending on fishing locale), and possession and use of sea turtle handling and release gear on board all pelagic longline vessels (NOAA, 2012b). There are also time–area closures in various locations (Gardner *et al*., 2008, Garrison and Stokes, 2012). Based on observer coverage of 10.9 percent by hooks and 10.1 percent by sets, total interactions for the fishery were estimated and found to be well below historical levels for both loggerhead and leatherbacks, and below pre-regulation levels for leatherbacks only (Garrison and Stokes, 2012).

In the WCPFC, Australia formulated a sea turtle mitigation plan for its Eastern Tuna and Billfish Fishery that sets "trigger level" interaction rates of ≤ 0.0048 turtles per 1 000 hooks for each turtle species or 0.0172 turtles per 1 000 hooks overall (DAFF, 2009). Australia reported in 2013 that the trigger levels had been exceeded for the third year in a row. As a consequence, the Australian Fisheries Management Authority's (AFMA) revoked the mitigation plan as of 1 March 2013 and now requires that all shallow-set longliners in the Eastern Tuna and Billfish Fishery use large circle hooks consistent with the WCPFC measure (CMM 2008-03; Patterson, Sahlqvist and Larcombe, 2013).

3.4 REVIEW OF MITIGATION METHODS

The preceding discussion of management measures for sea turtles has highlighted that there are few available data that can be used to evaluate their effectiveness. This is because most compliance data are either not collected, not standardized, not reported or not publicly available. Performance data that are available often represent the combined effects of a number of mitigation components that are together intended to reduce interactions below levels that would trigger additional management measures (e.g. in the Hawaii pelagic longline fishery or Australia's Eastern Tuna and Billfish Fishery). As a result, it can be difficult to attribute successful mitigation to individual mitigation components.

As a result of these issues, information on the effectiveness of individual sea turtle mitigation measures is most commonly available in the scientific literature. While some published studies have had the advantage of designing an experiment to isolate and test the effects of a specific mitigation measure, such studies may not be able to replicate commercial fishing operations. In contrast, research based on analysis of actual commercial fishing operations may find it difficult to control for confounding factors, especially when multiple fleets and areas are involved (see Andraka *et al*., 2013). For this reason, the effectiveness of many of the techniques discussed below is not categorical and will often vary in degree under different circumstances. The following sections discuss the current state of knowledge regarding the effects of hook type, bait type, depth, closures, fleet communication, deterrents, and safe handling and release techniques in mitigating impacts on sea turtle populations.

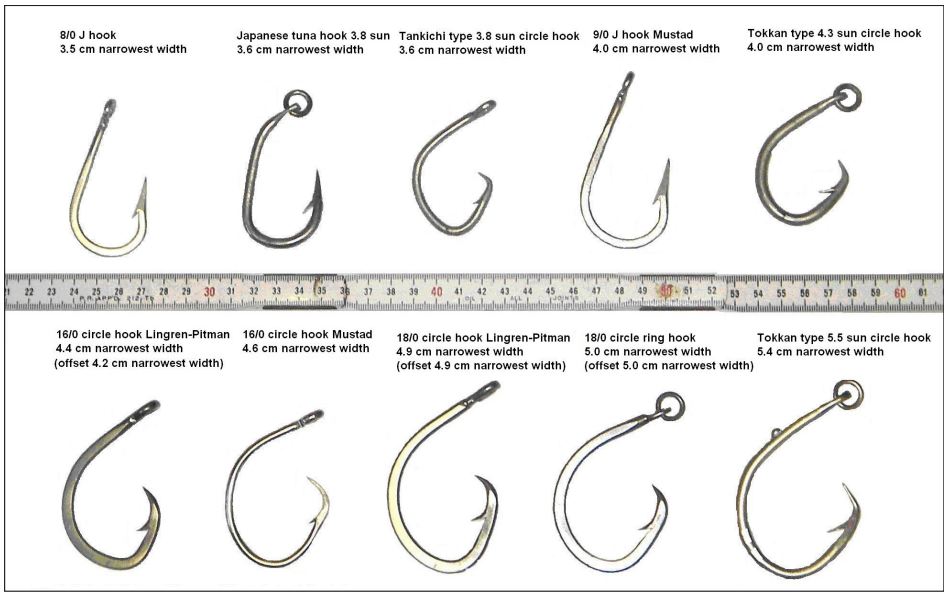
3.4.1 Hook type

Current understanding of the effects of circle hooks on sea turtle interaction and mortality rates has benefited greatly from a 2011 international symposium involving many of the most active researchers in the field. A summary of the proceedings was able to synthesize the findings of more than 80 presentations and draw a number of important conclusions not only for sea turtles but for sharks and target species as well (Serafy *et al*., 2012). It is noted that with regard to sea turtles, the symposium focused on interactions involving ingestion of hooks, rather than foul-hooking, which mainly affects leatherback turtles (see Section 3.1.2). Therefore, while the following discussion focuses on hook width (e.g. gape or minimum width) as it relates to ingestion impacts, hook shape (e.g. circle versus J hooks) rather than hook width may be the more important factor in mitigating foul-hooking impacts for leatherback sea turtles (Gilman, 2011).

With regard to sea turtles, the symposium provided further evidence that large circle hooks (e.g. size 18/0, Figure 26) reduce hooking rates and the probability of deep hook ingestion.[21] The latter factor is expected to benefit sea turtles mainly if there is proper training and equipment available for the safe removal of hooks. Both effects had been found in several prior studies (e.g. Watson *et al*., 2005; Gilman *et al*., 2006a; Read, 2007; Stokes *et al*., 2011) and were further confirmed by research presented at the symposium from Colombia, Ecuador, Guatemala, the Pacific coast of Nicaragua, Panama, and Portuguese and United States longline fleets fishing in the Atlantic (Serafy *et al*., 2012). However, contrary results were reported for small (size 13/0) circle hook tests off Italy, which found no difference in overall interaction rates (although there was a significant difference in interaction rates for large juveniles), and no reduction in deep hook ingestion between circle and J hooks (Cambiè *et al*., 2012). A previous study in the Canadian longline fishery also found no significant reduction in interaction rates or the probability of gut hooking in loggerhead turtles when using smaller (size 16/0) circle hooks versus J hooks (Carruthers, Schneider and Neilson, 2009).

FIGURE 26

A number of types of circle and J hooks arranged in ascending order by width at the narrowest point



Note: Scale in centimetres.
Photo credit: E. Gilman.

[21] A comprehensive catalogue of hook types is available at www.iattc.org/downloads/hooks-anzuelos-catalogue.pdf

While these results suggest that it may be the size of the hook that is most important, there are likely to be several other hook characteristics that determine interaction impacts. Circle hooks with larger offsets, i.e. angular deviation of the point from the centre line of the shank, may be more likely to impact sea turtles than those with smaller offsets even if size is held constant (Serafy *et al*., 2012). This may be because the offset hooks, particularly if they are J hooks, are more likely to lodge in soft tissue if swallowed and pulled back out (Stokes, Epperly and McCarthy, 2012). However, no significant differences were found in sea turtle interaction rates or anatomical hooking locations between size 14/0 circle hooks with and without 10° offsets in a study off Costa Rica (Swimmer *et al*., 2010). Differences between hooks with and without 10° offsets were also not found to be significant in a study off Hawaii (Curran and Bigelow, 2011). In that side-by-side comparison of large circle hooks with and without offsets (size 18/0), Japanese style tuna hooks (size 3.6 sun) and J hooks (size 9/0), too few turtles were caught to draw conclusions about the effects of hook size and shape on catch rates of turtles *per se*. However, it was hypothesized that minimum width (Figure 27) rather than gape or straight total length, is the most important driver of observed lower catch rates for 14 of 18 non-turtle and non-tuna species on circle hooks (minimum width of 4.9 cm) as compared with tuna hooks (minimum width of 3.1 cm; J hooks are intermediate with a minimum width of 3.9 cm).

FIGURE 27
Hook terminology used in Curran and Bigelow (2011) showing the key parameter minimum width

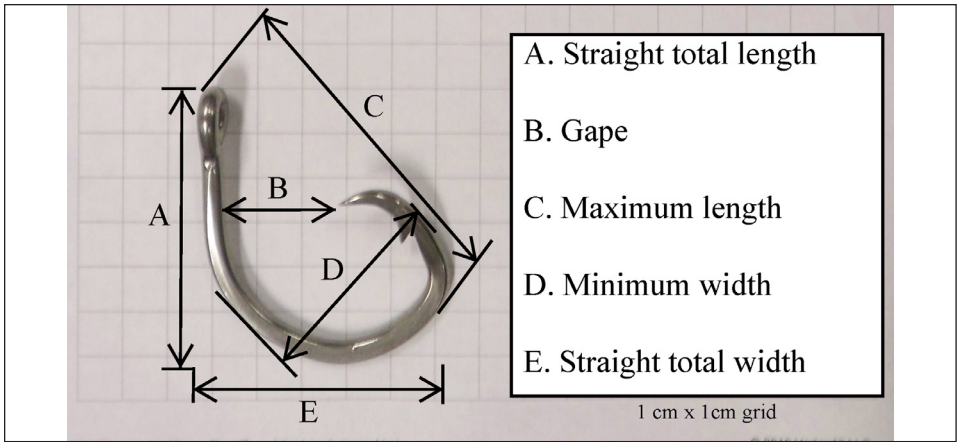


Photo credit: D. Curran.

Some studies have also explored whether appendages added to smaller hooks to increase their dimensions would be effective. One study off Costa Rica found that hooks with appendages caught fewer turtles but reduced target species catch and had no effect on the probability of deep hooking (Swimmer *et al*., 2011). Another study also found that hook appendages did not substantially decrease deep ingestion rates (Hataway and Stokes, 2012). This complex situation with regard to hook morphometrics lends further weight to the recommendations of Serafy *et al*. (2012) for better standardization of hook description terminology for both scientific and compliance purposes.

It is also important to consider that what is considered a large circle hook will vary with the size and morphology of the organism with which it is potentially interacting.

In circle hook trials off Brazil specifically aimed at testing sea turtle interactions, an observed higher catch rate of larger loggerheads was attributed to the possibility that the greater width of circle hooks prevented ingestion in smaller turtles (Sales *et al*., 2010). Curran and Bigelow (2011) also found that large circle hooks significantly reduced catch rates for smaller-mouthed (non-turtle) species in Hawaii, but on the United States east coast turtle size was not a significant factor in determining anatomical hooking location (Stokes *et al*., 2012). This could be explained by the fact that turtles of straight carapace length greater than 65 cm have been shown in the laboratory to be capable of swallowing hooks of up to 18/0 size (Stokes, Epperly and McCarthy, 2011). Thus, once a certain size threshold is reached, hook size may no longer be an important factor. The international circle hook symposium noted the importance of further research into synergies between hook types, bait types (see next section), mouth morphology and feeding behaviour as a means of gaining deeper insights (Serafy *et al*., 2012).

3.4.2 Bait type

As specified in some of the management measures currently in force (see Section 3.3), finfish bait is required either in conjunction with circle hooks or as an alternative mitigation technique. A number of studies of bait and hook type combinations have been undertaken to explore whether shifting from the traditional use of squid bait to mackerel bait would further reduce sea turtle interactions (e.g. Watson *et al*., 2005). A comprehensive summary of several of the early studies in the Atlantic concluded that although J hooks showed more turtle interactions than circle hooks, this difference was reduced when mackerel bait was used on the J hooks. Moreover, circle hooks performed better in terms of minimizing turtle interactions when used with mackerel bait rather than squid bait. Bait type appeared to be more of a factor for leatherbacks whereas loggerheads benefited more from a change from J hooks to circle hooks (Read, 2007). Studies of the Portuguese fleet in the equatorial Atlantic found that when mackerel bait was used instead of squid, the probability of interactions with olive ridley turtles declined by 56 percent (Santos *et al*., 2012). In the Pacific, interaction rates for loggerhead turtles were 75 percent less when using mackerel bait than when using squid bait (Yokota, Kiyota and Okamura, 2009).

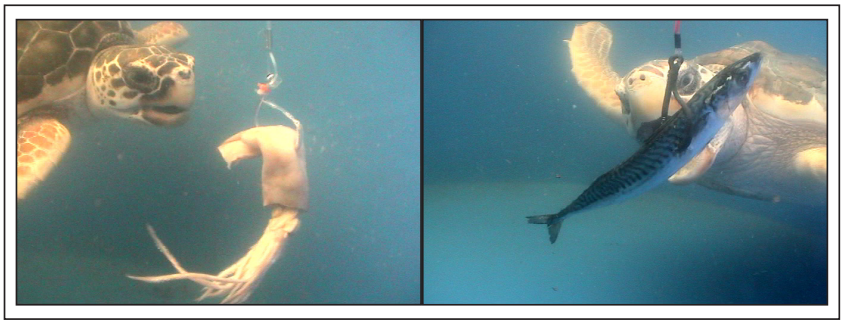
Previous studies in the Pacific had drawn a similar conclusion regarding lower catch rates with mackerel bait, hypothesizing that the rubbery texture of squid made it likely that turtles would attempt to swallow it whole, thus swallowing the hook as well (Kiyota *et al*., 2004, Plate 3). Further support for this theory is provided by laboratory studies that found that turtles are four times as likely to attempt to swallow hooks baited with squid than hooks baited with mackerel (Stokes *et al*., 2011). The evidence is mixed, however, as Watson *et al*. (2005) found no difference in anatomical hooking location between bait types, and Santos *et al*. (2012) found to the contrary that the probability of deep hooking increased when mackerel bait was used. Laboratory research also suggests that sea turtles may prefer squid to finfish owing to natural chemical attractants present in squid (Piovano, Farcomeni and Giacoma, 2012b).

Some studies have also investigated whether dyeing the bait a dark blue colour can also help to reduce interaction rates, but this technique has not proved reliable. Laboratory experiments with loggerheads suggest that each individual has its own consistent colour preferences (Piovano, Farcomeni and Giacoma, 2012a). Swimmer *et al*. (2005) also found that colour preferences demonstrated in a laboratory setting, i.e. avoidance of blue-dyed bait by loggerheads and Kemp's ridley turtles, were not expressed in field trials. Experimental fishing in the Pacific also failed to find any significant effect of dyed bait on turtle interaction rates (Yokota, Kiyota and Okamura, 2009).

A final factor that appears not to have received much research attention thus far is the placement of bait on hooks (Plate 4). A laboratory study compared turtles' attempts to swallow "single-baited" (hooked once) and "threaded" (hooked twice) squid and sardines and found that, regardless of species, threaded baits were 2.5 times more likely to be swallowed – probably because they are more difficult to strip from the hook (Stokes *et al*., 2011). Fishers in the United States Atlantic are said to choose whether or not to thread bait based on the importance of bait retention versus the speed of hook baiting and other factors (Stokes *et al*., 2011). Given the potential relevance to sea turtle mitigation, it is surprising that not more is known about the use of bait threading in other fisheries. Highlighting the necessity of taking fishers' bait preferences into account, some studies have noted that changing hook and bait combinations may be

PLATE 3

There is some evidence that sea turtles bite off pieces of finfish bait and thus avoid longline hooks, whereas owing its rubbery texture, sea turtles tend to swallow squid bait whole and thus also swallow the hook.



Courtesy of NOAA Fisheries, Southeast Fisheries Science Center.

resisted by fishers if they result in lower catches of target species (Santos *et al*., 2012) or if they feel that larger hooks require larger, and presumably more expensive, baits (Villagran *et al*., 2012).

3.4.3 Other gear and operational controls

In addition to the two types of mitigation measures discussed above, i.e. modified hook design or modified baiting, additional gear or operational modifications such as changing the depth at which the gear fishes and time of operations have also been trialled. Changing the location of the fishing activity is discussed in the following section.

There are three components to avoiding sea turtle interactions by depth: (i) the depth preferences of the turtles must be known; (ii) the technology for setting the gear at a particular depth must be available; and (iii) the fishery must remain economically viable even if the depth of fishing is altered to avoid sea turtles. Knowledge regarding the first component has improved greatly with advances in telemetry systems and their use in studies of several, but not all, sea turtle species of conservation concern (Godley *et al*., 2008; Gaos *et al*., 2012). Oceanographic features such as temperature, fronts, lunar cycles sea-bed depth, chlorophyll *a* and magnetic forces have been explored for some species in some areas (see Section 3.2.1). Despite high inter- and intra- (i.e. between population) species variability, and even variability among individuals (Gaos *et al*., 2012), it is generally accepted that most turtles spend the majority of their time at depths shallower than 40 m, and that even when diving, most species remain shallower than 100 m (Polovina *et al*., 2003; FAO, 2010). Many longline operations targeting tuna will fish deeper than 100 m, but avoiding shallow setting is not likely to be feasible for longline operations targeting swordfish. Therefore, for those operations that are

PLATE 4

Illustration of single-baiting (left) and threaded-baiting (right) techniques for finfish. Single-baited finfish are hooked in the eye. Threaded baits are additionally hooked distal to the initial point of entry determined by the hook width. Squid are attached similarly except that the single-baiting attachment point is the mantle.

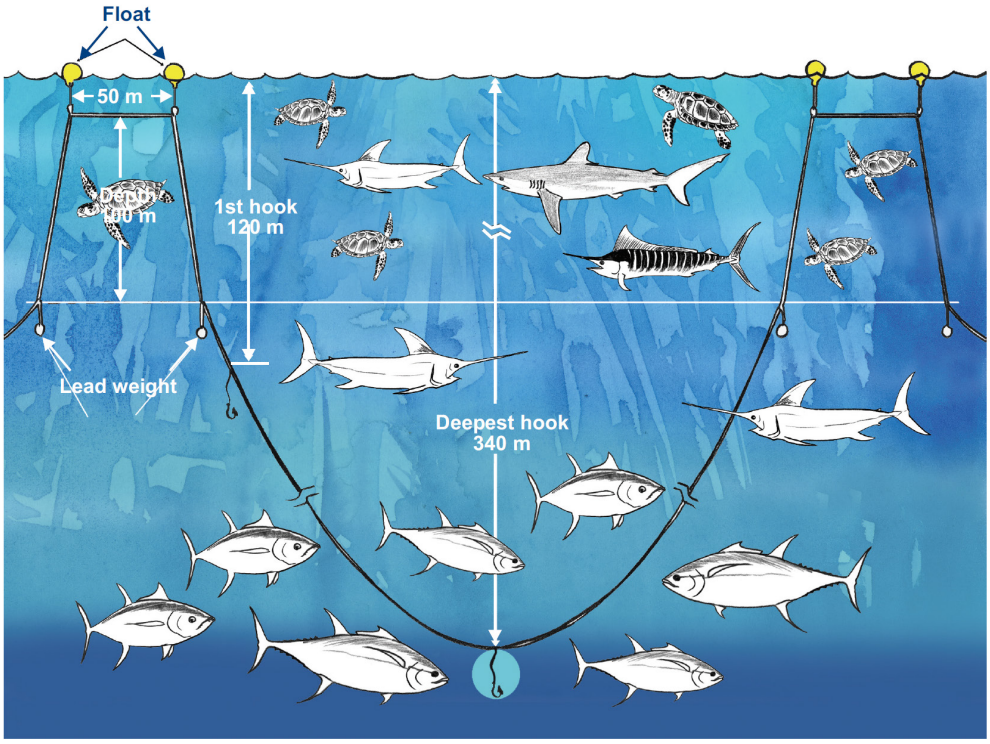
Source: Text from Stokes et al. (2011).
Courtesy of NOAA Fisheries, Southeast Fisheries Science Center.

willing to set deeper to avoid sea turtles, the key issue will be technically how best to accomplish this.

Several options are presented in FAO (2010). One involves extending the length of the branch lines that are deployed closest to the floats (i.e. at the two [shallow] ends of the catenary curve that forms between floats) or widening the gap between the floats and their nearest branch lines (Figure 28). Of these alternatives, the latter is recommended (FAO, 2010), perhaps owing to the increased likelihood of entanglement when branch lines are lengthened. A more advanced option was tested in the Hawaii longline fishery in 2006 (Beverly *et al*., 2009). This design involves suspending the main line below the floats with 3 kg weights so that the depth of the first hook is increased from 44 m to 104 m. The trials indicated that catch rates for target tuna species were not affected but catch rates for non-target species were reduced. No sea turtles were caught in either type of set but the technique was confirmed to be effective in moving the gear out of sea turtle habitat. The cost of the gear modification was minimal, but it was noted that deployment and haulback were slower by 30 and 120 minutes respectively (Beverly *et al*., 2009). Another configuration involving one or two buoys deployed between each main float designed to lift and flatten the catenary curve of the main line was tested off Japan. This gear modification was found to reduce the depth difference between branch lines from 55.1 m in the standard configuration to 5 m with 2 buoys and 26 m with 1 buoy (Shiode *et al*., 2005).

A number of older and often unpublished studies explored the effects of soak duration and time of hauling on sea turtle interaction rates. However, owing to a lack of definitive results, as well as perhaps the negative ramifications for target species, these operational modifications have not been subject to further research or regulatory implementation. For example, Gilman *et al*. (2006a) review two studies, one that found higher interaction rates at night (but was confounded with shallow setting) and another that found an effect of total (not daylight) soak time on loggerheads only. While there may be further potential for mitigation through manipulation of soak duration and time of hauling (Gilman, 2011), issues of experiment design and inter- and intra-specific differences have led to conflicting results thus far and no clear research agenda.

FIGURE 28
Configuration of weighted gear with 20 hooks per basket and a target depth for the shallowest hook of 120 m



Note: All baited hooks are below the 100 m line.
Sources: FAO (2010), after Beverly and Robinson (2004).

3.4.4 Habitat avoidance

Time/area closures of longline fisheries in the United States of America represent a form of habitat-avoidance-based mitigation measure. Such measures may follow a prescriptive approach where certain fishing grounds are closed based on historical, current or predicted occurrence of sea turtles. This approach was applied by the United States of America when it temporarily closed the Northeast Distant statistical area of the Atlantic longline fishery south and east of Newfoundland in the early 2000s (Gardner *et al.*, 2008). More recently United States closed area regulations have been refined and complemented with gear-based mitigation measures required to be used when fishing in certain areas (NOAA, 2004; Garrison and Stokes, 2012).

Another approach, applied in the United States of America to the Hawaii longline fishery involves setting a cap on sea turtle interactions and closing the fishery if the cap is exceeded. Under this approach, fishers are responsible for avoiding interactions using whatever combination of methods they consider most effective. In response to closure of the shallow-set fishery for most of 2006 (see Section 3.3), the National Oceanic and Atmospheric Administration (NOAA) developed a real-time "Turtle Watch" tool to distribute to fishers as a means of assisting them to avoid sea turtle interactions (Figure 29). This tool was based exclusively on SST data and the general rule that sets should remain south of the 18.5 °C isotherm (i.e. in warmer waters). When the fishery re-opened in the first quarter of 2007, the fishery did not remain south of the recommended isotherm; instead it moved north with increased effort and lower sea turtle interaction rates. However, the majority of interactions in 2007 occurred in areas discouraged by the Turtle Watch tool, which lends credence to the

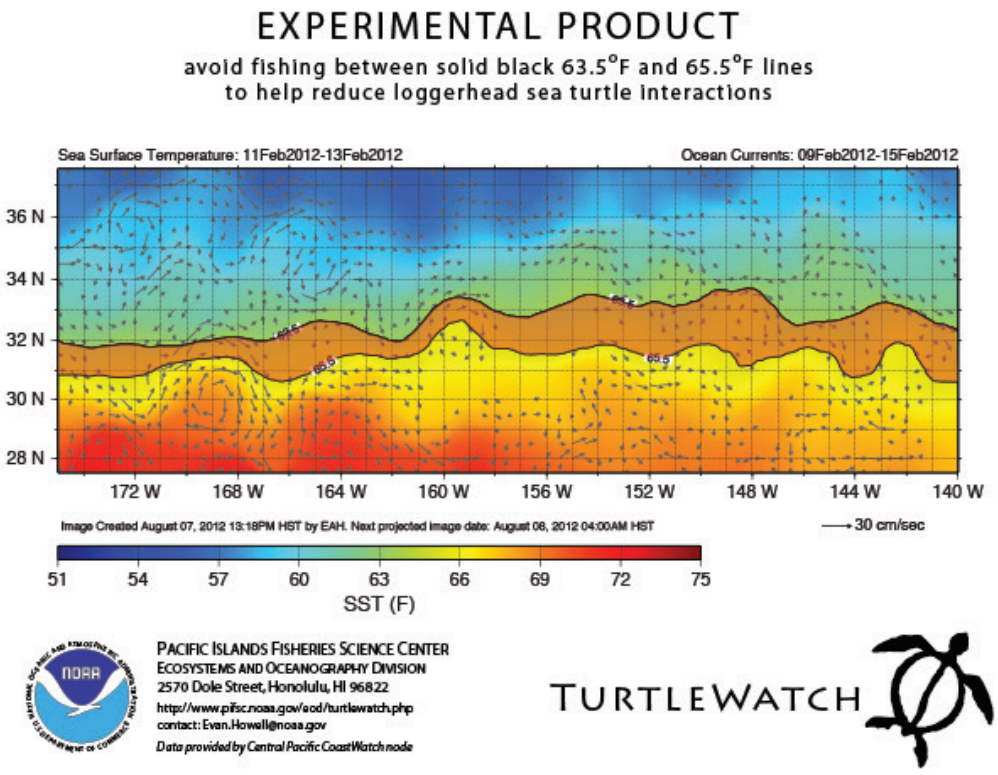
possibility that the requirement to deploy circle hooks with finfish bait was responsible for the reduced catch rates (Howell *et al*., 2008).

Research comparing the Hawaii shallow-set longline fishery before and after implementation of sea turtle mitigation regulations and prior to the closure in 2006 found that 23 percent of the interactions involved ≥ 1 sea turtle. This clustering effect suggested that if a vessel shifted position, e.g. away from an oceanic front or gyre, seamount or shelf break, further interactions could be avoided. Even greater reductions of interaction rates would be expected if fishers exchanged this information freely throughout the fleet (Gilman *et al*., 2007b).

Although a fleet communication system to report real-time bycatch hotspots was trialled in the Hawaii pelagic longline fishery (Gilman *et al*., 2006a), it has not become a major factor determining sea turtle interaction rates. However, an earlier fleet communication system was implemented in the North Atlantic swordfish fishery from 2001 to 2003, primarily to reduce loggerhead and leatherback sea turtle interactions. At that time, although the fishery was still using J hooks, sea turtle interaction rates were 50 percent lower than previously. Although the system has formally ceased to operate, it is believed to continue to operate informally as an effective means of mitigation (Gilman, Dalzell and Martin, 2006; FAO, 2010). Several preconditions may exist for successful use of fleet communication systems including: (i) strong economic incentives to reduce interactions; (ii) infrequent interactions; (iii) adequate observer coverage; and (iv) the organization of large fleets into fishery associations (Gilman, Dalzell and Martin, 2006b).

FIGURE 29

Example of the "Turtle Watch" tool developed by NOAA and provided to fishers as guidance as a means for avoiding interactions with sea turtles by remaining south of the 18.5 °C isotherm



Note: Black line on graph = 18.5 °C isotherm.
Source: Howell et al. (2008).

3.4.5 Deterrents

As knowledge of marine organismal sensory systems improves, the opportunities for exploiting differences between target and non-target species attraction to baited hooks expands. A review of this topic focused on longline fisheries, large pelagic fishes and sea turtles discussed visual, auditory and chemical detection cues (Southwood *et al*., 2008). Auditory deterrents were dismissed for several reasons including similar frequency detection ranges in target species and sea turtles, the likelihood of habituation and the need to avoid increasing anthropogenic sound levels in the marine environment. Chemical deterrents were considered to have potential but chemosensory responses are believed to be of secondary importance when compared with visual ones. In addition, even when preferred squid baits were treated with supposed deterrents, this did not affect their feeding behaviour. Visual cues were considered to have the greatest potential particularly owing to differences between billfishes and turtles in terms of the visible light spectrum. Billfishes' peak sensitivities are toward the violet-blue end of the spectrum and they cannot perceive ultraviolent light, whereas turtles' peak sensitivities are shifted toward the green range of the spectrum and they can perceive a wide range of ultraviolent frequencies. While these differences are probably not great enough to avoid inadvertent attraction of turtles to lightsticks deployed to attract billfishes (see Wang *et al*., 2007), aversion to some light frequencies has been documented for hatchling loggerheads. Another possibility would be to use ultraviolet light frequencies to lure turtles away from longline hooks (Southwood *et al*., 2008).

In contrast to research that suggests that sea turtles are attracted to longline hooks by lightsticks designed to lure billfishes, one study found no attractive effect, and in some cases a repellent effect, of lightsticks on juvenile leatherback turtles in the laboratory. However, these results contrast with those of loggerheads, which show attraction to lightsticks in the laboratory (Wang *et al*., 2007) but which in the field are mostly caught in daylight sets (Gless, Salmon and Wyneken, 2008). Testing of photosensitivity in leatherback turtles on nesting beaches has provided further evidence for species-specific differences among sea turtles (Crognale *et al*., 2008). Leatherbacks were found to be better adapted to low light levels, as would be expected from their known deep-diving habits. As the spectra of light perceived by leatherback turtles is similar to that of swordfish, coloured lightsticks designed to attract swordfish would also probably attract leatherbacks. However, leatherbacks have a considerably lower flicker sensitivity, probably owing to feeding strategies, therefore deploying flickering lightsticks could maintain attractiveness to swordfish while reducing interactions with leatherbacks (Crognale *et al*., 2008). Research on sensory deterrents in sea turtles, similar to that reported in Swimmer and Brill (2006) and Wang, Fisler and Swimmer (2010) on *inter alia* shark shapes and net illumination, is continuing in a number of trials in Central and South America focused on coastal net fisheries (Y. Swimmer, personal communication, August 2013).

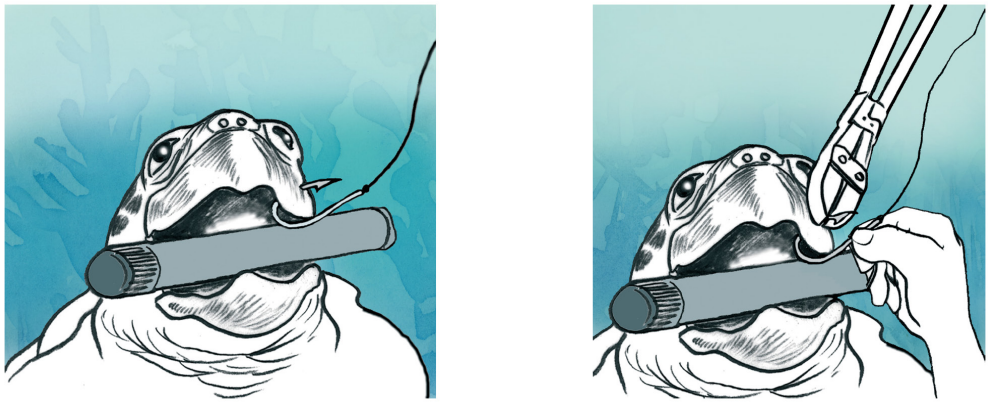
3.4.6 Release using line cutters / de-hookers

While some of the mitigation methods described in the preceding sections work by avoiding hooking of sea turtles, others are premised upon both avoiding hooking and preventing deep hook ingestion, i.e. hook and bait types. In the latter cases, sea turtles still become hooked but it is simpler and safer to remove the hook and thus prevent serious injury to the turtle. The overall effectiveness of these techniques will thus depend both on the probability that safe release techniques are attempted and on the success rate of these techniques in avoiding immediate or post-release mortality.

Best-practice sea turtle safe release procedures have primarily developed in the United States of America and are required in both Atlantic and Pacific pelagic longline fisheries (NMFS, 2008). These procedures cover handling, de-hooking (if possible), resuscitation and release, and include a list of line and bolt cutters, de-hookers and

mouth gags that must be carried on board. In brief, these procedures state that turtles up to 90 cm in length should be boated and hooks removed if this can be accomplished without causing further harm to the turtle (Figure 30). A number of sources consider that it is more difficult to safely remove a circle hook and that in a considerable number of cases the best option is to leave the hook in place (Parga, 2012). If it is necessary to leave the hook where it is, branch lines should be cut as close to the turtle as possible. Turtles brought on board should be moist and shaded while they are resuscitated for up to 24 hours by allowing their lungs to drain until they appear active again (FAO, 2010).

FIGURE 30
Recommended procedure for removing a hook whose barb is visible



Note: If the barb is not visible, remove as much of the branch line as possible and then cut it as close to the turtle as practical.
Source: FAO (2010).

Outside of the United States of America, all t-RFMOs and some countries have specifications for safe handling and release of sea turtles, but data on implementation and success rates for these procedures are not available. In particular, fishers' behaviour with respect to their desire to retrieve the hook is likely to influence injury rates considerably (Parga, 2012). Limited data and a number of assumptions from longline fisheries in the early 2000s suggested that post-release mortality rates are between 19 and 40 percent (Gilman *et al.*, 2006a; Parga, 2012). Although the frequency of deep hooking and severity of handling and de-hooking injuries to turtles may have decreased since that time, these rates are both high enough and uncertain enough to justify an emphasis on regulating fisheries on the basis of interactions rather than on estimates of mortalities *per se*.

3.5 CONCLUSIONS REGARDING SEA TURTLE INTERACTIONS

The world's seven sea turtles species suffer detrimental impacts from egg and meat consumption, habitat degradation and marine pollution, but it is widely recognized that interaction with fishing gear is one of the most serious threats to their continued survival. At a global scale, longlines have similar sea turtle interaction rates as trawls and set nets, but show relatively lower mortality and are expected to have a lower impact overall. Nevertheless, previous global studies have extrapolated interaction rates as high as hundreds of thousands per year, and more recent minimum estimates suggest longline interactions with about 56 000 sea turtles between 1990–2008.

Moreover, longlines can have significant population-level impacts in particular regions on specific sea turtle populations, especially where high densities of turtles and fishing gear overlap in time and space.

Some aspects of sea turtle interactions with longline fisheries are determined by species-specific biological and behavioural characteristics. For example, owing to its lack of a hard shell, the leatherback sea turtle is more likely to be entangled or foul hooked than other species, and this leads to a higher probability of safe release for these species assuming it avoids asphyxiation before and during haulback. Also, it is likely that each species (and life stage) has its own temperature and depth preferences, and as these become better known, further opportunities for avoiding interactions may emerge. This is similarly the case for diurnal behaviour patterns, visual and chemical (taste) preferences, and physical feeding mechanics.

Sea turtle species have been listed on CITES for several decades, and there are three regional treaties promoting intergovernmental cooperation on their conservation. All but one of the seven species is listed on the IUCN Red List in a "threatened" category (critically endangered, endangered or vulnerable). However, several of these listings require updating, and the status of regional populations may be better or worse than the listing indicates. Large-scale studies of risk and impact mapping have recently determined that four populations (leatherbacks in the southwest Indian Ocean and southwest Atlantic, loggerheads in the South Pacific, and green turtles in the Mediterranean) are both at high risk and are sustaining high impacts from longline fishing. With regard to interaction rates, loggerheads in the Atlantic have the highest levels of any species in any ocean. These rates have been reduced with the implementation of mitigation measures in the United States Atlantic longline fishery and mortalities decreased by 80 percent from 1990 to 2007. However, Atlantic loggerheads still showed an overall declining population trend during this period. Interaction and mortality rates from longline fisheries are lower in the Pacific than in the Atlantic with the exception of the South Pacific loggerhead and green turtles. United States government-mandated mitigation measures for Pacific longline fisheries have reduced interactions considerably, and they now total fewer than 50 per year for all species combined. The Indian Ocean also has relatively low interaction rates, but some of the highest mortality rates in the world. Although impacts on turtles may have decreased with reduced longlining effort in the Indian Ocean owing to piracy, a recent review by the regional sea turtle conservation secretariat (IOSEA) found little certainty in the current understanding of impact levels and their effects on turtle populations.

Although t-RFMOs have had sea turtle conservation and management measures in place for several years, these measures are often not specific in their requirements and there are few data available to evaluate their effectiveness. The WCPFC is the only t-RFMO with detailed requirements for circle hook or finfish bait use (adopted in 2008), but as yet there has been no evaluation of compliance. Despite the fact that there are substantial questions regarding whether the available data are adequate to support the objectives, both the ICCAT and the IOTC had sea turtle assessments planned for 2013. At the national level, the United States of America and Australia have regulatory caps on the number of interactions that are allowed to occur before the longline fishery is closed. In both cases, further mitigation measures have been triggered when caps are exceeded.

A large number of techniques have been tested as a means of mitigating sea turtle impacts. However, it is often difficult to isolate the effectiveness of a single technique and/or to interpret results, which vary from fishery to fishery. The use of circle hooks appears to be effective in many fisheries but this effectiveness may be limited to those circle hooks that are large, particularly with regard to their minimum width, relative to the size of turtles that might take the hook. It is not always clear whether circle hooks reduce the number of turtles that swallow the hook, the number of turtles that

become deep hooked, or both. The use of finfish bait in conjunction with circle hooks usually shows a clear effect in reducing hooking rates, reducing deep hooking or both. A considerable amount of research has focused on predicting or monitoring the ocean environment to avoid fishing in areas of prime sea turtle habitat. The overall result has been, however, that inter- and intra-species variation, as well as the interaction of complex oceanographic features, are not yet well enough understood to rely on these methods exclusively. Similarly, further research on deterrents, such as colours or chemicals, and their effectiveness across species and between life stages will be required. In the case of setting gear deeper than turtles' preferred habitat, while effective as mitigation, this technique may not be economically feasible for fisheries whose target species inhabit the same depth range as turtles (e.g. swordfish). When mitigation takes the form of increased rates of mouth hooking, for this to have a positive effect on mortality rates, fishers need to be prepared to practise safe release techniques. While these techniques are fairly well understood and widely disseminated, their actual rates of implementation in most fisheries remain unknown.

The most successful longline fishery mitigation techniques will be those that are both efficient in terms of avoiding sea turtle interactions, and economically viable in terms of the degree of disruption caused to fishing operations for target species (FAO, 2010). It may be that the best way of determining which techniques are most successful is not to mandate particular methods for particular fisheries but rather to set maximum interaction rates and allow fishers to find the optimal approach. In such cases, the emphasis should shift from gear research to operational monitoring, and to refining estimates of post-release mortality.

4. Seabirds

4.1 OVERVIEW OF SEABIRD INTERACTIONS WITH LONGLINE GEAR

4.1.1 Definitions and concepts

Seabirds are vulnerable to mortality in longline, trawl and gillnet (including driftnet) fisheries. Among longline fisheries, pelagic longlining has been identified as a major threat to albatross species in the family Diomedeidae and petrel/shearwater species in the family Procellariidae (Croxall *et al*., 2012). Seventeen of the 22 species of albatross are threatened with extinction (BirdLife International, 2013), and the main cause of their population decline has been reported to be longline-induced mortality (e.g. Brothers, 1991; Croxall *et al*., 1998; Gales, Brothers and Reid, 1998; Klaer and Polacheck, 1998; Inchausti and Weimerskirch, 2001; Croxall, 2008; Anderson *et al*., 2011; Croxall *et al*., 2012).

At least 160 000 seabirds are estimated to be killed annually from a combination of pelagic and demersal longline fisheries, of which at least 50 000 are from pelagic fisheries. This level of impact is not sustainable for some source populations and may drive some albatrosses and petrels to extinction (Anderson *et al*., 2011). Moreover, these estimates are likely to be only half of the actual mortality levels, because seabird catch data are not generally collected at the time when the lines are set (which is when most seabirds become entangled) but are instead usually recorded during line hauling, when many seabirds may have already been dislodged (Brothers *et al*., 2010). Taking these uncertainties into consideration, the catch in pelagic longline fisheries may be as high as 100 000 seabirds annually. Consequently, much effort has been made to acquire relevant quantitative data, including investigations into the spatio-temporal overlap between seabird species and longline fishing effort (e.g. Tuck, Polacheck and Bulman, 2003; BirdLife International, 2004; Waugh *et al*., 2005; Jiménez *et al*., 2010). In addition, in response to known or potential population level effects, various seabird mitigation measures have been developed and tested, some of which have been found to be highly effective in reducing seabird mortality in longline fisheries (see Section 4.3).

The taxonomy of the albatrosses has been debated ever since Linnaeus (1758) first established a single genus, *Diomedea*. Species have been repeatedly divided and lumped together (e.g. Reichenbach, 1852; Mathews, 1912, 1948; Marchant and Higgins, 1990; Robertson and Nunn, 1998), resulting in prolonged debate and controversy (see ACAP, 2010). The uncertainty has been due to the few specimens in collections, the poor quality of those that do exist in terms of provenance information and diagnostic detail (e.g. soft- and hard-part colours), the lack of adequate scientific knowledge of most taxa in live form, the complex age and sex-linked plumage changes (which gave a false impression of intra-specific variability), the lack of taxon-specific variation in vocalization, and the very homogeneous structure and appearance of albatrosses as a group (Onley and Scofield, 2007; ACAP, 2010). This uncertainty has led to errors in identification and categorization of incidentally captured seabirds, often persisting into

reported and/or published data. Some of the main and recent changes in albatross and petrel taxonomy are summarized below and in Table 15 to indicate the sources of some of the confusion.

From the 1930s to the early 1990s, largely as a reaction to perceived previous oversplitting, albatross taxa were placed in two genera with 13 or 14 species depending on the addition of the Amsterdam albatross (*Diomedea amsterdamensis*) (e.g. Marchant and Higgins, 1990; Sibley and Monroe, 1990; Robertson and Warham, 1992). In the late 1990s, on the basis of pioneering data from molecular genetics, radical changes were suggested by Robertson and Nunn (1998) proposing 4 genera with 24 species and promoting most subspecies to full species status. Almost all of these revisions have since been supported by genetic and biological research (e.g. Abbott and Double, 2003a, 2003b; Burg and Croxall, 2004; Chambers, Moeke and Steel, 2009; Rains, Weimerskirch and Burg, 2011). A derived taxonomic ranking, recognizing 22 albatross species and 2 subspecies, is currently supported by BirdLife International (2013) and ACAP (2013) and has been widely accepted.

Recent papers by Techow, Ryan and O'Ryan (2009) and Techow *et al*. (2010) support: the prevailing view that there are two, and not three, species of giant petrel *Macronectes*; the newly proposed species status for spectacled petrel (*Procellaria conspicillata*); and the lack of differentiation among white-chinned petrels (*P. aequinoctialis*), except for a well-marked taxon, at subspecies level, in the New Zealand region.

TABLE 15

Main and recent changes in albatross and petrel taxonomy

Taxonomy (1)	Taxonomy (2)	Taxonomy (3)	Common name
to c. 1995	*c. 1996 - c. 2003*	*c. 2004 - Present*	
Diomedea irrorata	→	→	Waved albatross
Diomedea albatrus	→	→	Short-tailed albatross
Diomedea nigripes	→	→	Black-footed albatross
Diomedea immutabilis	→	→	Laysan albatross
Diomedea exulans	→	*Diomedea dabbenena*	Tristan albatross
	Diomedea antipodensis		
	Diomedea gibsoni	*Diomedea antipodensis*	Antipodean albatross
	Diomedea chionoptera	*Diomedea exulans*	Wandering albatross
Diomedea amsterdamensis	→	→	Amsterdam albatross
Diomedea epomophora	→	→	Southern royal albatross
	Diomedea sanfordi	→	Northern royal albatross
Phoebetria fusca	→	→	Sooty albatross
Phoebetria palpebrata	→	→	Light-mantled albatross
Diomedea melanophrys	→	→	Black-browed albatross
	Thalassarche impavida	→	Campbell albatross
Diomedea cauta	→	→	Shy albatross
	Thalassarche steadi	→	White-capped albatross
	Thalassarche eremita	→	Chatham albatross

Taxonomy (1)	Taxonomy (2)	Taxonomy (3)	Common name
to c. 1995	*c. 1996 - c. 2003*	*c. 2004 - Present*	
	Thalassarche salvini	→	Salvin's albatross
Diomedea chrysostoma	→	→	Grey-headed albatross
Diomedea chlororhynchos	→	→	Atlantic yellow-nosed albatross
	Thalassarche carteri	→	Indian yellow-nosed albatross
Diomedea bulleri	→		
	Thalassarche platei		
		Thalassarche bulleri	Buller's albatross

(1) 14 species according to Sibley and Monroe (1990).

(2) 24 species according to Robertson and Nunn (1998).

(3) 22 species according to ACAP (2013) and BirdLife International (2013).

This review follows taxonomy used in BirdLife International (2012) and applies the term "albatrosses" collectively for all species of the Diomedeidae, and "petrels" for the Procellariidae. Table 16 provides a list of albatrosses and petrels known or considered to be vulnerable to tuna longline fisheries. The definition of "seabird catch" used in this review applies to incidental mortality caused by longline fishing operations. This happens primarily during line setting, when foraging seabirds are attracted to the bait, become hooked or entangled, and are then dragged under water and drown (Brothers, 1991; Nel *et al*., 2000). In addition, seabirds may also be hooked in the bill/gape, wing, leg or body during line hauling and escape, or be released, with the hook still attached, but most individuals later die of their injuries (Tuck, Polacheck and Bulman, 2003; Varty, Sullivan and Black, 2008).

TABLE 16
Albatross and petrel species known to be vulnerable to tuna longline fisheries

Family	Genus	Scientific name	Common name
Diomedeidae	Phoebastria (North Pacific Albatrosses)	*Phoebastria irrorata*	Waved albatross
		Phoebastria albatrus	Short-tailed albatross
		Phoebastria nigripes	Black-footed albatross
		Phoebastria immutabilis	Laysan albatross
	Diomedea (Great Albatrosses)	*Diomedea exulans*	Wandering albatross
		Diomedea antipodensis	Antipodean albatross
		Diomedea amsterdamensis	Amsterdam albatross
		Diomedea dabbenena	Tristan albatross
		Diomedea sanfordi	Northern royal albatross
		Diomedea epomophora	Southern royal albatross
	Phoebetria (Sooties)	*Phoebetria fusca*	Sooty albatross
		Phoebetria palpebrata	Light-mantled albatross
	Thalassarche (Mollymawks)	*Thalassarche melanophrys*	Black-browed albatross
		Thalassarche impavida	Campbell albatross
		Thalassarche cauta	Shy albatross
		Thalassarche steadi	White-capped albatross
		Thalassarche eremita	Chatham albatross
		Thalassarche salvini	Salvin's albatross
		Thalassarche chrysostoma	Grey-headed albatross
		Thalassarche chlororhynchos	Atlantic yellow-nosed albatross
		Thalassarche carteri	Indian yellow-nosed albatross
		Thalassarche bulleri	Buller's albatross

Family	Genus	Scientific name	Common name
Procellariidae	Macronectes	*Macronectes giganteus*	Southern giant petrel
		Macronectes halli	Northern giant petrel
	Daption	*Daption capense*	Cape petrel
	Procellaria	*Procellaria aequinoctialis*	White-chinned petrel
		Procellaria conspicillata	Spectacled petrel
		Procellaria westlandica	Westland petrel
		Procellaria parkinsoni	Black petrel
		Procellaria cinerea	Grey petrel
	Pterodroma	*Pterodroma cahow*	Bermuda petrel
	Puffinus	*Puffinus creatopus*	Pink-footed shearwater
		Puffinus griseus	Sooty shearwater
		Puffinus tenuirostris	Short-tailed shearwater
		Puffinus yelkouan	Yelkouan shearwater
		Puffinus mauretanicus	Balearic shearwater
		Puffinus gravis	Great shearwater
		Puffinus carneipes	Flesh-footed shearwater
		Puffinus pacificus	Wedge-tailed shearwater
		Puffinus huttoni	Hutton's shearwater
	Calonectris	*Calonectris edwardsii*	Cape verde shearwater
		Calonectris diomedea	Cory's shearwater

Source: Cooper and Baker (2008).

4.1.2 Factors influencing seabird interactions

Various operational, biological and environmental factors influence seabird–fishery interactions and thereby the nature and level of catches of albatrosses and petrels. In pelagic longline fisheries, the most important factors include the extent and season of fishing operations, the timing of longline setting and hauling, and the application and type of mitigation measures. Climate conditions such as wind speed/direction and cloud cover, lunar phase, oceanic characteristics associated with productivity such as SST and chlorophyll, and the availability of natural prey species have also been found to be relevant to seabird catch (Brothers, Gales and Reid, 1999; Weimerskirch, Capdeville and Duhamel, 2000; Hyrenbach, Anderson and Fernandez, 2002; Petersen *et al*., 2008). Seabird ecology, particularly foraging behaviour, may also have a significant effect on seabird–fishery interactions – different seabird species search for food at sea using different foraging strategies that may also differ between life-history stages and sex, and be influenced by dominance hierarchies between species and individuals (Granadeiro *et al*., 2011, Jiménez *et al*., 2011).

With so many potential factors involved, the interactions between seabirds and fishing fleets are complex, and result in large differences in the number of individuals caught by species and location (Weimerskirch, Capdeville and Duhamel, 2000). In the absence of sufficient data on seabird catch, the risk of catching seabirds can be broadly characterized by mapping the overlap between the spatio-temporal distributions of seabird foraging and longline fishing effort (e.g. Tuck, Polacheck and Bulman, 2003; BirdLife International, 2004; Waugh *et al*., 2005; Jiménez *et al*., 2010). Analysis of spatial and temporal overlap between seabird distribution and longline fisheries effort has already been used by t-RFMOs, alongside available data on seabird catch, to aid discussion of the potential impacts of longline fleets and to determine priorities and appropriate conservation action (e.g. BirdLife International 2006a, 2006b, 2007, 2008). Here, biological factors that potentially influence seabird interactions with longline fleets, specifically focusing on seabird foraging behaviour, are discussed.

4.1.2.1 Migration and at-sea distribution

Seabirds' long and narrow wings allow them to search for food over considerable distances using energy-efficient flight, referred to as "dynamic soaring" (Brooke, 2004). Their distributions may be remote and wide-ranging, and this has meant

at-sea distributions, migration routes and foraging areas were poorly understood until recent years when tracking devices (e.g. platform terminal transmitters, global positioning system loggers, and global location sensing loggers known as geolocators) became available. These tracking technologies (as well as direct observations and band recoveries, where tracking studies have not been conducted) have shown that many *Puffinus* and *Calonectris* species, with the exception of yelkouan (*P. yelkouan*) and Balearic (*P. mauretanicus*) shearwaters, have longitudinal interhemisphere migrations, whereas *Procellaria* species, with the exception of black petrel (*P. parkinsoni*), and the northern albatrosses generally have longitudinal, within-hemisphere migrations – these extend to circumglobal migrations for many of the southern hemisphere albatrosses (e.g. Croxall *et al*., 2005). Tracking studies have also revealed the extraordinary flight capability of seabirds. For example, male wandering albatrosses (*Diomedea exulans*) made foraging trips ranging from a minimum of 3 664 km in 14 days to a maximum of 15 200 km in 33 days during the incubation period (Jouventin and Weimerskirch, 1990), and grey-headed albatrosses (*D. chrysostoma*) after breeding circumnavigated the Southern Ocean in as few as 46 days (Croxall *et al*., 2005).

Tracking studies have revealed that foraging ranges and specific destinations may differ depending on the year, time of year, stage of the breeding cycle, age class, sex, specific source population and even among individuals (e.g. Prince *et al*., 1992; Weimerskirch *et al*., 1993; Weimerskirch, 1998; Weimerskirch and Wilson, 2000; Croxall *et al*., 2005; Phillips *et al*., 2008). Sufficient tracking data now exist to delineate the main foraging range (and to understand many aspects of the variation in this) for most seabird species that are highly susceptible to longline fisheries, throughout their breeding cycle stages and for most of their main nesting sites. There is also good knowledge of their principal wintering areas and the main migratory routes to these areas. These data have shown that during breeding, when adult seabirds are constrained by the need to return to the colony to defend nest sites, incubate eggs and feed their chicks, most species have restricted foraging ranges and areas (Mackley *et al*., 2010). However, after breeding, adult seabirds, especially those of highly pelagic species such as albatrosses, disperse much more widely, often exhibiting persistent individual preferences for wintering zones (Weimerskirch and Wilson, 2000; Croxall *et al*., 2005; Phillips *et al*., 2005, 2008). These non-breeding seabirds often show evidence of one or more of three distinct migration strategies: (i) residence within the breeding season home range; (ii) focused migrations to specific wintering areas; and (iii) wide dispersion across oceans, including foraging in areas used in (i) and (ii) above (Croxall *et al*., 2005).

Tracking data sets have proved highly valuable in identifying the areas and times of the greatest potential for interaction between seabirds and fishing effort. However, these data are still sparsely available for juveniles and immature seabirds, and it is these individuals that may be most important because, for some species, immature seabirds in particular regions are suspected to experience higher levels of incidental mortality than mature adults (e.g. shy albatross [*Thalassarche cauta*] in South African waters [Ryan, Keith and Kroese, 2002] and black-browed albatrosses [*T. melanophrys*] in the central and eastern South Atlantic [Phillips *et al*., 2005]).

4.1.2.2 Ship-following behaviour

Except for the two species of giant petrel, which also scavenge on land, albatross and petrel species forage exclusively at sea, congregating at areas of high ocean productivity (Hunt and Schneider, 1987; González-Solís, Croxall and Wood, 2000). Many species are surface feeders, not only catching squid, fish and crustaceans from the surface of the sea but also scavenging discards and offal from fishing vessels (Croxall and Prince 1994). Some seabird species such as albatrosses and larger petrels (body mass > 600 g, petrels listed in Table 16) are known to be strongly attracted to such vessels, and often gather in large numbers during longline fishing operations (Baker *et al*., 2002). This

behaviour makes them particularly vulnerable. At the fishing vessels, albatrosses and petrels often compete inter- and intra-specifically for the "free" food discarded by the vessels, with larger species having a higher probability of success (Jiménez *et al*., 2011).

4.1.2.3 Diurnal/nocturnal behaviour

Many seabirds, albatrosses in particular, are most active during daylight hours (often especially so at dawn and dusk) and less so at night (except during periods of full moon), probably because their visual acuity makes it difficult for them to locate prey in darkness (Hedd, Gales and Brothers, 2001; Phalan *et al*., 2007; Phillips *et al*., 2008). They may also spend time foraging at night, apparently relying more on olfactory cues (Nevitt, Losekoot and Weimerskirch, 2008). Unlike these active diurnal feeders, some petrels, including white-chinned, black and spectacled petrels, are known to forage actively by both night and day (Shealer, 2001; ACAP, 2009a, 2009b).

Studies have shown that these foraging behaviours are important in explaining the temporal variation in the catch rate among seabird species. Examples include the higher incidental mortality of Cory's shearwaters (*Calonectris diomedea*) from the middle of the afternoon until dusk, of shy albatrosses at night under bright moon conditions (Hedd *et al*., 1998), and of white-chinned petrels during the night (Mackley *et al*., 2010) probably related to their high levels of nocturnal activity.

4.1.2.4 Feeding methods

Albatross and petrel feeding methods also affect the vulnerability of the species. Most species are surface feeders. In trying to take bait as lines are set or hauled, they become hooked or entangled in the fishing gear and are subsequently killed. In these instances, the interactions are considered "primary". Most albatross species generally dive only up to 2 m, although some smaller species, such as shy, black-browed, grey-headed and light-mantled (*P. palpebrata*) albatrosses, are able to plunge dive for their prey and may reach maximum depths of 10–12 m (Hedd *et al*., 1998; Prince, Huin and Weimerskirch, 1994). Petrels, in contrast, and particularly those of the genus *Puffinus* (and *Procellaria* to a lesser extent), are known to dive relatively deeply with maximum depths greater than 10 m and frequently attaining 40–50 m (e.g. Weimerskirch and Cherel, 1998; Ronconi, Ryan and Ropert-Coudret, 2010; Rayner *et al*., 2011). These deeper-diving species are not only vulnerable to being killed by the primary attack, they can also contribute to secondary attacks in which another seabird or a group of seabirds attempts to steal a bait or baited hook that has been returned to the surface by a diving species (Melvin, Guy and Read, 2014), thus making the bait available to seabirds on the surface, including "great" albatrosses, which are unable to dive. Jiménez *et al*. (2011) reported that high densities of these petrel species increased longline catches of albatrosses, apparently because of this interaction. Successful seabird mitigation measures must avoid impacts caused by both primary and secondary attacks (Melvin, Heinecken and Guy, 2009; and see Section 4.3.1).

4.1.2.5 Collateral effects and cryptic mortality

In addition to direct adverse effects on seabird populations, longline fisheries may also result in various indirect collateral effects. In scavenging from fishing vessels, seabirds are modifying their natural diet by taking demersal fish that are inaccessible to them under natural conditions (Tasker *et al*., 2000). This may have positive effects for some seabird species. For example, discards from trawl and purse seine fishery could increase hatching success and nesting success of Balearic shearwater (Louzao, Igual and McMinn, 2006), and fledging success of gulls was higher in a discard-rich year than in a discard-poor year (Oro, Jover and Ruiz, 1996). However, breeding success decreased in Cape gannet (*Morus capensis*) when chicks received low-energy food from discards

(Gremillet and Pichegru, 2008). Moreover, reliance on discards may cause negative population-level effects if discards become less available (Tasker *et al*., 2000).

Fishing activities might decrease availability of small fish, important in the diet of many seabirds, through reduction in the relative or absolute abundance of tuna, which bring baitfish to the surface where they become accessible to seabirds (Furness and Tasker, 1997; Tasker *et al*., 2000). Conversely, some fisheries may lead to increases in small fish by reducing numbers of larger fish that may compete with seabirds (Furness and Tasker, 1997).

Most species of albatross and petrel vulnerable to longline fisheries have lengthy chick-rearing periods, and fledging success depends on biparental care. Loss of one member of the pair almost invariably results in the death of the chick (reduced productivity) and also lowers the reproductive rate because it takes two or more years for adults to form a new pair-bond (Gilman *et al*., 2013).

Another issue concerns cryptic mortality of seabirds, such as pre-catch losses, ghost fishing and post-release fishing mortality. These are well recognized in relation to fish species (e.g. Suuronen *et al*., 1996), but for seabirds, because of observation difficulties, data are few. However, in longline fisheries, it is well known that the observed catch rates significantly underestimate actual catch rates (Gales, Brothers and Reid, 1988; Brothers *et al*., 2010; Anderson *et al*., 2011) and as few as 50 percent of all seabirds observed caught during line setting may be retrieved when the line is hauled aboard because they have died and dropped off prior to hauling. It is noted, however, that some seabirds observed caught during the set might dislodge the hook, or disentangle from the line, and escape (Brothers *et al*., 2010). No data are currently available for seabirds concerning post-release mortality and effects of ghost fishing.

4.1.3 Species risk profiles

Seabirds are among the most threatened group of birds and their status has continued to deteriorate over the years. In particular, the family Diomedeidae (albatrosses) has a higher proportion of threatened species than any other bird family (Butchart *et al*., 2004; Croxall *et al*., 2012). The 2013 IUCN Red List designates 17 of the 22 species of albatrosses as threatened with extinction, and of these, 3 species are Critically Endangered (CR), 5 are Endangered (EN), 9 are Vulnerable (VU), and all the remaining species are Near Threatened (NT), with none listed as Least Concern (LC) (IUCN 2013a; Table 17). Despite the overall poor status of the species, some breeding sites have shown increases in population size, whereas most others have been decreasing. The four species with increasing populations are still considered threatened, mainly because their population sizes are small and are still recovering from past, major declines. For example, aside from the exception of black-footed albatross (*Phoebastria nigripes*), populations are all fewer than 50 000 (11 000 for Chatham albatross [*T. eremita*], fewer than 2 500 for short-tailed albatross [*P. albatrus*], and 49 000 for Campbell albatross [*T. impavida*]). The remaining species have populations that are either stable (3 species), declining (13 species) or unknown (2 species). Those most frequently interacting with longline fisheries are those with small global populations.

A recent paper by Croxall *et al*. (2012) reviewed threats to seabirds, including invasive alien species, bycatch, climate change/severe weather, pollution, human disturbance and problematic native species. When considering pelagic seabirds alone, bycatch was identified as the threat with the highest impact. Indeed, evidence has shown that declines in the numbers of many pelagic seabird species are closely linked to interactions with longline fishing operations (both pelagic and demersal), particularly those of the Southern Ocean (e.g. Brothers, 1991; Croxall *et al*., 1998; Gales, Brothers and Reid, 1998; Klaer and Polacheck 1998; Inchausti and Weimerskirch, 2001; Croxall, 2008; Jiménez, Domingo and Brazeiro, 2009; Anderson *et al*., 2011). These pelagic seabird species, such as the albatrosses and petrels in Table 16, typically have small

TABLE 17
IUCN Red List status for albatross species including total population size and population status

Common name	The total number of breeding pairs	The total number of mature individual	2013 IUCN	Population trend in each area based on breeding colonies				Overall trend
				Atlantic	*Eastern Pacific*	*Western and Central Pacific*	*Indian*	
Waved Albatross	15 600–18 200	34 700	CR	–	↓	–	–	↓
Short-tailed Albatross	–	2 200–2 500	VU	–	–	↑	–	↑
Black-footed Albatross	64 500	120 000	VU	–	Unknown	Variable	–	↑
Laysan Albatross	590 926	1 180 000	NT	–	Unknown	Variable	–	→
Wandering Albatross	8 114	26 000	VU	↓	–	Variable	Variable	↓
Antipodean Albatross	8 050	44 500	VU	–	–	↓ / Unknown	–	↓
Amsterdam Albatross	24–31	100	CR	–	–	–	↑	↓
Tristan Albatross	2 700	11 300	CR	↓	–	–	–	↓
Northern Royal Albatross	5 800	17 000	EN	–	–	↑ / ↓	–	↓
Southern Royal Albatross	8 200–8 600	28 000–29 500	VU	–	–	→	–	→
Sooty Albatross	12 500–19 000	42 000	EN	↓ / Unknown	–	–	↓ / Unknown	↓
Light-mantled Albatross	19 000–24 000	58 000	NT	Unknown	–	→	↑	↓
Black-browed Albatross	600 852	1 200 000	EN	↓	Unknown	→ / Unknown	→ / Unknown	↓
Campbell Albatross	24 600	49 000	VU	–	–	↑	–	↑
Shy Albatross	12 000–13 000	26 000	NT	–	–	Variable	–	Unknown
White-capped Albatross	95 000	100 000–499 999	NT	–	–	Unknown	–	↓
Chatham Albatross	5 247	11 000	VU	–	–	→	–	↑
Salvin's Albatross	30 750	62 000	VU	–	–	Unknown	Variable	Unknown
Grey-headed Albatross	99 000	250 000	VU	↓	Unknown	↓ / →	–	↓
Atlantic Yellow-nosed Albatross	27 500–41 600	55 000–83 200	EN	↓	–	–	–	↓
Indian Yellow-nosed Albatross	41 580	85 000	EN	–	–	–	↓ / Unknown	↓
Buller's Albatross	31 939	64 000	NT	–	–	↑ / Unknown	–	→

Sources: IUCN (2013a). Details of the data sources and assessments are provided in BirdLife International (2012, 2013) and ACAP (2013).

breeding populations, a limited number and range of breeding sites, and exceptionally high k-selected life history traits (e.g. delayed sexual maturity, low breeding frequency, small clutch size, prolonged breeding seasons, and long life span with high adult survivorship under natural conditions) (Phillips *et al*., 2008; Croxall *et al*., 2012). This makes them extremely vulnerable to increased levels of mortality and makes it difficult for populations to recover once they have declined (Delord *et al*., 2008, Phillips *et al*., 2008, Croxall *et al*., 2012), especially if the causes of decline are not eliminated or substantially reduced.

4.2 SEABIRD INTERACTIONS BY AREA

In the late 1980s and early 1990s, despite the fact that data remained sparse and were predominantly from the Oceania region, studies highlighted the potential scale of seabird catch in pelagic longline fisheries (e.g. Brothers, 1991; Vaske, 1991; Murray *et al*., 1993). By the early 2000s, seabird catch had been reported from eight pelagic longline fishing nations worldwide (Brothers, Cooper and Løkkeborg, 1999; Nel and Taylor, 2003), although, based on seabird range data, interactions were potentially occurring in at least 25 countries (Nel and Taylor, 2003). A decade later, onboard observer seabird-catch data collection remains sparse in most pelagic longline fisheries, much lower than the levels recommended for effective monitoring of seabird interaction levels (Lawson, 2006). However, a review in 2011 identified that data had been reported from more than 25 pelagic longline fisheries, covering 15 countries (Anderson *et al*., 2011). In addition to these fleets, seabirds are also caught by illegal, unreported and unregulated (IUU) longline fisheries. Owing to the very nature of IUU fishing, it is extremely difficult to quantify these interactions accurately. However, one estimate of seabird catch by IUU fishing in tuna and swordfish longline fisheries in the high seas south of 30°S suggested that 2 739–6 326 seabirds are caught each year (MRAG, 2005).

In the absence of representative data from longline fisheries, alternative approaches have been used to attempt to quantify the impacts of longline fisheries on seabird populations, particularly on albatrosses and petrels. These have included: use of demographic data to identify impacts on adult survival and juvenile recruitment (e.g. Weimerskirch and Jouventin, 1987; Croxall *et al*., 1990; Weimerskirch, Brothers and Jouventin, 1997; Cuthbert *et al*., 2003); population modelling (Moloney *et al*., 1994; Tuck *et al*., 2001; Tuck, 2004; Wanless *et al*., 2009, Tuck *et al*., 2011a); identification of spatial and temporal overlap between seabird distribution and pelagic longline fishing effort (BirdLife International, 2004; Cuthbert *et al*., 2005; Phillips *et al*., 2006; Suryan *et al*., 2007); and, more recently, ecological risk assessment (reviewed in Small, Waugh and Phillips, 2013). These approaches have been facilitated by: (i) the availability of long-term and detailed demographic data from a number of albatross breeding sites, particularly from South Georgia and the South Sandwich Islands in the South Atlantic, and Kerguelen Islands and Crozet Islands in the Indian Ocean; (ii) the increasing availability of remote-tracking data for seabirds, enhanced by the development of ever-smaller and longer-lasting tracking devices, allowing a wider range of species to be tracked across more of their life cycle; and (iii) the collaboration since 2003 of albatross and petrel tracking data owners worldwide to establish the Global Procellariiform Tracking Database, in order to facilitate such analysis (BirdLife International, 2004) The majority of the analyses of remote-tracking data referred to in the sections below were undertaken through collaboration between the data contributors to the Global Procellariiform Tracking Database to produce a document distributed at the Third Joint Meeting of the Tuna RFMOs (Alderman *et al*., 2011).

4.2.1 Atlantic

In 2007–2010, ICCAT undertook an assessment of the threat from Atlantic pelagic longline fisheries to seabirds that breed in the Atlantic. More than 60 seabird populations were considered, and the assessment involved: the identification of populations most likely to be at risk, analyses of overlap with fishing effort, estimation of total annual seabird catch, and an evaluation of the estimated impact on selected populations for which there were sufficient data on bird distribution and demography (Tuck *et al*., 2011b).

The risk assessment identified that the populations at the highest level of risk from pelagic longline fisheries included 6 of the 7 species of albatross breeding in the Atlantic, together with 2 species of petrel (white-chinned and grey [*P. cinerea*]), 6 species of shearwaters (Balearic, Cape Verde [*C. edwardsii*], Cory's, great [*P. gravis*], sooty [*P. griseus*] and yelkouan [*P. yelkouan*]), and the southern giant petrel (*M. giganteus*) (Phillips and Small 2007; Table 18).

TABLE 18
High-risk seabird populations identified by the ICCAT preliminary risk assessment

Species	Breeding island group	IUCN status	Population status	Overlap with ICCAT	Behavioural susceptibility to capture	Life-history strategy	Risk score 1	Risk score 2	Risk score 3
Sooty albatross (*Phoebetria fusca*)	Tristan da Cunha	3	3	3	?	3	4.2	3	4.24
Tristan albatross (*Diomedea dabbenena*)	Tristan da Cunha	3	3	3	3	3	4.1	3	4.24
Wandering albatross (*Diomedea exulans*)	South Georgia and the South Sandwich Islands	2	3	3	3	3	3.8	3	4.24
Grey-headed albatross (*Thalassarche chrysostoma*)	South Georgia and the South Sandwich Islands	2	3	3	3	3	2.8	3	4.24
Black-browed albatross (*Thalassarche melanophrys*)	South Georgia and the South Sandwich Islands	3	3	3	3	2	2.8	2.75	3.91
Atlantic yellow-nosed albatross (*Thalassarche chlororhynchos*)	Tristan da Cunha	3	3	3	3	2	2.8	2.75	3.91
Black-browed albatross (*Thalassarche melanophrys*)	Falkland Islands (Malvinas)	3	2	3	3	2	2.6	2.5	3.91
Balearic shearwater (*Puffinus mauritanicus*)	Balearics	3	3	3	3	2	2.8	2.75	3.61
White-chinned petrel (*Procellaria aequinoctialis*)	South Georgia and the South Sandwich Islands	2	3	3	3	2	2.6	2.75	3.61
Southern giant petrel (*Macronectes giganteus*)	Chile	1	?	3?	3	2	2.4	2.75	3.61
Grey petrel (*Procellaria cinerea*)	Prince Edward Island	1	?	?	3	2	2.4	2.75	3.61

Species	Breeding island group	IUCN status	Population status	Overlap with ICCAT	Behavioural susceptibility to capture	Life-history strategy	Risk score 1	Risk score 2	Risk score 3
Grey petrel (*Procellaria cinerea*)	Crozet	1	?	?	3	2	2.4	2.75	3.61
Grey petrel (*Procellaria cinerea*)	Kerguelen Islands	1	?	?	3	2	2.4	2.75	3.61
Grey petrel (*Procellaria cinerea*)	Tristan da Cunha	1	?	3	3	2	2.4	2.75	3.61
Cory's shearwater (*Calonectris diomedea*)	Mediterranean	0	3	3	3	2	2.2	2.75	3.61
Cape Verde shearwater (*Calonectris edwardsii*)	Capo Verde	1	1?	3	?	2	2.4	2.75	3.61
Great shearwater (*Puffinus gravis*)	Tristan da Cunha	0	?	3	3	2	2.2	2.75	3.61
Grey-headed albatross (*Thalassarche chrysostoma*)	Indian Ocean	2	3	1	3	3	2.4	2.5	3.61
Wandering albatross (*Diomedea exulans*)	Crozet	2	2	1	3	3	2.2	2.25	3.61
Sooty albatross (*Phoebetria fusca*)	Indian Ocean	3	3	1	?	3	2.6	2.5	3.61
Grey-headed albatross (*Thalassarche chrysostoma*)	Chile	2	?	1	3	3	2.4	2.5	3.61
Sooty shearwater (*Puffinus griseus*)	Falkland Islands (Malvinas)	1	2	3	?	2	2.2	2.5	3.61
Yelkouan shearwater (*Puffinus yelkoan*)	Mediterranean	0	2	3	3?	2	2	2.5	3.61

Notes:

Key to table: a) global IUCN status: critically endangered/endangered = 3, vulnerable = 2, near threatened = 1 and least concern = 0; b) breeding population status: rapid decline (> 2 percent/year) = 3, decline = 2, stable = 1, increase = 0; c) degree of overlap with ICCAT fisheries: high = 3, medium = 2, low = 1, based on available information (of varying quality) on year-round distribution, and pending more detailed analysis of tracking data; d) behavioural susceptibility to capture: high = 3, low = 1, based on the tendency to follow fishing vessels and relative interaction rates in ICCAT or other fisheries; e) life-history strategy: biennial breeder/single egg clutch = 3, annual breeder/single egg clutch = 2, annual breeder/multiple egg clutch = 1; risk score 1: sum of the five attributes a)–e); risk score 2: the sum of attributes b)–e), i.e. excluding global IUCN status, which duplicates population status to some extent; risk score 3: the Euclidean distance to the origin of an integrated index of potential susceptibility to ICCAT fisheries (the mean of overlap with ICCAT and behavioural susceptibility to fisheries) plotted against life-history strategy.
Source: Phillips and Small (2007).

Remote tracking data show that more than 20 percent of the total global distribution of breeding albatrosses during the breeding season occurs in the Atlantic, along with an estimated 13 percent of total global non-breeding albatross distribution (including juveniles and immature individuals, plus adults during the non-breeding season) (Tables 19 and 20). Albatross and petrel distribution in the Atlantic is concentrated south of 30°S, although this extends northward to 20°S, and even to 10°S along the coast of Namibia and Angola (ACAP, 2010; Figures 31 and 32). Cory's shearwater is also distributed exclusively and widely in the Atlantic and Mediterranean (ACAP, 2010). Tracking data have also identified a high degree of overlap between ICCAT

pelagic longline fishing effort and Tristan and Atlantic yellow-nosed albatrosses and Cory's shearwater (> 75 percent of their year-round distribution) (Taylor, Anderson and Small, 2009; ACAP, 2010). Black-browed albatross and white-chinned petrel also have high degrees of overlap with Atlantic pelagic longline fishing effort during their non-breeding season (April–September), when they migrate northwards (ACAP, 2010; Table 20). Wandering and grey-headed albatross have lower levels of overlap with Atlantic pelagic longline fishing effort, as they tend to forage at higher latitudes. However ICCAT longline fishing effort extends into areas between 30 and 50°S in April–June, particularly offshore of Uruguay and southwest Brazil, and southwest of South Africa, resulting in overlap. In addition, there is overlap between Atlantic pelagic longline fishing effort and the non-breeding distribution of seabirds from New Zealand and Australia, including northern royal (*D. sanfordi*) and white-capped albatrosses (ACAP, 2010; Tuck *et al.*, 2011b; Table 20). This analysis (from 2010) lacked data for Balearic shearwater, spectacled petrel and grey petrel, as well as various Mediterranean and North Atlantic species, and further work is needed to understand the overlap of these species with pelagic longline effort.

TABLE 19
Percentage time spent at sea within waters managed by t-RFMOs during the breeding season for all albatross and giant petrel species, based on data held in the Global Procellariiform Tracking Database

	Threat status	Percentage global population tracked	CCSBT	IATTC	ICCAT	IOTC	WCPFC
Albatrosses							
Amsterdam albatrosses	CR	100	100			100	
Antipodean albatrosses	VU	97	91	2		1	97
Atlantic yellow-nosed albatrosses	EN	20	98		100	1	
Black-browed albatrosses	EN	71	49	4	18		
Black-footed albatrosses	VU	95		21			78
Buller's albatrosses	NT	44	88			1	88
Campbell albatrosses	VU	100	59				72
Chatham albatrosses	VU	100	99				99
Grey-headed albatrosses	VU	38	31	1	4	9	14
Indian yellow-nosed albatrosses	EN	70	100			100	
Laysan albatrosses	NT	100		14			84
Light-mantled albatrosses	EN	30	2		1	1	8
Northern royal albatrosses	EN	100	98				98
Salvin's albatrosses	VU	4	85				91
Short-tailed albatrosses	VU	89					97
Shy albatrosses	NT	100	82			81	81
Sooty albatrosses	EN	52	75		47	5	
Southern royal albatrosses	VU	99	98				98
Tristan albatrosses	CR	100	97		100		

	Threat status	Percentage global population tracked	CCSBT	IATTC	ICCAT	IOTC	WCPFC
Wandering albatrosses	VU	100	90		7	24	
Waved albatrosses	CR	99		89			
White-capped albatrosses	NT	99.9	92			11	92
Southern giant-petrel	LC	21	25		24		13
Northern giant-petrel	LC	51	16		15	1	28
Combined % time in RFMO by tracked non-breeding seabirds			61	5	13	14	47

Notes: Threat status: LC = least concern (not threatened); NT = near threatened; VU = vulnerable; EN = endangered; CR = critically endangered (IUCN, 2013a). Percentage of global population tracked: the proportion of the global breeding population represented by the breeding pairs at each site for which tracking data were available.
Source: Reproduced with permission from Alderman *et al*. (2011).

TABLE 20
Percentage time spent at sea within waters managed by t-RFMOs during the non-breeding season for all albatross and giant petrel species, based on data held in the Global Procellariiform Tracking Database

	Threat status	Percentage global population tracked	CCSBT	IATTC	ICCAT	IOTC	WCPFC
Albatrosses							
Amsterdam albatrosses	CR	100	100		2	98	
Antipodean albatrosses	VU	97	80	6		12	84
Atlantic yellow-nosed albatrosses	EN	20	84		98	2	
Black-browed albatrosses	EN	50	68	2	58		
Black-footed albatrosses	VU	95		43			50
Buller's albatrosses	NT	44	59	36		7	63
Campbell albatrosses	VU	0					
Chatham albatrosses	VU	100	14	77			27
Grey-headed albatrosses	VU	23	10	9	7	2	1
Indian yellow-nosed albatrosses	EN	65	100			93	
Laysan albatrosses	NT	99		1			97
Light-mantled albatrosses	EN	0					
Northern royal albatrosses	EN	100	57	24	40	4	14
Salvin's albatrosses	VU	4	48	41			56
Short-tailed albatrosses	VU	89		2			93
Shy albatrosses	NT	100	100			100	46
Sooty albatrosses	EN	34	58		53	9	
Southern royal albatrosses	VU	0					

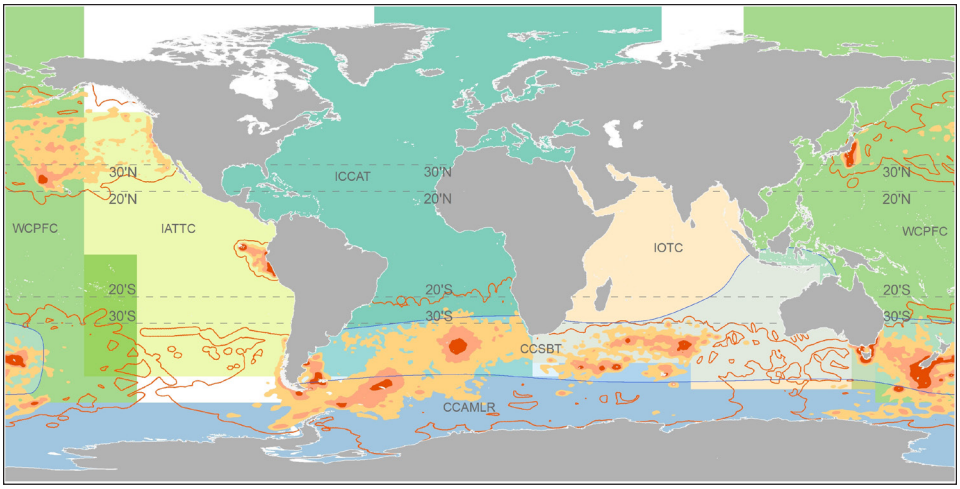
	Threat status	Percentage global population tracked	CCSBT	IATTC	ICCAT	IOTC	WCPFC
Tristan albatrosses	CR	100	93		84	13	
Wandering albatrosses	VU	55	93	2	5	73	10
Waved albatrosses	CR	0					
White-capped albatrosses	NT	99	95		47	5	47
Southern giant-petrel	LC	18	30	1	27	1	9
Northern giant-petrel	LC	51	32	6	24		17
Combined percent time in RFMO by tracked non-breeding seabirds			56	12	22	21	31

Notes: Threat status: LC = least concern (not threatened); NT = near threatened; VU = vulnerable; EN = endangered; CR = critically endangered (IUCN, 2013a). Percentage of global population tracked: the proportion of the global breeding population represented by the breeding pairs at each site for which tracking data were available.
Source: Reproduced with permission from Alderman *et al*. (2011).

In all, 37 species of seabirds have been recorded as caught in ICCAT fisheries (Phillips, Small and Howgate, 2007). As part of the ICCAT seabird assessment, an attempt was made to estimate the total number of seabirds killed by Atlantic pelagic longline fleets per year. Using available seabird catch data and annual fishing effort data on a 5 × 5 grid (but not taking seasonal variation into account), and applying assumed seabird catch rates for fisheries where no actual rates were available, the number of seabirds killed in ICCAT pelagic longline fisheries was estimated to be 16 568 in 2003, 10 021 in 2004, 9 879 in 2005, and 12 081 in 2006, with annual variation broadly reflecting changes in fishing effort (Klaer, 2012). However, the study also noted that the estimates were hampered by data gaps, including few data available from distant-water fleets.

FIGURE 31

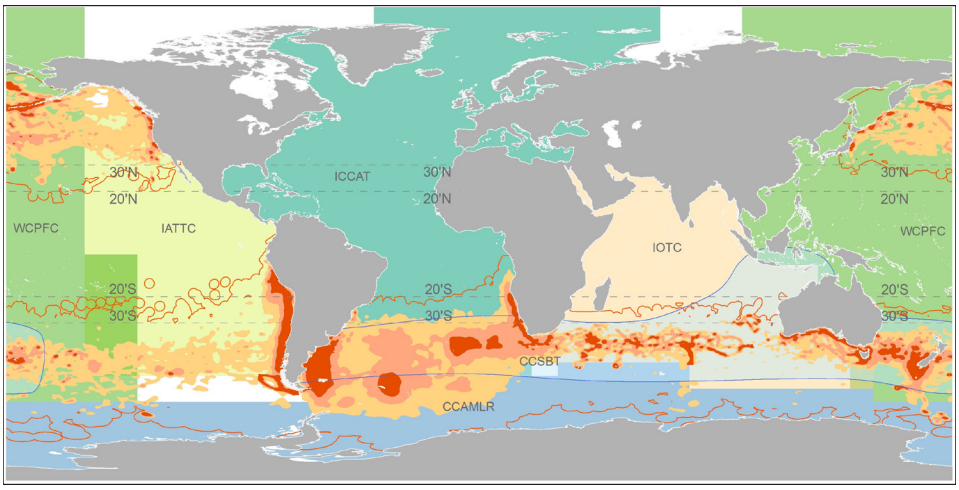
Global density distribution of albatrosses and giant-petrels during their breeding season, in relation to the areas managed by the t-RFMOs and the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR)



Notes: Red, pink and orange shaded areas indicate the 50, 75, and 95 percent probability contours of albatross and giant-petrel distributions, and the brown line indicates the full range based on data available to these analyses.
Source: Alderman et al. (2011).

Yeh *et al.* (2013) independently estimated 3 446–6 083 seabirds killed per year in ICCAT fisheries, making use of 5 years of data from the Taiwan Province of China observer programme and data from other published sources. However, it is not clear how to compare this estimate with the one by Klaer (2012), in part because of a continuing lack of data from the central south Atlantic (Phillips, 2013).

FIGURE 32
Global density distribution of albatrosses and giant-petrels during their non-breeding season, in relation to the areas managed by the t-RFMOs and the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR)



Notes: Red, pink and orange shaded areas indicate the 50, 75, and 95 percent probability contours of albatross and giant-petrel distributions, and the brown line indicates the full range based on data available to these analyses.
Source: Alderman et al. (2011).

4.2.1.1 South Atlantic

In the South Atlantic, major species caught by longline fisheries are black-browed albatross, Atlantic yellow-nosed albatross, wandering albatross, Tristan albatross, shy-type albatrosses (*T. cauta* and *T. steadi*), grey-headed albatross, and white-chinned petrel (Tuck *et al.*, 2011b; Yeh *et al.*, 2013). Annual survival of Atlantic yellow-nosed albatross on Tristan da Cunha has been found to be negatively correlated with pelagic longline fishing effort in the South Atlantic Ocean (Cuthbert *et al.*, 2003). Population models of Atlantic yellow-nosed, black-browed and wandering albatrosses, undertaken as part of the ICCAT seabird assessment, also demonstrated the impacts of fishing (for all gear types) on these species, and highlighted the unsustainability of current seabird catch levels (Thomson, Phillips and Tuck, 2009; Tuck *et al.*, 2011b).

The largest pelagic longline fleets in the South Atlantic south of 25°S are those of Taiwan Province of China and Japan, followed by Spain, Brazil, South Africa and Uruguay (ICCAT, 2013). Using data from Taiwan Province of China, Yeh *et al.* (2013) estimated that seabird catches by distant-water fleets ranged from 2 234 to 4 141 seabirds killed per year between 2004 and 2008 (not partitioned by fleet), of which 61 percent were albatrosses (although more than two-thirds of the seabirds recorded by Taiwan Province of China could not be identified to species). This compares with previous estimates of 936 seabirds killed per year for the fleet of Taiwan Province of China alone in the South Atlantic (Huang, Chang and Tai, 2009).

Data from the Japanese Southern Bluefin Tuna fleet observer programme between 1997 and 2009 report an average seabird catch rate of 0.3 seabirds per 1 000 hooks for the southeast Atlantic (below 30°S) in April–September (no data available for October–March). Grey-headed albatross was the species most commonly caught, followed by

black-browed, wandering and white-capped albatross (*T. steadi*), both giant petrels (*M. giganteus* and *M. halli*), grey petrel and white-chinned petrel (Inoue *et al*., 2012). In addition, Japanese data offshore from Namibia (15–20°S) recorded catch of white-chinned petrels at rates of > 0.1 seabirds killed per 1 000 hooks, although this was based on a small sample size and fewer than 10 individual seabirds were recorded caught (Inoue *et al*., 2012).

No data are available from the Republic of Korea for the South Atlantic, although Anderson *et al*. (2011) estimated that if the fleet had a similar seabird catch rate to the fleet of Taiwan Province of China, then this would amount to 67 seabirds killed per year, based on fishing effort reported by the fleet of Taiwan Province of China in the South Atlantic. In addition, no seabird catch data are available from China's fleet in the Atlantic; however, data reported to ICCAT in 2002–2010 indicate that all the fishing effort for the Chinese fleet was north of 25°S (ICCAT, 2013).

Seabird catch data are available for Spanish pelagic longline vessels undertaking experimental research in the Atlantic Ocean, fishing between 47.5°N and 22.5°S (Mejuto, Garcia-Cortes and Ramos-Cartelle, 2008). However, no data are available south of 25°S, where Spanish fishing effort (mainly targeting swordfish) deployed an average of 2.5 million hooks per year in 2002–06 and 1.75 million hooks per year in 2006–2010 (ICCAT, 2008b, 2013).

About 2 000–3 000 seabirds are estimated to be killed per year in the Brazil domestic pelagic longline fishery (Anderson *et al*., 2011), based on an average mortality rate of 0.229 seabirds per 1 000 hooks and an annual estimate of 9 million hooks per year (Bugoni *et al*., 2008). Seabird catch rates were highest in June–November and affected mainly black-browed albatross (55 percent of seabirds captured), white-chinned petrel, spectacled petrel, and Atlantic yellow-nosed albatross (Bugoni *et al*., 2008).

High seabird catch rates have been recorded off Uruguay, reflecting the high densities of albatrosses and petrels foraging in this area. Data from the period 1998–2004 indicate an average mortality rate of 0.42 seabirds killed per 1 000 hooks (range 0.11–2.48 seabirds killed per 1 000 hooks; Jiménez, Domingo and Brazeiro, 2009). However, fishing effort by the pelagic longline fleet has fluctuated considerably in recent years, making it difficult to estimate an average number of seabirds killed per year.

In South African waters, the foreign-flagged fleet is required to have 100 percent observer coverage, and seabird catch is closely monitored. Vessels are required to use a combination of seabird mitigation measures and, since 2008, a cap has been placed on allowable seabird mortality. As a result, seabird catches have been reduced by more than 80 percent in this fishery. The most recent data available (Ryan *et al*., 2009) estimate 35 seabirds killed per year. Seabird catch in the South African longline fleet (operating within its exclusive economic zone) is estimated at 299 seabirds per year across the Atlantic and Indian Oceans combined, based on data from the period 1998–2005 (Petersen, 2008).

Available seabird catch data from the Namibian pelagic longline fleet date from 2004 to 2006. Based on these data, and fishing effort data from that period, Petersen, Honig and Nel (2007) estimated about 207 seabirds killed per year. No data are available from Angola, although if seabird catch rates are similar to those in Namibia, about 250 seabirds would be killed per year, based on available data on fishing effort (Anderson *et al*., 2011).

4.2.1.2 North Atlantic

Fewer seabird data are available from the North Atlantic Ocean, but in general the North Atlantic fleets are assumed to be having a lower impact on seabirds (lower seabird catch rates on less vulnerable species) compared with those in the South Atlantic.

The Canadian pelagic longline fleet operating off the Scotian Shelf and Grand Banks is estimated to have a seabird catch of 1 400 seabirds per year (mostly northern gannet [*Morus bassanus*], herring gull [*Larus argentatus*] and great black-backed gull [*L. marinus*]), estimated from observer coverage of 3–10 percent per year and data from the period 1986–1999 (Fisheries and Oceans Canada 2007). Hata (2006) estimated average annual seabird catch in the United States pelagic longline fisheries in the northwest Atlantic, Caribbean and Gulf of Mexico to be about 230 seabirds per year (range: 139–333 seabirds per year), based on data from 1992 to 2004, with great shearwaters being the most commonly caught species, followed by gulls and northern gannets. A more recent study estimated 81 seabirds were killed in 2008 (Winter, Jiao and Browder, 2011).

Seabird catch data are available for Spanish pelagic longline vessels fishing in the Atlantic Ocean between 47.5°N and 22.5°S, covering more than 430 000 hooks (Mejuto, Garcia-Cortes and Ramos-Cartelle, 2008). No seabird catches were reported. However, the data were collected during cruises that were testing baits (squid versus mackerel) and hook types (circle, semi-circular and J hooks) in sets from late afternoon to midnight as part of sea turtle mitigation trials. It is possible that these factors may have affected seabird interaction rates.

Yeh *et al.* (2013) identified that the bulk of fishing by the Asian fleets occurred in the north-central Atlantic Ocean, but seabird catch data from Taiwan Province of China were sparse, and the authors concluded it was impossible to extrapolate to estimate an annual seabird catch figure. Japanese data have also been reported for the central and north Atlantic from 1997 to 2009. These data indicate generally low seabird catch rates in these areas, but some noteworthy interactions have been reported, including northern gannets caught off Western Sahara, and gulls caught off North Carolina, the United States of America (Inoue *et al.*, 2012).

4.2.1.3 Mediterranean

In the Mediterranean, the most substantial pelagic longline seabird catch data to date have come from Spanish vessels in the western Mediterranean (e.g. García-Barcelona *et al.*, 2010). The average seabird catch rate for this nine-year study covering six longline types was 0.038 seabirds per 1 000 hooks. However, only three of the six monitored gear types caught seabirds: traditional longline, bottom longline and albacore vessels (the average for gear types with seabird catches was 0.049 seabirds per 1 000 hooks). No seabird catches were recorded by the vessels using American-style longline, Japanese bluefin or semi-pelagic gear. Cory's shearwater and yellow-legged gull (*L. michahellis*) were the most commonly caught species. In contrast, Balearic shearwater was only caught in the longline fishery targeting albacore (0.005 seabirds per 1 000 hooks), which uses smaller hooks in comparison with the other fisheries examined.

Fleets from Taiwan Province of China operating in the Mediterranean reported that no seabirds were caught, although data were limited to fewer than 10 000 hooks observed (Yeh *et al.*, 2013). Little is known about seabird catch rates in other longline fleets operating there. Based on available fishing effort data and the data from the Spanish fleet, Anderson *et al.* (2011) estimated that the Mediterranean pelagic longline fleets may be catching 1 000 or more seabirds per year. Tuck *et al.* (2011b) noted the paucity of data from the Mediterranean Sea as a cause for concern. The European Commission has recently taken steps towards an EU Plan of Action-Seabirds to reduce the incidental catch of seabirds in longline and other fisheries, and this may stimulate further study of seabird interactions in the Mediterranean (Anderson *et al.*, 2011).

4.2.2 Eastern Pacific

The EPO is home to the waved albatross, which breeds on the Galapagos Islands. Breeding waved albatross distribution has a high overlap with pelagic longline fishing

effort, with fishing effort estimated to be occurring across 70 percent of its breeding distribution (ACAP, 2008a; Alderman *et al*., 2011; Table 19). In addition, the Northeast Pacific is used by breeding Laysan (*P. immutabilis*) and black-footed (*P. nigripes*) albatrosses, although to a lesser extent than the Northwest Pacific (Table 19). Nevertheless, overall the EPO has a low proportion of global breeding albatross distribution compared with other regions (Table 19).

The EPO is also important for non-breeding albatrosses from New Zealand, such as Buller's, Chatham and Salvin's albatross (*T. bulleri*, *T. eremita* and *T. salvini*), and non-breeding populations of the Chilean black-browed and grey-headed albatrosses, which all migrate to forage in the Humboldt Current. This results in an estimated 12 percent of total global non-breeding albatross time being spent in the EPO, and in albatrosses being distributed much farther north than in other Southern Hemisphere regions. The use of the Southeast Pacific by non-breeding seabirds has been documented by coastal and at-sea observations (e.g. Jehl, 1973; Stahl *et al*., 1998; Robertson *et al*., 2003; Spear, Ainley and Webb, 2003; Goya and Cárdenas, 2003), but more recently has been confirmed by remote tracking data (Figure 32, Table 20). Analysis in 2008 confirmed that the distribution of albatrosses in the Southeast Pacific peaks between April and September, corresponding to periods when seabirds are not constrained to returning to breeding sites and are able to disperse to the rich foraging grounds of the Humboldt Current (ACAP, 2008a). No non-breeding remote tracking data were available for waved albatross but, based on band recoveries and other sightings, it appears to remain in the EPO during the non-breeding season (ACAP, 2008a). In addition, in the Northeast Pacific, there is widespread distribution of non-breeding black-footed albatross from Hawaii, together with lower occurrence of Laysan and short-tailed albatross (Suryan *et al*., 2007).

In the EPO, Japan, the Republic of Korea, Taiwan Province of China and China have the largest number of large longliners (more than 24 m length overall), account for about 90 percent of the longline catches, and operate exclusively on the high seas, principally targeting bigeye and some yellowfin tunas. Other countries, including Spain (mostly targeting swordfish), the United States of America (swordfish and albacore) and French Polynesia (albacore), also record substantial levels of longline fishing effort in the region. Longline fishing effort is predominantly in tropical areas (15°N to 15°S), but extends to 35°N and 40°S (IATTC, 2013b).

Fewer data are available for the Eastern Pacific pelagic longline fleets in comparison with other ocean regions, but these data are summarized below.

4.2.2.1 Southeast Pacific

Data from Taiwan Province of China for the period 2002–07 report rates of seabird catches of > 0.1 seabirds killed per 1 000 hooks (predominantly Buller's albatross) in one 10–15-degree grid square in the high seas offshore of Chile and Peru, and an average for the Galapagos Islands region (defined as the area near 100°W and between 20°S and 5°N) of 0.032 seabirds killed per 1 000 hooks. Across the tropical EPO, this fleet is estimated to kill 270 seabirds per year. Too few data were available for the Southeast Pacific to allow an estimate to be made of interaction rates for this region as a whole (Huang and Yeh, 2011; Huang, Chang and Tai, 2008).

The Spanish industrial swordfish fleet has reported seabird interactions from observer data collected in 1998–2005 (Mejuto *et al*., 2007), with a reported catch rate of 0.04 seabirds per 1 000 hooks across the region as a whole. Using effort data from the period 2002–03 (average of 6.5 million hooks per year), Anderson *et al*. (2011) extrapolated to estimate that the fleet may be catching 260 seabirds per year. However, this estimate must be considered very provisional, and it is now more than ten years old.

Japan reported Southeast Pacific catch rates of more than 0.5 seabirds per 1 000 hooks in some 5 × 5 grid squares for April–December (the period in which observer data were collected; Inoue *et al*., 2011a). This was reported from high seas areas from 15 to 45°S and 75–115°W.

Dai, Xu and Song (2006) reported observer data from 2003 (0.7 percent coverage) from the Chinese fleet that operates between 3 and 17°S, and 96–146°W. A catch rate of 0.02 seabirds killed per 1 000 hooks was reported. Anderson *et al*. (2011) estimated that this would correspond to about 866 seabirds caught per year, based on Chinese fishing effort from 2003 of 43 million hooks per year. China's fishing effort in the EPO was much lower in 2004–2010, but in 2011 regained levels similar to 2003 (IATTC, 2013b).

In Peru's artisanal pelagic longline fisheries, Jahncke, Goya and Guillen (2001) estimated that some 2 370–5 610 albatrosses may be caught in these fisheries each year, based on interviews with fishers. In contrast, observer data from 2005–06 yielded an estimate of 190 seabirds caught per year (Pro Delphinus, 2006).

Farther south, 517–923 seabirds were estimated to be killed per year in the Chilean pelagic longline fishery for swordfish, with an average catch rate of 0.29 seabirds killed per 1 000 hooks (Moreno *et al*., 2007). Albatrosses comprised 79 percent of those caught, with wandering albatross caught most frequently. However, since 2008, Chile has had 100 percent observer coverage in this fleet, and vessels now set lines at night and use line weights and bird streamer lines to deter seabirds, leading to a 95 percent reduction in the number of seabirds being caught (BirdLife Global Seabird Programme, 2010).

4.2.2.2 Northeast Pacific

In the United States Hawaiian fisheries for tuna and swordfish, seabird catches, which are predominantly with black-footed and Laysan albatrosses, decreased by 85–90 percent following the adoption of mandatory mitigation measures, from about 2 300 albatrosses per year in 1999 to fewer than 200 in 2005 (Clemens, 2006; Rivera, 2008). In 2009–2011, captures were slightly higher, with a total catch of 260 seabirds in the deep-set tuna fishery (0.006 seabirds killed per 1 000 hooks) and 68 seabirds in the shallow-set swordfish fishery in 2011 (about 0.045 seabirds killed per 1 000 hooks) (NOAA, 2013b).

Seabird catches of black-footed albatross and Laysan albatross have also been reported by observers from Taiwan Province of China in the northeast Pacific between 15 and 45°N (average regional rate of 0.023 seabirds killed per 1 000 hooks). Based on 2002–07 data (and given relatively low fishing effort by this fleet in the region), the estimated annual seabird catch is 58 seabirds killed per year (Huang, 2011; Huang and Yeh, 2011). However, no northeast Pacific seabird catch data are available from Japan or other fleets that may be operating in the region.

4.2.3 Western and Central Pacific

Analysis of remote tracking data of albatrosses and petrels has highlighted the importance of the WCPO for albatrosses, an area in which pelagic longline fisheries are managed by the WCPFC (ACAP, 2008b; Alderman *et al*., 2011). More than 45 percent of the total global distribution of breeding albatrosses is in the WCPFC area, and the southern WCPFC area overlaps with more than 75 percent of the breeding distribution of almost all populations of albatrosses breeding in New Zealand and Australia, with highest concentrations around southeast Australia and New Zealand including the Tasman Sea (Table 19, Figure 31). In the North Pacific, a high proportion of the breeding and non-breeding distribution of Laysan and black-footed Albatross is within the WCPFC area. Both species are wide-ranging, and spend much of their time in high seas areas (64 percent and 47 percent, respectively), even during the breeding season (ACAP, 2008b). In addition, the short-tailed albatross is almost exclusively distributed

in the WCPFC area throughout the year, albeit with a much more coastal distribution, along shelf waters of the Pacific rim (Suryan *et al*., 2007).

A seabird ecological risk assessment has been undertaken for the WCPO, using fishing effort data, estimates of seabird distribution, and data on vulnerability of species to longline fisheries calculated using data from New Zealand's onboard observer programme (Waugh *et al*., 2012). Seventy species of seabirds were considered, of which 36 have been recorded as caught by longline fisheries. The top ten species considered to be most likely to suffer population effects when exposed to longline fishing activity included large albatross species plus black petrel and Chatham albatross. These were followed by the smaller albatrosses (*Thalassarche* and *Phoebetria* spp.) and the larger petrels from the genera *Procellaria*, *Macronectes* and *Pterodroma*. In the Northern Hemisphere, the highest areas of risk were identified as between 20 and 40°N, especially in the northern autumn and winter. In the Southern Hemisphere, the high-risk areas were identified as between 20 and 50°S, especially to the east and south of the New Zealand mainland, and in the Tasman Sea (Figure 33). A mixture of coastal States with nesting seabird populations in their national waters (New Zealand, Australia and the United States of America), distant-water fishing nations (Japan, Taiwan Province of China), and vessels flagged to Vanuatu contributed 90 percent of the risk to seabird populations (Waugh *et al*., 2012).

The largest pelagic longline fleets in the WCPO that overlap with vulnerable seabird habitat belong to the distant-water vessels of Taiwan Province of China (predominantly targeting yellowfin and albacore), Japan (mostly targeting bigeye and yellowfin), the Republic of Korea (mostly targeting bigeye and yellowfin), Spain (mostly targeting swordfish), and the domestic tuna and billfish fisheries of Australia, New Zealand, Spain and the United States of America. However, in recent years, there has also been an increase in pelagic longline effort by fleets flagged to China, and the domestic South Pacific fleets, and some of these are operating in areas south of 20°S (Williams and Terawasi, 2013), where the fishing grounds probably overlap with albatross distributions.

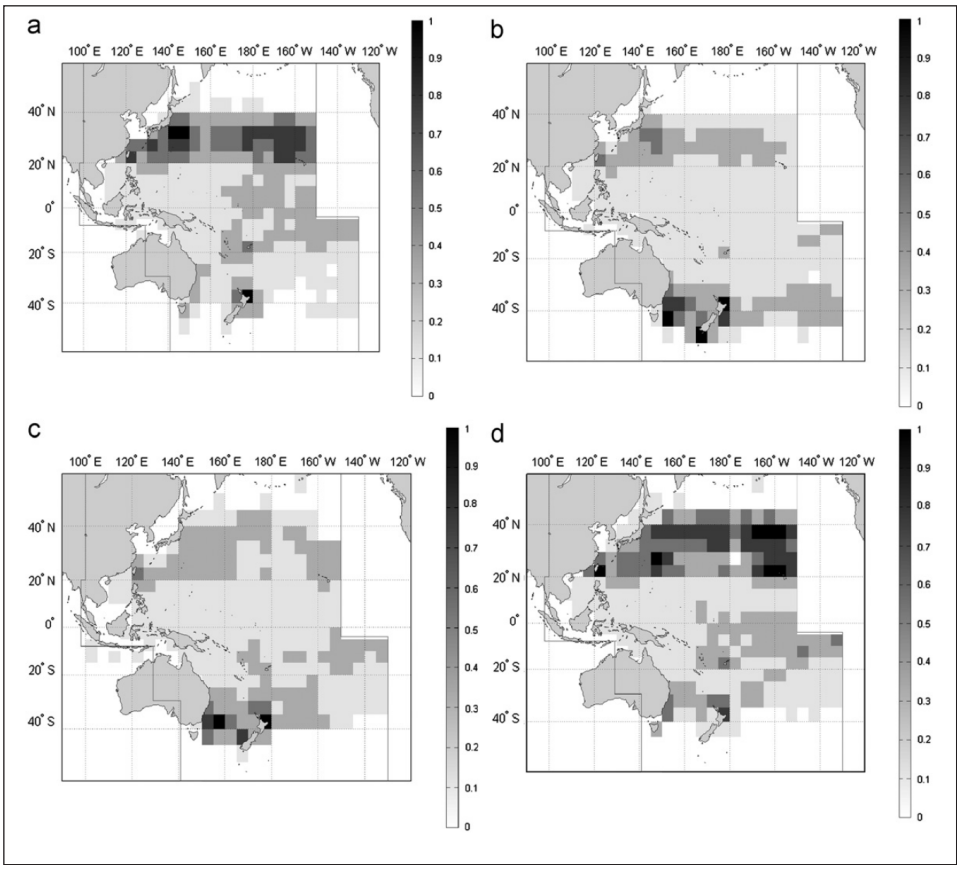
4.2.3.1 Southwest Pacific

As noted in the introduction to Section 4.2, some of the earliest reported seabird catch data from pelagic longline fleets were from Australia and New Zealand. Based on data from the period 1998–2002, Baker and Wise (2005) concluded that the level of seabird catch observed in Australia's Eastern Tuna and Billfish fishery (1 794–4 486 seabirds per year) was most probably unsustainable and threatened the survival of flesh-footed shearwater (*P. carneipes*). However, use of mitigation measures has dramatically reduced seabird catch rates, leading to a recent estimate of 209 seabirds caught per year (Baker and Finley, 2008; Anderson *et al*., 2011).

In the New Zealand pelagic longline fishery, there were 740 (95 percent confidence interval of 547–1 019) seabirds estimated caught in the 2010–11 fishing year for the domestic and charter fleets combined (MPI, 2013), based on an average observer coverage of 26 percent (Abraham and Thompson, 2009). In recent years, it is estimated that the annual number of seabird interactions may have ranged from several hundred to more than a thousand per year (MPI, 2013).

Data are also now available from Taiwan Province of China, Japan and Spain. Data from Taiwan Province of China (2002–07) indicate that seabird catches occurred in the area between 25–35°S and 170°E–165°W between April and September. These data suggest that an average of 0.025 seabirds were killed per 1 000 hooks (360 seabirds killed per year) for the Southwest Pacific as a whole for this six-month period (Huang, Chang and Tai, 2008; Huang, 2011; Huang and Yeh, 2011). However, there were no observer data reported for the South Pacific for the October–March period, so seabird

FIGURE 33
Four seasonal analyses of risk for 70 species of Procellariiform seabirds



Notes: Showing Southern Hemisphere: a) winter (June–August); b) spring (September); c) summer (December-February) and d) autumn (March-May); and risk scores. Black areas indicate highest risk; white areas indicate lowest risk. Risk scores are normalized.
Source: Waugh *et al.* (2012).

catch rates during this season are unknown. Very low levels of seabird catches were reported from the western tropical Pacific.

Observer data from the period 1992–2010 from the Japanese southern bluefin tuna fleet and Japanese research vessels identified seabird catches occurring in the Tasman Sea and areas of the Southwest Pacific (although fewer observer data were available for the latter). The overall seabird catch rate was about 0.05 seabirds killed per 1 000 hooks, lower than the seabird catch rates reported by the southern bluefin tuna fleet in the Atlantic and Indian Oceans. Catches mainly consisted of black-browed, Buller's, shy and wandering albatrosses (Inoue *et al.*, 2011a). No seabird interactions were reported from the area of 0–15°S in the Western Pacific.

Elsewhere, data remain sparse. Spanish pelagic longline vessels in the Western Pacific target swordfish, and initial seabird catch data have been reported from the period 1998–2005, with a rate of 0.032 seabirds caught per 1 000 hooks (Mejuto *et al.*, 2007). However, further data will be required on both seabird catch rates and fishing effort in order to estimate the number of seabirds currently being killed per year in this fishery.

Data provided in Dai and Zhu (2008) indicated that China's fishing effort in the WCPFC convention area was concentrated between 15°N and 20°S. However, China's longline fleet has grown rapidly in recent years (86 vessels in 2007 increasing to 275 vessels by 2011 [Anon., 2012]). No seabird catch data are currently available. In addition, few seabird catch data are currently available from the fleet of the Republic

of Korea for the WCPFC convention area (Kim *et al*., 2010); however, this fleet fishes in tropical areas where seabird catch rates are considered low.

4.2.3.2 Northwest Pacific

Information from the United States Hawaii-based longline fleet has been covered in Section 4.2.2, and total seabird catches have been estimated to be 220 and 260 seabirds killed per year in 2010 and 2011, respectively (NOAA, 2013b).

Data from Taiwan Province of China covering the entire Pacific indicate that seabird catch rates (and total number of seabirds killed) was highest in the Northwest Pacific area between 25–45°N and 165°E–160°W between October and March. These data indicate that, on average, 0.22 seabirds are killed per 1 000 hooks for the Northwest Pacific as a whole in October–March, with black-footed albatross and Laysan albatross particularly affected. This would equate to an average of 1 775 seabirds killed per year between October and March (Huang, Chang and Tai, 2008; Huang, 2011; Huang and Yeh, 2011). No data were available for this region for the period April–September.

No comprehensive seabird catch data have been reported by Japan for the Northwest Pacific. However, experimental testing of seabird mitigation techniques, conducted in a one-month trial in December 2010 reported seabird catch rates of 0.11 seabirds killed per 1 000 hooks when using single streamer lines, and 0.06 seabirds killed per 1 000 hooks when using paired streamer lines, with Laysan albatross particularly affected (Sato *et al*., 2013a).

4.2.4 Indian Ocean

Seven of the 18 species of Southern Hemisphere albatrosses have breeding colonies on islands in the southern Indian Ocean, and all but one of the 18 (Atlantic yellow-nosed albatross) forage in the Indian Ocean at some stage in their life cycle. Overall, an estimated 14 percent of global breeding albatross and giant-petrel distribution, and 21 percent of non-breeding distribution, occurs in the Indian Ocean (Tables 19 and 20). The area is particularly important for the endemic Amsterdam albatross and Indian yellow-nosed albatross (*T. carteri*), as well as wandering albatross (74 percent of global breeding pairs), sooty albatross (*P. fusca*, 39 percent of global breeding pairs), light-mantled albatross (32 percent of global breeding pairs), grey-headed albatross (20 percent of global breeding pairs), and the non-breeding distribution of shy albatross, which breeds in Tasmania and forages in the area of overlap between the areas managed by the IOTC and WCPFC. The Indian Ocean is also important for northern and southern giant petrel (26 percent and 30 percent of global breeding pairs, respectively) (IOTC, 2012c).

Albatrosses exploit the same productive areas as pelagic longline vessels in the Indian Ocean (Weimerskirch, 1998; Tuck *et al*., 2001; Baker *et al*., 2007). Weimerskirch and Jouventin (1987) deduced that reduced survival of female and juvenile wandering albatrosses compared with males was probably the result of greater overlap with pelagic longline fishing. Analysis of albatross remote-tracking data has identified that albatrosses breeding on Southern Indian Ocean islands spent 70–100 percent of their foraging time in areas overlapping with IOTC longline fishing effort, and confirmed that Amsterdam, Indian yellow-nosed, wandering, shy, grey-headed and sooty albatrosses and white-chinned petrels had a high overlap with pelagic longline fishing. However, data on distribution during the non-breeding season was lacking for many species, including black-browed albatrosses and white-capped albatrosses (known to be among the species most frequently caught by longline fisheries) (ACAP, 2007; IOTC; 2012c).

In the period 2009–2011, new tracking data were presented to the IOTC that filled gaps in earlier analyses, particularly for Amsterdam, sooty and Indian yellow-nosed albatrosses and for distributions of juveniles (Delord and Weimerskirch, 2009, 2010,

2011). The additional tracking data identified that extensive overlap exists between IOTC pelagic longliners and albatrosses and petrels. It also identified that core breeding distribution is concentrated below 30°S, but that non-breeding and juvenile seabirds forage in areas up to 20°S. A concentration of non-breeding albatross in the Southeast Indian Ocean was also identified (Delord and Weimerskirch, 2011).

Seabird catches by longline fisheries in the Indian Ocean have been identified as being linked to decreased adult survival, juvenile recruitment and population declines (Weimerskirch, Brothers and Jouventin, 1997; Barbraud *et al*., 2008; Delord and Weimerskirch, 2011). Nel *et al*. (2003) found that annual survival rates of breeding wandering albatross at Marion Island were negatively correlated with Japanese pelagic longline fishing effort in the southern Indian Ocean for the period 1984–2000.

Some of the earliest seabird catch data from the Indian Ocean come from South African national waters. In those waters, foreign-flagged vessels are required to have 100 percent observer coverage, and seabird interactions are monitored closely. In 2008, 141 seabirds were killed by this fleet in the Indian Ocean (P.G. Ryan, unpublished data). As discussed in Section 4.2.1.1, seabird catches in the South African domestic fleet are estimated to be about 299 seabirds per year across the Atlantic and Indian Oceans combined based on data from 1998–2005 (Petersen, 2008).

In the Australian Western Tuna and Billfish Fishery, Baker *et al*. (2007) estimate that fewer than 50 seabirds are killed per year, based on reduced fishing effort and increased use of seabird mitigation measures (use of night setting and streamer lines, with no offal discharge during setting and hauling). However, observer coverage is low (< 5 percent). Of the seabirds that are killed, most are flesh-footed shearwaters, with a small number of albatrosses.

The largest Indian Ocean fleets operating in areas overlapping with albatross distribution are those from Taiwan Province of China and Japan. Data from the former for 2004–07 indicate seabird catch rates of 0.0158 seabirds per 1 000 hooks in the southern Indian Ocean and 0.0002 seabirds per 1 000 hooks in tropical areas in 2004–08. It was estimated that 311–715 seabirds were caught annually in 2004–08 (Huang and Liu, 2010).

Data from the Japanese observer programme in the Indian Ocean include data from almost 15 million hooks observed in the period 1992–2009. These data identify occurrences of seabird catches in the Southwest Indian Ocean in April–September (few data available for October–March), and in the Southeast Indian Ocean in June–December (few data available for January–June). Grey-headed albatross was the species most frequently caught, followed by black-browed albatross and Indian yellow-nosed albatross, as well as shy and wandering albatross and white-chinned petrel (Inoue *et al*., 2011b).

For the Spanish distant-water fleet, the available data on seabird catches come from experimental cruises in 2005 (Ariz *et al*., 2006). Only three seabirds were observed caught, equating to an interaction rate of 0.0056 seabirds killed per 1 000 hooks between 25 and 35°S, much lower than rates observed by South African observers on vessels in similar areas. Anderson *et al*. (2011) extrapolated this to the fleet as a whole (6.5 million hooks set in 2006), to give an estimate of 37 seabirds caught in 2006. However, the experimental cruises were testing the effect of different fishing methods on sea turtles, including different circle hook designs and use of coloured bait, and this may have concomitantly reduced seabird catch rates (Ariz *et al*., 2006).

In 2007, data provided in Xu, Dia and Song (2007) indicate that China's fishing effort in the IOTC was concentrated north of 25°S. However, in 2006–2010, China deployed an average of 2.3 million hooks per year south of 25°S (IOTC, 2012d). There are no seabird catch data reported for this fleet. In addition, no seabird catch data are available from the fleet of the Republic of Korea in the IOTC region. However, Anderson *et al*. (2011) estimated that if the fleet had a similar seabird catch rate to the

Taiwan Province of China fleet, then it would be catching about 97 seabirds per year. This estimate was considered very provisional given that no direct data are available.

4.3 MANAGEMENT MEASURES AND THEIR EFFECTIVENESS

4.3.1 Introduction

Løkkeborg (2008, 2011) defines a seabird mitigation measure as a modification to gear design or a fishing operation that reduces the likelihood of catching seabirds. Seabird mitigation measures for longline fishing can be divided into four main categories:

- avoiding fishing in areas and at times when seabird interactions are most likely and most intense (time/area closures, night setting);
- limiting seabird access to baited hooks (underwater setting devices, weighted lines, side-setting);
- deterring seabirds from taking baited hooks (streamer lines);
- reducing the attractiveness or visibility of the baited hooks (retention of or strategic dumping of offal, artificial baits, blue-dyed bait) (FAO, 2009).

Some of the highest seabird catch rates globally have been recorded in the pelagic longline fisheries of the Southern Hemisphere (Stagi *et al*., 1998; Bugoni *et al*., 2008; Jiménez, Domingo and Brazeiro, 2009). Owing to the nature of the gear configuration, pelagic longlines are arguably one of the most challenging gear types in which to mitigate seabird interactions. Unlike demersal longline gear, which readily accepts substantial weight added to increase sink rates, pelagic longline gear needs to be positioned at a specific depth in the water column to maximize target catch rates, so the amount of weight that can be added is much less than in demersal longline fisheries. In addition, pelagic gear has significantly longer branch lines and more floats deployed along the average longline, which creates more opportunity for entanglements with streamer lines (Sato *et al*., 2013a), which, along with line weighting and night setting, is one of the most commonly prescribed seabird mitigation measures.

In recent years, considerable progress has been made in the refinement of existing mitigation measures, and when used in combination they have been demonstrated to reduce seabird catches significantly (e.g. Melvin, Guy and Read, 2013; Jiménez *et al*., 2013). These are discussed in Section 4.3.2. The uptake of some of these mitigation techniques by management organizations in then described in Section 4.3.3.

Mitigation measures to reduce seabird interactions can cause operational challenges for fishers. It is critically important that in the design and testing phase of new measures (or combinations of measures) operational issues such as slowing down line setting and hauling operations and gear entanglements are addressed to minimize the impact on the efficiency of fishing operations.

4.3.2 Mitigation techniques supporting management measures

This section summarizes research aimed at providing scientifically robust evidence to support the adoption of conservation measures to reduce seabird catches by longline fisheries. It should be noted that much of the research summarized here assumes that seabird attack rates on baited hooks during line setting operations can serve as a proxy for seabird mortality to assess the effectiveness of mitigation techniques (Melvin, Guy and Read, 2013; Domingo, Jiménez and Abreu, 2011; Gianuca *et al*., 2011; Sato *et al*., 2013a, 2013b). This assumption requires the classification and systematic recording of seabird activity during line setting. The most commonly used classification divides interactions into primary and secondary attacks. A primary attack is an unambiguous attempt by an individual seabird to take bait from a hook –typically a dive or plunge directly over a sinking hook. A secondary attack occurs when another seabird or a group of seabirds attempts to steal bait or a baited hook from a seabird that is making a primary attack (Melvin, Guy and Read, 2013). This includes baited hooks that have

been returned to the surface by diving species (e.g. white-chinned petrels), thus making the bait available to seabirds on the surface, including "great" albatrosses, which are unable to dive.

4.3.2.1 Streamer (tori) lines

Pioneering work by the Japanese tuna fisheries in the Southern Ocean in the early 1990s led to the development of streamer lines (sometimes referred to as "bird-scaring" or "tori" lines) to scare seabirds away from baited hooks as they enter the water, both to reduce bait loss and seabird interactions (Brothers 1991). Since that time, they have become one of the most commonly prescribed mitigation measures to reduce seabird catches by longline fisheries (Melvin *et al*., 2004; Løkkeborg, 2011).

There has long been debate about the optimal design to maximize the effectiveness of streamer lines in reducing seabird catches by longline fisheries, including how to minimize impact on fishing operations, which is a particular challenge in pelagic longline fisheries. In 1991, the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) was the first fisheries management body to prescribe the mandatory use of streamer lines to reduce seabird catches by demersal longline fisheries. These streamer lines were required to have "streamers that reach the sea surface in the absence of wind and swell" (CCAMLR Conservation Measure 25-02 [CCAMLR, 2012]).

However, until recently, research to identify optimal streamer line design and configuration was lacking (Melvin *et al*., 2004). This lack of research and scientific evidence to support the adoption of the most effective streamer line configurations has led to a degree of uncertainty in the most effective design for pelagic longline fisheries and the terminology used to describe different styles of streamer lines. For several years, a commonly held view was that the "conventional" line (with long streamers, *sensu* CCAMLR Conservation Measure 25–02 [CCAMLR, 2012]) was the most effective and appropriate for pelagic longlines. However, owing to the nature of pelagic longline configurations, i.e. long branch lines and multiple buoys deployed throughout each setting operation, long streamers can entangle with fishing gear, and it is also likely that streamers will tangle around traditional towed devices, such as buoys (Domingo *et al*., 2013; Melvin, Guy and Read, 2013; BirdLife International and ACAP, 2012a).

The aerial extent of a streamer is the critical factor in determining its effectiveness in protecting baited hooks as they sink to a depth below which the majority of seabirds can dive to access the bait, which is conventionally considered to be 10 m depth (e.g. Melvin, Guy and Read, 2013). The aerial extent of a streamer is influenced by a number of factors including:

- the forward motion of the vessel;
- the attachment height and position on board the vessel (usually at the stern);
- the amount of drag created by the towed device at the distal/seaward end of the "backbone" of the streamer lines;
- the weight of the materials used to configure the streamer line.

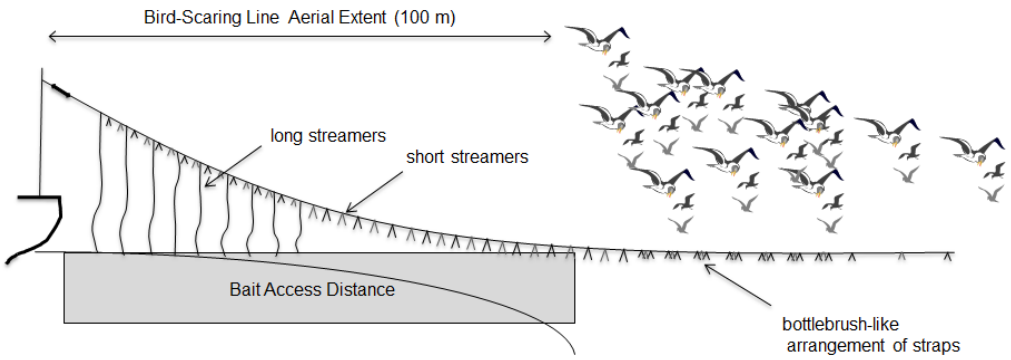
Another important factor in determining the effectiveness of streamer lines is their position relative to the baited hooks as they sink below the sea surface. Streamer lines are less efficient under conditions of strong crosswinds, as these can blow the streamers to the side of the longline, leaving baited hooks exposed to seabirds (Løkkeborg, 1998). Reduced efficiency under crosswind conditions may partly be counteracted by attaching the streamer line to the windward side of the vessel or by using paired streamer lines.

Streamer lines are also likely to be less efficient in reducing catches of diving seabirds, particularly in pelagic fisheries, as seabirds may still reach baited hooks beyond the aerial extent of the streamer line (or lines) (e.g. Jiménez *et al*., 2012; Melvin,

Guy and Read, 2013). This deficiency may be significantly reduced by using weighted branch lines in combination with streamer lines (see Dietrich, Melvin and Conquest, 2008; Gianuca *et al*., 2011; Domingo *et al*., 2013; Melvin, Guy and Read, 2013, and Section 4.3.2.2).

Innovation in the design and materials used for streamer lines in recent years, often undertaken by fishers, has led to the testing and adoption of new configurations for pelagic longline fisheries. A "Japanese light" (hereafter called light) streamer line (with short streamers; Yokota, Minami and Kiyota, 2008) and a hybrid streamer line (with long and short streamers, Melvin, Guy and Read, 2013; Figure 34) have been developed for pelagic longlining in order to prevent entanglement of the streamer with the longline gear. A generic schematic of various streamer line designs is provided in Sato *et al*. (2013b).

FIGURE 34
An example of a hybrid streamer line, as used in research with the Japanese joint venture research in South African national waters



Source: Melvin, Guy and Read (2014).

The only study testing the relative effectiveness of paired versus single streamer lines at reducing attack rates of seabirds on baited hooks was conducted over 25 sets in the North Pacific in 2010–11 (Sato *et al*., 2013a). Two types of streamer lines were used – a light streamer line and a hybrid streamer line – and the length of both streamer lines was maintained at 70–80 m throughout all operations. In a fishing area dominated by non-diving seabirds, 88 percent of the attacks were by Laysan albatross. Overall, the mean number of albatross attacks on single streamer lines was 25.69 per 1 000 hooks whereas on paired streamer lines it was 12.29 attacks per 1 000 hooks. Relative to single streamer lines, paired streamer lines reduced primary attacks by 47.8 percent. Paired streamer lines also resulted in fewer attacks within 75 m of the stern (where hooks would be expected to be closer to the surface). Although this study suggests superior performance of paired streamer lines compared with single lines, further testing with a larger sample size is required, particularly for the Southern Ocean where a broader range of diving seabird species is present.

Recent research in the western North Pacific examined the effectiveness of light streamer lines in reducing seabird catches (Sato *et al*., 2013b). In the first stage, logbook data from 567 sets on 20 Japanese offshore surface commercial longliners were used to compare the effectiveness of two different types of single streamer lines (light streamer [1 m] and long streamer [up to 7 m]) and of two different colours (yellow and red) of light streamer. The aerial extent reported for all streamer lines was 50–60 m throughout all operations. No significant differences in the number of seabird catches

were identified. In the second phase, 24 sets from a research vessel tested the effects of light, hybrid and modified streamer types of 90–100 m aerial extent. No significant differences in primary attack rates for Laysan albatross, which constituted 81 percent of recorded attacks, were found.

Recent research in regions with different seabird assemblages and vessel characteristics has greatly improved understanding of the optimal design and performance of streamer lines. This has underpinned the formulation by the Agreement for the Conservation of Albatrosses and Petrels (ACAP) of best-practice advice for streamer lines as follows (ACAP, 2011):

- Vessels > 35 m: Simultaneous use of two bird streamer lines, one on each side of the sinking longline, with an aerial extent of 100 m. If large vessels use only one streamer line, the streamer line should be deployed to windward of sinking baits.
- Vessels < 35 m: A single streamer line using either long and short streamers, or short streamers only, has been found to be effective on smaller vessels. A minimum aerial extent of 75 m is recommended.

4.3.2.2 Weighted branch lines

It is widely recognized that increasing the sink rate of the baited hook is currently the most effective means of reducing seabird catches in pelagic longline fisheries. However, using line weighting to reduce seabird mortality in pelagic longline fisheries is more complicated than in demersal longline fisheries owing to secondary interactions near the surface. Secondary interactions rarely, if ever, occur in demersal longline fisheries, which target fish species that forage on or near the sea bed, because branch lines are extremely short (< 0.6 m) and the mainline is heavy or anchored to the bottom. In contrast, pelagic branch lines can be 15–40 m in length and are often lightweight (BirdLife International/ACAP, 2012b).

A range of bait characteristics can affect the sink rate of baited hooks. These include the size of the bait, whether the bait is alive or dead, and whether it is frozen or thawed. However, there are two aspects of branch line weighting that are critically important to achieving faster sink rates: the length of the leader (the distance between the lead weight and baited hook), and the mass of the added weight. Leader length is the main determinant of the initial phase of the sink profile, whereas the weight of the attached lead is the main determinant of the final phase of the sink profile. The initial phase starts when the baited hook lands in the water and ends when the leader becomes taut. In this phase, the lead weight sinks much faster than the baited hook. The final phase of the sink rate occurs when the slack in the leader is taken up and the baited hook comes under maximum load of the lead weight (Figure 35). To reduce the availability of baits in all depths of the water column, it is important to increase both the initial and final phases of sink profiles. This is achieved by using heavier weights closer to hooks (Robertson *et al*., 2010; Jiménez *et al*., 2013; Gianuca *et al*., 2013; Robertson, Candy and Hall, forthcoming).

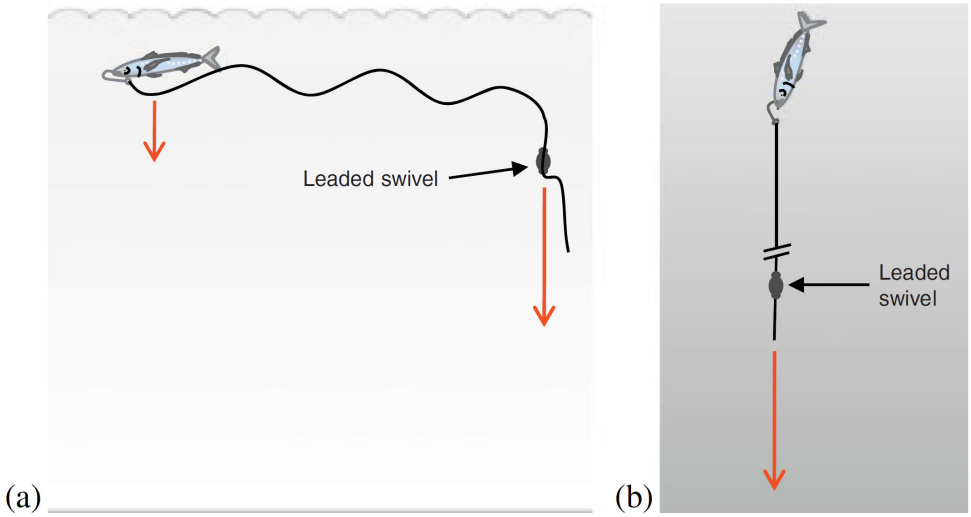
Traditionally, line weighting in pelagic longline fisheries has involved crimping lead swivels to branch lines/snoods. Many pelagic longline fisheries around the world have adopted this practice to deliver hooks to target fishing depths as efficiently as possible and as part of regulatory measures to reduce seabird interactions. However, there is reluctance by many fleets to use leaded swivels owing to safety concerns because weighted swivels may cause serious injuries to crew when they recoil in the event of a line breakage (Løkkeborg, 2011; see Section 4.4.5 for discussion of how sliding and lumo leads can reduce this risk). The ACAP best practice line weighting advice (ACAP 2011) recommends the following as a minimum standard for branch line weighting configurations:

- greater than 45 g attached within 1 m of the hook; or

- greater than 60 g attached within 3.5 m of the hook; or
- greater than 98 g weight attached within 4 m of the hook.

Positioning weights farther than 4 m from the hook is not recommended.

FIGURE 35
The source of the difference between the initial (a) and final (b) stages of line sinking



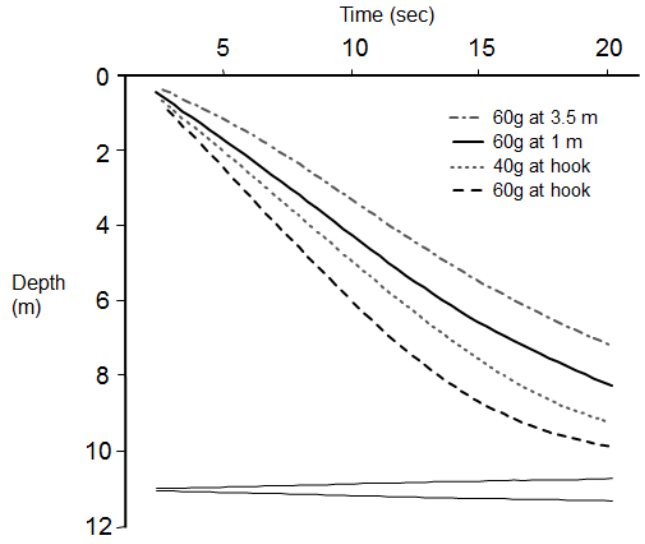
Notes: Bait sinks slower than the leaded swivel until the length of line connecting them becomes taut.
Source: Robertson et al. (2010).

A series of trials conducted in the Australian east coast tuna fishery in 2010–11 compared the impact on target catch rate of two previously untested line weighting regimes versus the industry standard (Robertson, Candy and Hall, forthcoming). The first compared the industry standard line weighting regime (60 g at 3.5 m) with a 120 g sliding lead (see Section 4.4.5) positioned 2 m from the hook. The second experiment compared the industry standard line weighting regime with a 40 g sliding lead placed at the hook. Sink rates for these two weight regimes plus 60 g at the hook and 60 g at 1 m from the hook were also tested at sea under controlled conditions (not in a standard fishing operation) and compared with data for a range of other weight regimes collected from static trials in a pool.

Branch lines with 60 g at the hook sank the fastest, followed by 40 g at the hook, 60 g at 1 m from the hook, and 60 g at 3.5 m from the hook (Figure 36). The differences between all four profiles were statistically significant throughout the entire depth range. The results to 5 m were 9.7 s (0.51 m/s) and 13.6 s (0.37 m/s) for the 40 g hook lead and 60 g at 3.5 m, respectively. There were no statistical differences in catch rates of target and non-target fish between industry standard branch lines and branch lines with both 120 g leads at 2 m and those with 40 g leads at the hook (Robertson, Candy and Hall, forthcoming).

Trials conducted during ten sets on a research vessel operating on the shelf slope off Uruguay from 2009 to 2012 assessed seabird attack rates with branch lines set during daylight hours with two different weight regimes: 75 g at 4.5 m from the hook, and a 65 g sliding lead at 1 m from the hook (Jiménez *et al.*, 2013). The seabird attack rate was 59 percent lower and the seabird catch rate was 50 percent lower on the branch lines having 65 g weights within 1 m of the hook compared with branch lines with 75 g weights placed 4.5 m from the hook. The capture rate of target fish species was assessed during an additional 43 fishing sets and no statistical differences were found for the

FIGURE 36
Mean sink profiles of the line weighting regimes (60 g at 3.5 m and 40 g lead at the hook) compared in the fish catch experiment



Notes: Profiles for the 60 g lead at the hook and 60 g at 1 m are also shown because they have potential relevance to the fishery. The 95 percent confidence intervals are shown as horizontal bars at the bottom of the figure. If the difference between any two average profiles for a given time exceeds the difference between the upper and lower limits of the confidence interval, then the difference is statistically significant. Thirty replicates were tested for each treatment.
Source: Robertson, Candy and Hall (forthcoming).

main target species between the two branch line treatments. Sink profiles showed that baited hooks on the experimental branch line reached at least 4 m depth within the average streamer line aerial coverage (about 65–70 m). In contrast, using standard branch lines, on average, the baited hooks were located at 2 m depth beyond the area covered by the streamer line. This research provides further evidence that reducing the hook–weight distance in the pelagic longline branch lines reduces seabird attacks and catch rates without reducing catch rates of target species (Jiménez *et al*., 2013). There is good evidence to suggest that, if such line weighting regimes were used in combination with streamer lines, a considerably greater than 50 percent reduction in seabird catch could be achieved (Melvin, Guy and Read, 2013; Melvin, Guy and Read, 2014). Night setting would probably further reduce seabird catch levels (Gianuca *et al*., 2011).

In 2012, trials were conducted in the Brazilian domestic pelagic longline fleet on 92 sets over 9 cruises, to compare the catch rate of target species on lines with leaded swivels placed at 2 m and 5.5 m from the hooks. No significant difference was detected in catch rates between the two weight regimes, the mean difference being less than one fish per 1 000 hooks (Gianuca *et al*., 2013).

There is a growing body of evidence from several regions of the world and from coastal areas such as Brazil (Gianuca *et al*., 2013), Hawaii (Gilman, Kobayashi and Chaloupka, 2008), and Uruguay (Jiménez *et al*., 2013), as well as from "high seas" vessels off South Africa (Melvin, Guy and Read, 2013) that placing weights close to the hook significantly reduces seabird catches, and does not reduce the catch of target species.

4.3.2.3 Night setting

Most seabirds, including albatrosses, are visual feeders and forage during daylight hours. Therefore, setting longlines at night (i.e. between the hours of nautical dusk and dawn) can reduce the number of seabirds attacking baited hooks (Løkkeborg, 2011). Night setting is arguably the most widely tested seabird mitigation measure in both demersal longline fisheries (e.g. Cherel, Weimerskirch and Duhamel, 1996;

Weimerskirch, Capdeville and Duhamel, 2000; Reid *et al*., 2004) and pelagic longline fisheries (e.g. Brothers, Gales and Reid, 1999; Klaer and Polacheck, 1998), both of which have demonstrated considerable reductions in mortality rates for longlines set at night, particularly for albatrosses. Off the east coast of Australia, where shearwaters predominate, night setting alone in the tuna longline fishery markedly reduced seabird catch rates compared with daytime sets (Baker and Wise, 2005).

Analyses of observer data from Japanese tuna longline vessels operating in the Australian Fishing Zone from 1991 to 1995 showed an 85 percent reduction in seabird catches for hooks set at night. Moreover, seabirds were 3.6 times more likely to be caught on night-set hooks set in bright moonlit conditions than on those sets when there was no moonlight (Brothers, Gales and Reid, 1999). In the South African joint venture pelagic longline fishery, seabird interaction rates were 4.6 times higher during daylight hours (2 seabirds per 1 000 hooks; 52 seabirds in total) than at night (0.439 seabirds per 1 000 hooks; 28 seabirds in total), and night catch rates near the full moon doubled (Melvin, Guy and Read, 2013).

Despite these promising results, all Procellaria petrels (i.e. including white-chinned and black petrel) and some Puffinus shearwaters forage actively both day and night (Shealer, 2001; ACAP, 2009b) and therefore remain vulnerable to night setting. Seabird assemblage composition may thus have substantial impact on the effectiveness of night setting (Murray *et al*., 1993). In the toothfish fishery around the Prince Edward and Marion Islands, albatross catches were reduced by a factor of ten using night setting, but catches of white-chinned petrels were only halved (Ryan and Watkins, 2002). Similarly, Melvin, Guy and Read (2013) found that mortality rates of albatross species in night setting were much lower than in daytime setting, whereas mortality rates of diving seabirds such as white-chinned petrel were slightly lower than in daytime setting. Off the east coast of Australia, where shearwaters predominate, night setting alone is less effective, although seabird catch rates in the tuna longline fishery are still lower for night sets than for daytime sets (Baker and Wise, 2005). As these examples illustrate, the effectiveness of night setting must be considered in relation to the composition of the seabird assemblages present in the area of interest. Moreover, in many, if not most, circumstances, it will be essential to apply night setting in conjunction with appropriate combinations of line weighting and/or streamer lines.

4.3.2.4 Side-setting

Side-setting refers to setting longline gear from the side of the vessel rather than the conventional position at the stern. In theory, when side-setting is practised the crew throw baited hooks forward and close to the side of the vessel hull where seabirds, such as albatrosses, are unable or unwilling to pursue them. Ideally, by the time the hooks pass the stern, they have sunk beyond the reach of seabirds (Gilman, Brothers and Kobayashi, 2007).

Short-term at-sea experiments in the domestic Hawaiian pelagic longline fleet have demonstrated that the deployment of side-setting and 60 g weights reduced seabird catches to zero (Gilman, Brothers and Kobayashi, 2007). Analysis of a multiyear data set before and after adoption of mitigation regulations in this fishery concluded that side-setting used in combination with line weighting and a bird curtain (a short streamer line that hangs from a rigid frame on the stern quarter of the vessel to protect the baited hooks as they pass the stern of the vessel) had the potential to achieve long-term reductions of seabird catches to negligible levels (Gilman, Kobayashi and Chaloupka, 2008). The conversion of smaller coastal vessels, such as those that operate in the domestic Hawaii fleet, is a simpler process than converting much larger high seas vessels that have dedicated setting structures (often including enclosed setting windows that are integral to the structure of the stern of the vessel).

The effectiveness of side-setting as a mitigation measures remains untested in Southern Hemisphere fisheries that interact with complex assemblages of diving seabirds. Therefore, despite the promising results in Hawaii, targeted research is required before side-setting can be considered for more widespread use.

4.3.2.5 Blue-dyed bait

The application of blue dye to fishing baits is thought to make the baits less visible by reducing the contrast between the bait and the sea colour, thereby making it more difficult for seabirds to detect the bait when foraging from above (Gilman, Kobayashi and Chaloupka, 2008b). The effectiveness of blue-dyed bait at reducing seabird catches has varied considerably among different trials. Some trials have shown significant reductions in attack rates on hooks baited with blue-dyed squid (Boggs, 2001), and on seabird catch rates (Minami and Kiyota, 2001). Other studies indicate it is less effective than other measures, including side-setting used in combination with line weighing and a bird curtain (Gilman, Brothers and Kobayashi, 2007).

Cocking *et al*. (2008) measured and modelled the spectral profiles of blue-dyed fish and squid to test the assumption that dyed baits are less visible and hence less attractive to seabirds. Results showed that no baits were perfectly cryptic against the background ocean, and only blue-dyed squid were relatively cryptic both in terms of chromatic and achromatic contrasts. At-sea trials under controlled (non-commercial fishing) conditions showed a 68 percent reduction in interactions with blue-dyed squid compared with non-dyed squid. Results suggest that blue-dyed fish were far less effective than squid at reducing seabird attack. The study also showed an increased rate of attacks on blue-dyed fish baits over the course of the trial. This suggests that, over time, seabirds may develop a degree of habituation to blue-dyed fish that enables them to identify and attack dyed baits.

Generally, blue-dyed squid is considered to have potential to reduce seabird catches when used in combination with other measures (e.g. line weighting, side-setting [in Hawaii], streamer lines and/or night setting). However, long-term trials are needed to understand the complex relationships between seabird behaviour, bait colour, environmental and operational factors. Moreover, the adoption of blue-dyed bait as a widespread mitigation measure is hampered by operational challenges. Comparative trials in Hawaii concluded that blue-dyed bait was impractical owing to the time needed to fully thaw and dye baits. However, it was considered to be potentially more feasible if commercially available blue-dyed baits became available (Gilman, Brothers and Kobayashi, 2007c).

While blue-dyed squid has been shown to be a potentially more promising mitigation measure than blue-dyed fish, and despite some promising results in the early trials, blue-dyed bait has failed to achieve widespread or long-term adoption as a mitigation measure. Until more extensive studies are completed and a product is commercially available, it is unlikely to be a mitigation measure that can be effectively used, even in combination with other proven mitigation measures.

4.3.2.6 Line shooters

A line shooter is a hydraulically operated device designed to deploy the mainline at a speed faster than the vessel's forward motion, which removes tension from the longline and in turn alters the underwater shape of the mainline and its target depth (Mizuno, Okazaki and Miyabe, 1998; Robertson, Candy and Wienecke, 2010). It has been demonstrated that variation in tension on the mainline will affect the sink rates of baited hooks and therefore the risks to seabirds (Robertson, Candy and Wienecke, 2010).

A controlled experiment in the Australian tuna fishery revealed that setting a loose mainline with a line shooter resulted in slower sink rates of baited hooks in surface

waters compared with baited hooks attached to mainline set without a line shooter (Robertson, Candy and Wienecke, 2010). It was considered that the most likely reason for this was that propeller turbulence slowed the sink rates of loose mainlines which, in turn, slowed the sink rates of baited hooks (Robertson, Candy and Wienecke, 2010). Although this experiment was conducted in the absence of observations of seabird behaviour and catches, it presents compelling evidence that line shooters should not be considered as a seabird mitigation measure (BirdLife International/ACAP, 2012a). Further research is required as part of a standard fishing operation to verify these results in the light of observed seabird interactions and on vessels with different operational configurations. For example, longliners that have the line shooter located forward of the stern corner and, thereby, unlike the Australian vessels observed by Robertson, Candy and Wienecke (2010), avoid setting the line in the propeller wash could show different results.

4.3.2.7 Bait casters

A bait-casting machine is a spring-loaded mechanical arm designed to aid the deployment of baited hooks during pelagic longline setting. A "gyrocast" bait caster was built in the mid-1990s that allowed directional control, improved fishing efficiency, and if used correctly also had the potential to reduce the risk of seabird catches by placing baited hooks within the protected area of the streamer line (or lines) (Brothers, Cooper and Løkkeborg, 1999).

However, later models of bait casters did not incorporate the key features necessary to reduce seabird catches, in particular distance control. Therefore, while bait casters are widely used, they have lost the characteristics that made them a potentially effective seabird mitigation measure. There is currently inadequate evidence to quantify the effectiveness of their contribution to seabird catch reduction (BirdLife International/ACAP 2012a).

4.3.2.8 Line setting chute

A line setting chute is a stern-mounted tube through which the baited hooks are set. This device delivers baited hooks underwater, thereby reducing the time they remain close to the surface and visible and accessible to seabirds. Trials in Hawaii demonstrated a significant reduction in seabird catches as well as a reduction in bait loss (Gilman, Boggs and Brothers, 2003). However, the technology has some operational challenges that have not been overcome, and despite the chute being under development for more than a decade, it remains unused in commercial operations.

4.3.2.9 Combinations of mitigation measures

Until recently, there was a lack of robust scientific evaluation of the effectiveness of deploying a combination of mitigation measures. However, there is a growing body of evidence that smaller longliners, operating in the Southern Hemisphere where there are complex seabird assemblages of diving seabirds (e.g. white-chinned petrels), can reduce seabird catches to negligible levels with a single hybrid or light streamer line used in combination with line weighting and night setting (Gianuca *et al*., 2013; Domingo *et al*., 2013). In addition, for high seas longliners operating around southern Africa, a combination of paired streamer lines and line weighting significantly reduced seabird attack rates and catches (Melvin, Guy and Read, 2013; Melvin, Guy and Read, 2014). This section reviews recent research that investigates successful combinations of mitigation measures.

Experimental research off Uruguay was conducted to compare the effectiveness of a single hybrid streamer line with line weighting (domestic fisheries regulations mandate 75 g weights at 4.5 m) with a control treatment of no streamer line (plus regulated line weighting) (Domingo *et al*., 2013). This resulted in a total of 50 seabird mortalities,

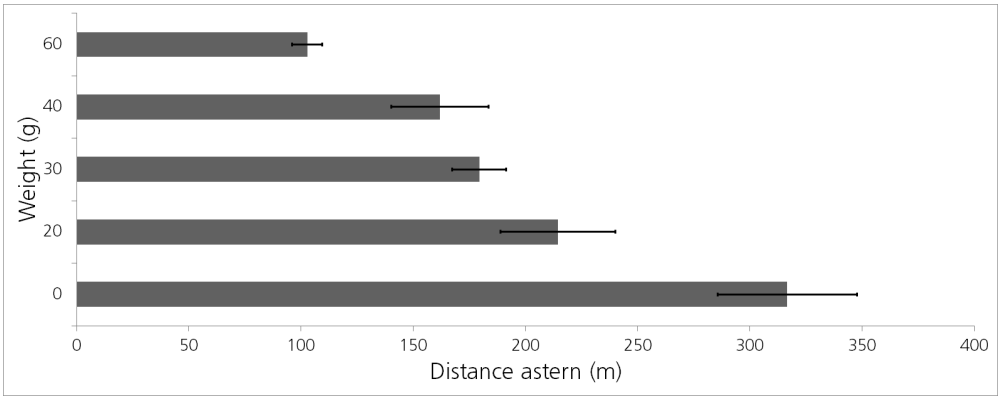
43 of which were captured on the control treatment (n=49 sets), and seven on the single hybrid streamer line treatment (n=51 sets). However, 5 of the 7 seabirds killed on the streamer line treatment occurred when the streamer line broke after becoming entangled, which occurred on 48 percent of the sets. A further 26 sets were then conducted with a davit-mounted device to alter the position of the single streamer line in relation to the wind and vessel direction, and with a towed device of plastic straps incorporated into the seaward end of the backbone of the streamer line to create drag and reduce entanglements (*sensu* Melvin and Walker, 2009). This resulted in only two recorded entanglements. Despite the relatively small sample size, these trials identified promising approaches to improve the performance of single streamer lines in smaller coastal longliners (Domingo *et al*., 2013).

A multiyear research programme on board Japanese longline vessels participating in the South African joint venture tuna fishery in 2009 (Melvin, Guy and Read, 2013) and 2010 (Melvin, Guy and Read, 2014) has been influential in determining best-practice streamer line design for large pelagic longline vessels and investigating the effectiveness of combinations of mitigation measures such as line weighting, streamer lines and night setting. In 2009, two streamer line designs – the Japanese "light" line (with short streamers) and the "hybrid" line (a mix of long and short streamers) – were compared during night and day setting using unweighted branch lines, but the differences were not statistically significant. Branch lines weighted with 60 g weights reduced the distance astern at which seabirds have access to baits (considered to be 10 m depth) to about 100 m, which is widely agreed as the target aerial extent of streamer lines. In sharp contrast, unweighted lines sank to around 10 m depth three times farther astern than branch lines with 60 g weights (Figure 37). This research highlights the need for line weighting to be used in combination with streamer lines.

This study also found compelling evidence that albatross mortality was driven by secondary attacks in which albatrosses took baited hooks from diving petrels. White-chinned petrels made 74 percent of the attacks on baits, but accounted for only 58 percent of mortalities, while albatrosses made 12 percent of the primary attacks but accounted for 38 percent of mortalities (Melvin, Guy and Read, 2013).

4.3.3 Organizations involved in the mitigation of seabird–longline

FIGURE 37
Estimated distance astern at which baited hooks sank beyond the reach of most white-chinned petrels (10 m) at a vessel speed of 9.5 knots over ground



Notes: Error bars are 95 percent confidence intervals.
Source: Melvin, Guy and Read (2013).

interactions

While substantial challenges remain to document and reduce seabird interactions in pelagic longline fisheries globally, recent advances in the five t-RFMOs have been critical in building the foundations for this work. In 1997, the CCSBT was the first of the five t-RFMOs to address the issue of seabird catches with the adoption of a recommendation for its longline vessels to use a streamer line when fishing south of 30°S (CCSBT, 1997). In 2002, ICCAT adopted Resolution 02–14, which encouraged members to collect data on seabird interactions and to implement, if appropriate, the International Plan of Action for Reducing Incidental Catches of Seabirds in Longline Fisheries (IPOA-Seabirds). This recommendation also expressed an intention for ICCAT to undertake an assessment of the impact of its fisheries on seabird populations when feasible to do so (ICCAT, 2002). Similar seabird measures were adopted by the IOTC, WCPFC and IATTC in 2005 (IOTC, 2005; WCPFC, 2005; IATTC, 2005).

Despite these measures, until 2006, the CCSBT remained the only t-RFMO to require vessels to use mitigation techniques to reduce seabird interactions. Beginning in 2006, there was a period of rapid progress, with a wide range of information on seabirds being presented to the t-RFMOs, including information on albatross distribution obtained from tracking studies, data on seabird catches, albeit still from a limited number of fisheries, and risk assessments. The first of the other t-RFMOs to adopt seabird mitigation requirements was the IOTC, which required pelagic longline vessels to use a streamer line when fishing in the Indian Ocean south of 30°S (i.e. consistent with the CCSBT recommendation; IOTC, 2006). In 2007, ICCAT also adopted a requirement for longline vessels to use a streamer line, in this case when fishing south 20°S (ICCAT, 2007b). In December 2006, the WCPFC adopted a slightly different approach that sought to incorporate the use combinations of mitigation measures for pelagic longline fisheries, and also the need to allow vessels some flexibility to use the measures that most suited their fishing operations. As such, the measure adopted by the WCPFC required longline vessels operating south of 30°S and north of 23°N to use two mitigation measures selected from a range of measures (night setting, streamer lines, line weighting, side-setting with a bird curtain, blue-dyed bait, offal management, underwater setting chute and line shooter), of which at least one needed to be from the first four options (WCPFC, 2006).

These measures were refined over the following years as knowledge increased regarding measures to reduce seabird interactions in pelagic longline fisheries. Overall, in the period 2006–2012, all five t-RFMOs agreed requirements for their longline vessels to use seabird mitigation measures in areas overlapping albatross habitat, with a total of two measures adopted by ICCAT (ICCAT, 2007b, 2011), four by the IOTC (IOTC, 2006, 2008, 2010, 2012e), two by the WCPFC (WCPFC, 2006, 2012e), and one by the IATTC (IATTC, 2011c).

In relation to the current combinations of seabird mitigation measures required, ICCAT, the IOTC and the WCPFC (vessels in the South Pacific) require their vessels to use two mitigation measures from a choice of streamer lines, line weights and night setting when fishing in areas overlapping with albatross distribution (south of 25°S for ICCAT and the IOTC, south of 30°S for the WCPFC, Table 21).

TABLE 21
Currently active t-RFMO conservation and management measures pertaining to seabirds

t-RFMO	CMM	Major provisions relevant to measures to mitigate seabird bycatch	A	B	C	D	E	F	G
CCSBT	October, 2011	Comply with all IOTC, WCPFC and ICCAT measures; implement the International Plan of Action for Reducing Incidental Catches of Seabirds in Longline Fisheries (IPOA-Seabirds) and the FAO Guidelines; report data on interactions to the Commission which is authorized to exchange it with other tuna RFMOs	X	X					X
IATTC	Recommendation C-11-02 (entry into force 1 Sept 2011 for vessels > 24m, 1 Sept 2012 for vessels <24m)	Use at least two of the mitigation measures in Table x, including at least one from Column A, in the area north of 23°N and south of 30°S, plus the area bounded by the coastline at 2°N, west to 2°N-95°W, south to 15°S-95°W, east to 15°S-85°W, and south to 30° S, with the minimum technical standards; report on implementation of the IPOA-Seabirds including the status of their National Plans of Action	X	X	X	X		X	X
ICCAT	Recommendation 07-07	Use streamer lines (bird-scaring/tori lines) in the area between 20°S to 25°S, with an exemption for vessels targetting swordfish which are setting lines at night and using line weights of >=60g within 3 m of the hook		X				X	X
	Recommendation 11-09 (entry into force from July 2013)	Use at least two of three mitigation measures: night setting with minimum deck lighting, streamer lines (bird-scaring/tori lines) and line weighting in the area south of 25°S with minimum technical standards; provide information on implementation and on the status of NPOA-Seabirds; ICCAT to evaluate the efficacy of mitigation measures. Vessels in the Mediterranean are encouraged to use mitigation measures on a voluntary basis.	X	X				X	X
IOTC	Resolution 12/06 (entry into force 1 July 2014)	Use at least two of the three mitigation measures: night setting with minimum deck lighting, streamer lines (bird-scaring/tori lines) and line weighting in the area south of 25°S with the minimum technical standards; record data on seabird bycatch by species and report annually; take photographs of seabirds caught for confirmation of identification; Scientific Committee to analyse the impact of the Resolution; Commission to hold a workshop to facilitate implementation		X			X	X	X
WCPFC	CMM 2012-07 (entry into force 1 July 2014)	Use two of three measures: weighted branch lines, night setting and tori lines, in the area south of 30°S; use at least two of the mitigation measures in Table x, including at least one from Column A, in the area north of 23°N; report implementation of IPOA-Seabirds including the status of NPOAs; CCMs are required to report annually on mitigation used, bycatch rates and total number of birds killed; vessels encouraged to undertake research and ensure safe handling and release	X	X	X	X		X	X

Notes: A = implement FAO guidelines; B = reporting of interactions; C = safe handling and release; D = conduct bycatch mitigation research; E = information for fishers; F = impact assessment and consideration of further mitigation; G = reference to specific mitigation measures. The CCSBT's convention area overlaps with those of ICCAT, IOTC and WCPFC.

The IATTC and WCPFC (North Pacific only) require their vessels to use two mitigation measures from a wider choice of measures (Tables 22 and 23). The CCSBT requires that its vessels comply with the mitigation requirements of the other t-RFMOs (CCSBT, 2011). Such measures mean that, globally, almost all pelagic longline vessels worldwide are now required to use seabird mitigation measures in most areas overlapping with albatross and petrel habitat. One of the main remaining exceptions is vessels less than 24 m in length in the Northwest Pacific, which are currently exempted from WCPFC requirements (WCPFC 2006).

TABLE 22
Seabird mitigation measures adopted by the IATTC

Column A	Column B
Side setting with a bird curtain and weighted branch lines	Streamer line (bird-scaring/tori line)
Night setting with minimum deck lighting	Blue-dyed bait
Streamer line (bird-scaring/tori line)	Deep setting line shooter
Weighted branch lines	Management of offal discharge
	Weighted branch lines
	Underwater setting chute

Note: Longline vessels that are required to comply with the measure (see IATTC [2011c] for details) must apply two different techniques in the table, at least one of which must be from Column A.

TABLE 23
Mitigation measures by the WCPFC in the North Pacific (north of 23°N)

Column A	Column B
Side setting with a bird curtain and weighted branch lineS	Streamer line (bird-scaring/tori line)
Night setting with minimum deck lighting	Blue-dyed bait
Streamer line (bird-scaring/tori line)	Deep setting line shooter
Weighted branch lines	Management of offal discharge

Note: Vessels that are required to comply with the measure (see WCPFC [2012e] for details) must apply at least two of the measures shown with at least one being from Column A.

The period 2006–2012 also saw all five t-RFMOs establishing requirements for their longline vessels to collect data on incidental catches, including seabirds (CCSBT, 2001; WCPFC, 2007; ICCAT, 2010; IATTC, 2011c; IOTC, 2011). Although there has been progress, overall levels of data collection by onboard observers remain below the target of 5 percent coverage of fishing effort (Gilman, Passfield and Nakamura, 2013), and few of these data are in the public domain, possibly owing to current low rates of reporting to the t-RFMOs. Moreover, it is important to note that in almost all pelagic longline fisheries, wider fisheries governance factors also affect the effectiveness of mitigation measures. For example, in recent years, t-RFMOs have been adopting a range of measures to increase their capacity to monitor compliance in their fisheries, which will also enhance their ability to implement effective mitigation measures (Small, 2005; Gilman, Passfield and Nakamura, 2013). At present, the level of implementation of prescribed mitigation measures currently remains a challenge to monitor and enforce owing to low levels of observer coverage and compliance inspections.

In addition to some progress within the t-RFMOs, another of the positive forces behind recent advances in reducing seabird interactions in pelagic longline fisheries is the Agreement for the Conservation of Albatrosses and Petrels (ACAP). In 2007, ACAP established a Seabird Bycatch Working Group (SBWG), which convenes invited technical experts and collaborators from around the world, including individuals from ACAP Parties.[22] The SBWG meets every 12–18 months and is emerging as a globally recognized body for expert advice on the reduction of seabird catches. One of the key outputs of the SBWG is "best practice" mitigation advice for a range of gear types, currently including pelagic and demersal longline and trawl fisheries.

[22] ACAP parties currently consist of Argentina, Australia, Brazil, Chile, Ecuador, France, New Zealand, Norway, Peru, Spain, South Africa, the United Kingdom and Uruguay.

Efforts within individual organizations are supported by an existing international policy framework that includes the FAO Code of Conduct for Responsible Fisheries (the Code) and its IPOA-Seabirds. The Code addresses the need to reduce incidental catches of non-target species in several of its articles (FAO, 1995). The IPOA-Seabirds (FAO, 1999b) was developed under the auspices of the Code to specifically address reduction of seabird catches in longline fisheries. In 2009, FAO published "best practice technical guidelines" to expand the remit of the IPOA-Seabirds beyond longline fisheries and to support development of robust NPOA-Seabirds. These guidelines state that "the use of well-trained observers is the most reliable means of monitoring fisheries performance with respect to seabird incidental catch and use of mitigation measures" (FAO, 2009).

4.4 EMERGING MITIGATION DEVICES AND TECHNIQUES

In addition to the mitigation methods described above that have been under study for some time, there are a number of promising emerging devices and techniques that may offer new solutions. As introduced below, these methods could make major contributions to mitigating seabird interactions in pelagic longline fisheries and help halt the decline of many albatross and petrel populations. The first three share a common approach of protecting the baited hook until it reaches below the diving depth of albatrosses and most petrels, and the latter two are innovative line-weighting options.

4.4.1 Setting capsule

The objective of this stern-mounted hydraulically driven device is to deliver baited pelagic longline hooks underwater to avoid detection by seabirds. It comprises components that are fixed to the vessel and a component (a capsule that holds the baited hook) that freefalls in the water column each time a hook is set.

The underwater setter consists of: a vertical track on the vessel transom; a bait-holding capsule; a box with hydraulics, relays and pulleys; and a control box that houses a programmable logical controller (PLC, Plate 5). The PLC runs the system and records data. To operate the device, the deckhand places a baited hook in the bait chamber of the capsule and presses the release button. At the end of the descent phase, the PLC engages the recovery motor and the capsule returns to the start position. The baited hook is flushed from the capsule on the ascent phases through a spring-loaded door at the bottom of the capsule. The cycle is repeated every 8 seconds. Target release depth can be varied from 4 m to > 10 m, depending on the diving capabilities of the species of seabirds interacting with the gear.

Trials conducted in Australia and Uruguay in recent years have identified several technical difficulties. These are currently being corrected in Australia before further testing in Uruguay in 2014. The capsule is an innovative device with the potential to reduce seabird catches to negligible levels without the necessity for additional seabird deterrent measures.[23]

4.4.2 Smart tuna hook

The smart tuna hook is a seabird mitigation device for pelagic longline fishing that prevents seabirds from accessing baits (Plate 6).[24] The system uses a modified tuna longline hook (circle or Japanese style) that accepts a specially designed steel shield that disarms the hook once it has been baited. The shield is easily and quickly snapped and held onto the baited hook by a clip that has a corrodible alloy link. The link begins to

[23] More information on the bait setting capsule can be found at: www.underwaterbaitsetter.com.au/
[24] More information on the smart tuna hook can be found at: www.oceansmart.com.au/

PLATE 5

The underwater setting capsule in situ on a vessel in Uruguay.

Courtesy of G. Robertson.

dissolve after immersion in salt water and after 15–20 minutes it breaks and releases the shield from the hook. The shield sinks to the seafloor where it corrodes within 12 months, leaving no pollution or toxic residue. The by-product is iron oxide and carbon.

The smart tuna hook works to reduce seabird catches in two ways:

- It adds weight to each branch line directly at the hook, thus increasing sink rate and reducing the availability of baited hooks to seabirds. The mild steel shield is hydrodynamic, weighs about 38 g, and once attached increases the sink rate of the baited hook to 0.6 m/s.
- The shield protects the hook from injuring or being ingested by seabirds in the event that diving seabirds manage to seize a baited hook during deployment.

PLATE 6
Smart tuna hook showing tuna hook in situ.

Courtesy of H. Jusseit.

4.4.3 Hook pod

The hook pod encapsulates the point and barb of pelagic longline hooks to prevent seabirds from accessing baits before a pressure sensitive mechanism operates at a predetermined depth to release baited hooks (Plate 7). This device is based on using the inherently reliable and predictable forces of pressure to operate a hook-release mechanism.

The pods are retrievable and have been designed to last for several thousand deployments. The pod is clipped to the mainline and when it reaches a predetermined depth (below the depth where seabirds can access baits), the pod releases and the branch line and hook sink to the target fishing depth. When the line is hauled, the pods are simply closed and stored ready for the next set.

There are two versions of the pod. One does not contain a light source, and can be used in fisheries that do not light baited hooks, and another contains a light-emitting diode (LED) incorporated into the pod to replace chemical lightsticks (used when targeting swordfish). Chemical lightsticks have been demonstrated to be a major source of marine pollution across the world's oceans (Grasiela, Pinho and Ihara, 2009). In trials conducted to date, there has been no significant increase in non-target catch on pods with the LED incorporated. The LED also offers significant cost savings to fishers compared with the cost of lightsticks and electric fishing lights. The pods are

made of a non-polluting polymer (which decomposes into water [H_2O] and carbon dioxide [CO_2] within 3–4 months), contain no lead and can be produced at a low-unit cost, making them an environmentally and economically attractive option.

Research and development of the pod has been under way for some time, and multiple at-sea trials in Australia, South Africa and Brazil have identified several technical issues that have now been largely overcome. Trials in 2013 were expected to finalize testing and result in a final product ready for commercial use in 2014.

PLATE 7

A hook pod being set in trials in Australia's Eastern Tuna and Billfish Fishery.

Courtesy of O. Yates.

4.4.4 Double line weighting system

In the course of the research conducted in the Japanese joint venture fleet operating in South Africa (see Section 4.3.2.9), the fishing master onboard developed a "double weight system", designed to improve crew safety from "fly-backs" of lead weights and to be more compatible with maintaining the coiled branch lines typical of the Asian high seas longline fleets.

The double-weight configuration consists of two leads placed at either end of a 1–1.5 m section of wire or wire trace (Plate 8). This weighted section is inserted into a monofilament branch line 2 m above the hook. The weight nearest the hook is free to slide along the branch line while the second lead is fixed. The double weight reduces the danger of weight recoil injury by: (i) spreading the mass of the weights (two smaller weights are better than one) across the wire trace; (ii) including a sliding weight that dampens the speed at which the weight can recoil; (iii) including a 1–1.5 m section of stretch-resistant line (wire) that serves to also reduce recoil energy; and (iv) positioning the larger of the two weights in or near the hands of a crew member as the fish is under maximum tension as it approaches the vessel (BMIS, 2013).

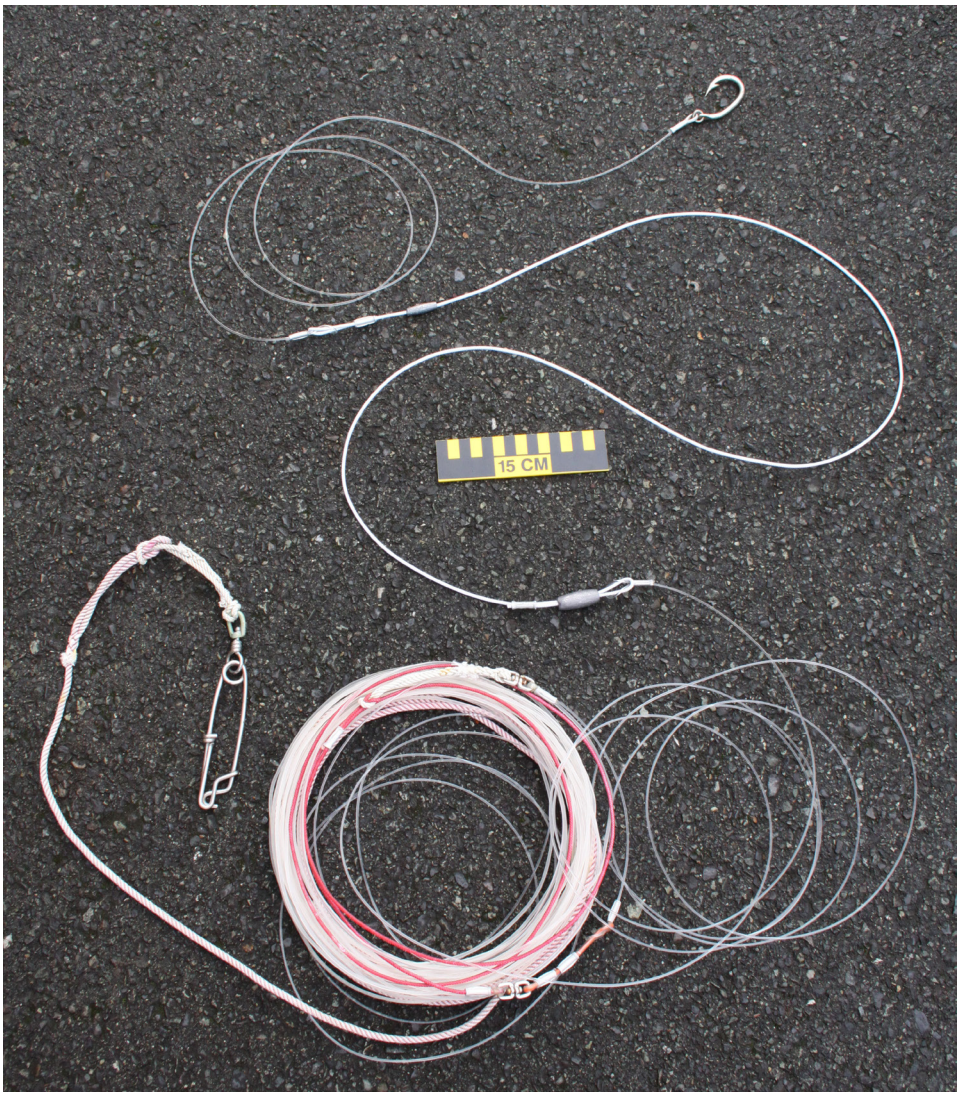
In 2010, more than 95 000 branch lines with the double weight system were hauled with no injuries to fishers, reducing seabird catches by 89 percent more than unweighted branch lines, with no effect on fish catch rates (BMIS, 2013). This

COURTESY OF O. YATES.

technology is beginning to be adopted by the Japanese fleet. In 2011, it won the WWF International Smart Gear competition.

PLATE 8

Double weight system on a Japanese tuna branch line. The white section (with scale) is the inelastic section with weights at each end.



Source: Melvin, Guy and Read (2014)

4.4.5 Sliding leads (formerly called "safe leads") and lumo leads

Line weighting is an indispensable tool in efforts to reduce seabird catches in pelagic longline fisheries. However, the adoption of appropriate line weighting has been hampered by understandable concerns for crew safety. In many pelagic longline fisheries around the world, there is reluctance to adopt a line weighting regime owing to safety concerns raised by traditional leaded swivels causing serious injuries, and even fatalities, when they fly back at the crew in the event of line breakage (e.g. from shark bite-offs) during line hauling (Sullivan *et al*., 2010). Rather than being crimped onto the line, sliding leads slide onto the monofilament and down or off the line in the event of a bite-off, dampening the energy of the recoiling line and preventing threats to crew safety.

There were technical issues with early prototypes breaking on branch lines that caught fish, which resulted in high loss rates (Melvin, Guy and Read, 2013). However, these have been overcome, and sliding lead technology has potential to address the safety concerns associated with conventional line weighting (Løkkeborg, 2011).

The original sliding lead was made up of two lead "halves" in a rubber carrier with two rubber 'O' rings that held the lead in place on the branch line. This technology was then adapted to develop a lumo lead (Plate 9), which has a core of lead encased by 2 mm luminescent nylon sleeve (which glows in the dark) that slides onto the line and is held in place by a screwing collar mechanism.[25] Lumo leads are currently available in two sizes, which weigh 40 and 60 g. The 40 g model is designed to be placed on the hook (Robertson, Candy and Hall, forthcoming), with a crimp sleeve that slides over and protects the crimp. The 60 g lumo is designed to be placed within 2 m of the hook to maximize the sliding safety feature.

Lumo leads have several advantages over weighted swivels: (i) they will improve crew safety by reducing the incidence of dangerous fly-backs during bite-offs; (ii) sink rates for 40 g lumo leads on the hook compare favourably with weights of greater mass farther from the hook (Robertson, Candy and Hall, forthcoming); (iii) the nylon coating prevents the crew from handling exposed lead during gear rigging, setting and hauling; and (iv) the nylon coating will glow for many hours after the set, attracting fish to the vicinity of the hook.

PLATE 9

Lumo leads positioned immediately above the hook in setting bins during trials in Australia's Eastern Tuna and Billfish Fishery.

Courtesy of G. Robertson.

[25] More information on sliding and lumo leads can be found at: www.fishtekmarine.com

4.5 CONCLUSIONS REGARDING SEABIRD INTERACTIONS

Many seabird species are becoming increasingly threatened on a global level. At least 42 species of seabirds in the families Diomedeidae (albatrosses) and Procellariidae (petrels/shearwaters) are vulnerable to interactions with pelagic longline fisheries, and these interactions are considered to be a major factor contributing to their population declines. Estimates suggest that 50 000–100 000 seabirds are killed annually by pelagic longline fisheries, and this level of impact is not sustainable for some populations. In particular, 17 of the 22 species of albatrosses are considered by the IUCN to be threatened with extinction, more than any other bird family. Although some threatened populations have shown increases in recent years, overall population sizes remain small as a result of previous impacts. Characteristics such as small breeding populations, a limited number and range of breeding sites, and exceptionally high k-selected life history traits (e.g. delayed sexual maturity, low breeding frequency, small clutch size, prolonged breeding seasons, and long life span with high adult survivorship under natural conditions), cause seabirds to be extremely vulnerable to increased levels of mortality and make it difficult for them to recover once populations have declined.

The increased availability of tracking data in recent years has considerably deepened understanding of spatio-temporal overlap between seabird distributions and longline fishing. The Atlantic contains more than 20 percent of the total global distribution of breeding albatrosses and giant petrels during the breeding season and about 13 percent of the total global non-breeding distribution (juveniles and immature individuals, plus adults during the non-breeding season). Most of the distribution of albatrosses and giant petrels in the Atlantic is concentrated south of 30°S, but these and other species have limited but important habitats in other areas. Recent estimates of the number of seabirds killed by Atlantic pelagic longline fisheries have ranged from 3 000 to 6 000 in one study and from 10 000 to 17 000 in another. The EPO has a low proportion of the global breeding albatross and giant-petrel distribution compared with other oceans, and only 12 percent of the total global non-breeding distribution. Nevertheless, important breeding habitats for waved, Laysan and black-footed albatrosses are found in the EPO. Habitats in the WCPO are considerably more important for seabirds overall, with more than 45 percent of the global total albatross and giant-petrel breeding distributions found there. Discrete areas in the northern and southern WCPO have an even higher proportion of the global total breeding habitat for some Northern and Southern Hemisphere species, respectively. Interactions with WCPO longline fisheries have been recorded for 36 seabird species. In the Northern Hemisphere, the highest areas of risk were identified as between 20 and 40°N, especially in the northern autumn and winter. In the Southern Hemisphere, the high-risk areas were identified as between 20 and 50°S, especially around New Zealand. The Indian Ocean contains an estimated 14 percent of global breeding albatross and giant-petrel distribution, and 21 percent of the global non-breeding distribution. Breeding seabirds including Amsterdam, Indian yellow-nosed, wandering, shy, grey-headed and sooty albatrosses, and white-chinned petrels spent 70–100 percent of their foraging time in areas that overlapped with IOTC longline fishing effort, mainly by fleets from Taiwan Province of China and Japan, below 20°S.

A review of mitigation techniques for seabird interactions in pelagic longlines considered three classes of methods: (i) avoiding fishing in areas at times of highest risk of seabird interactions (i.e. time/area closures and night setting); (ii) reducing access to baited hooks (weighted lines, streamer lines and side-setting); and (iii) reducing attraction to vessels or bait (offal management, artificial and dyed baits). Based on published scientific results and considerations of best practice, the most promising methods appear to be night setting, side-setting, line weighting and streamer lines, but further research on remaining limitations and the effectiveness of these methods in individual fisheries is required. Important emerging mitigation measures, including

underwater setting and new line-weighting developments, hold promise for future application.

Progress has been achieved in the five t-RFMOs, which all now require all or some of their longline vessels to use at least two seabird mitigation measures from lists that include, *inter alia*, streamer lines, night setting or branch-line weighting in areas that overlap albatross distributions. In addition, all five t-RFMOs now have requirements for longline observer programmes to cover 5 percent of pelagic longline fishing effort. Nevertheless, improved practices are still required, including not only the effective implementation of the measures already adopted, but also improving these to take into account current best practice. Under the broader scope of the IPOA-Seabirds and the supporting best-practice technical guidelines published by FAO, it is acknowledged that effective reduction of seabird mitigation relies on a combination of adequate data from onboard observer programmes, widespread and progressive use of proven mitigation measures, and the development and testing of new techniques. The challenge lies in achieving this in a timely and effective manner across all relevant fisheries. Critical elements in meeting this challenge include: (i) adopting and implementing best-practice mitigation measures; (ii) devoting resources to the development and testing of new and improved mitigation techniques; (iii) improving data collection from at-sea observer programmes, and (iv) ensuring that seabird issues are fully integrated into effective monitoring, control and surveillance systems.

5. Marine mammals

5.1 OVERVIEW OF MARINE MAMMAL INTERACTIONS WITH LONGLINE GEAR

5.1.1 Taxonomy and impact characterization

The largest marine animals interacting with pelagic longline fisheries are the cetaceans. This taxonomic order comprises 87 species in two suborders, the Mysticeti (baleen whales; 15 species) and the Odontoceti (toothed whales, dolphins, porpoises; 72 species). Both suborders are known to interact with pelagic longline gear, although the baleen whales are filter feeders and are less likely to be attracted by the depredation opportunities provided by baited hooks or caught fishes. A third suborder, Pinnipedia (sea lions, walruses and seals), interact mainly with net-based fisheries (Baraff and Loughlin, 2000; Hamer *et al*., 2013). However, interactions with longline fisheries do take place in a limited number of areas where these fisheries occur in close proximity to critical habitat, for example, for Hawaiian monk seals (*Monachus schauinslandi*; NMFS, 2007) and fur seals (*Arctocephalus forsteri*) in New Zealand's southern bluefin tuna fishery (Baird, 2008, 2011). Information on localized pinniped interactions is presented in the section on the Western and Central Pacific (Section 5.2.3). Other taxa of marine mammals include the order Sirenia (manatees and dugongs), the family Mustelidae (sea otters), and the family Ursidae (polar bears), although these are highly unlikely to interact with pelagic longline fisheries owing to lack of habitat overlap. A general review of the literature for pelagic longline fisheries indicates that interactions with cetaceans, and mainly with toothed cetaceans (odontocetes), are the primary concern; therefore, this review focuses on these species.

Marine mammal interactions with longline gear can be classified as either depredation or hooking/entanglement. Depredating marine mammals may remove part or all of a caught fish, but records of depredation are expected to provide minimum estimates because full removal often goes undetected. During depredation, marine mammals put themselves at risk of becoming hooked or entangled, possibly leading to injury or death. Again, records of such occurrences are likely to represent minimum estimates, as some animals may escape before the gear is hauled. Although depredation and hooking/entanglement may be two components of the same behaviour, it is also possible that either may occur in isolation. For example, marine mammals may attempt to feed on bait fish only and become hooked, or accidentally become entangled with longline gear when passing through the area of a set (e.g. baleen whales that would not feed on hooked fish).

Twelve species of cetaceans are confirmed to have depredated from or to have become hooked on, or entangled in, pelagic longline gear (Table 24), although this is undoubtedly a minimum list given that many interactions are not clearly observed. The most common interactions with pelagic longlines at lower latitudes are believed to involve the false killer whale (*Pseudorca crassidens*) and the pilot whales (*Globicephala* spp.; Hamer, Childerhouse and Gales, 2012). Information from United States pelagic longline fisheries clarifies that these two species commonly interact through depredation on deep-set longlines, whereas shallow-set longlines are more commonly depredated upon by bottlenose (*Tursiops truncates*) and Risso's (*Grampus griseus*)

dolphins.[26] Moreover, in United States pelagic longline fisheries, interactions in the form of accidental hooking and entanglement are reported to most commonly involve small dolphins, beaked whales (family Ziphiidae, a group of 21 species of odontocetes) and large whales (not specified, but probably the baleen whales and the sperm whale [*Physeter macrocephalus*]) (Forney, 2009).

TABLE 24
Marine mammal species with confirmed interactions with pelagic longline fisheries and their conservation status

	Scientific name	Common name	D	H	E	CITES	CMS	IUCN Red List
	*Tursiops truncatus*	Bottlenose dolphin	X			II†	I*, II*	LC
	*Delphinus delphis*	Common dolphin	X	X		II	I*, II*	LC
	*Lagenorhynchus obscurus*	Dusky dolphin	X	X		II	II	DD
	*Stenella coeruleoalba*	Striped dolphin	X			II	II*	LC
	*Grampus griseus*	Risso's dolphin	X	X		II	II*	LC
	*Pseudorca crassidens*	False killer whale	X	X		II	–	DD
	*Orcinus orca*	Killer whale	X	X		II	II	DD

[26] Note that interactions between Risso's dolphin and pelagic longline fisheries are not reported by Hamer, Childerhouse and Gales (2012).

	Scientific name	Common name	D	H	E	CITES	CMS	IUCN Red List
	*Globicephala* spp.	Pilot whale	X	X		II	*G. melas* (long-finned pilot whale) II*	DD
	*Physeter macrocephalus*	Sperm whale		X		I	I, II	VU
	*Eubalaena australis*	Southern right whale		X		I	I	LC
	*Monachus schauinslandi*	Hawaiian monk seal		X		I	–	CR
	*Arctocephalus forsteri*	New Zealand fur seal		X		II	–	LC

* Only some populations/subspecies are listed.

† Zero quota for live specimens exported from the Black Sea.

Notes: D = depredation; H = hooking; E = entanglement.

IUCN Red List categories: LC = least concern; DD = data deficient; VU = vulnerable; CR = critically endangered (IUCN 2013a).
Sources: Extracted from Hamer, Childerhouse and Gales (2012) and Forney (2009).

Photo credits: All NOAA Fisheries except for New Zealand fur seal courtesy of D. Hamer.

With regard to utilization, while cetaceans are intentionally taken either under objection or reservation to the moratorium on commercial whaling agreed by the members of the International Whaling Commission (IWC), or under scientific permits or strike/catch limits set for indigenous whaling activities (IWC, 2013a), none of these catches is prosecuted by pelagic longline fisheries. Owing to disagreement within the IWC regarding its remit for species other than the "great whales" (12 species of baleen whales plus the sperm whale), catches for the smaller cetaceans are often undocumented or not publicly available (IWC, 2013b; DEFRA, undated). Although fisheries taking these species also do not generally employ pelagic longline gear, in the absence of regulations to the contrary, smaller cetaceans caught inadvertently on pelagic longlines may be utilized (e.g. for food).

Pelagic longline fishing grounds overlap with the distributions of most cetacean species, but available data are too limited to estimate current or historical interaction or mortality rates by species (Hamer, Childerhouse and Gales, 2012). Reported cetacean interaction rates with longline fisheries worldwide have ranged from 0.05 to 0.231 individuals per set (number of hooks not given; Hamer, Childerhouse and Gales, 2012), but these rates vary by species and by area (see Section 5.2). As is the case for

most non-target species, interactions are under-reported by fishers, but this is even more likely to be true in the case of marine mammals because they escape or remove target species whole, leaving little evidence of their presence (Hamer, Childerhouse and Gales, 2012).

Although the primary focus of this chapter is on the impacts of longline fishing on marine mammal populations, it is also noted that in some cases the impacts of the marine mammals' behaviour on the fishery is severe. This mainly occurs through depredation, i.e. scavenging of hooked longline catch by predators. In some fisheries, economic impacts due to depredation resulting in damage to target species, bait loss and gear breakage, have been estimated as high as €340 (US$436) per 1 000 hooks (IOTC, 2007). Understanding depredation rates is also important for fisheries management because if depredation goes unnoticed or, as is more likely, unreported, fishing mortality to target species will be underestimated.

5.1.2 Factors influencing interactions and mortality

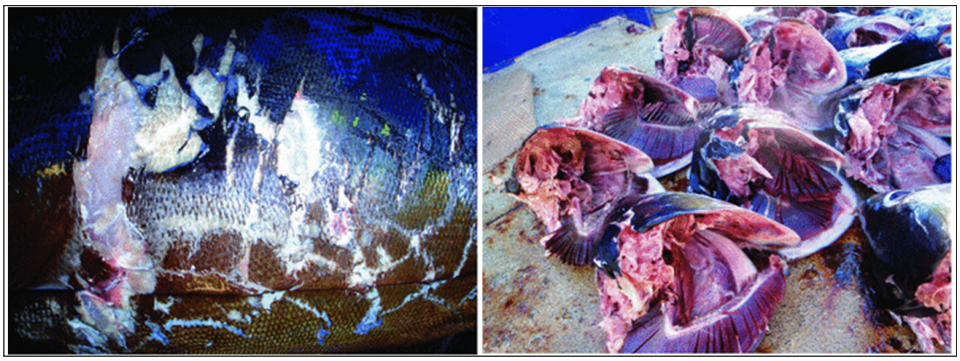
Marine mammals may interact with pelagic longline fisheries in ways that are negative for the marine mammals and somewhat negative for the fishery (e.g. hooking, entanglement), or positive for the marine mammals (e.g. an easy feeding opportunity) and strongly negative for the fishery (e.g. target species damage). In the latter case, marine mammals may benefit from better foraging opportunities and energy savings by preying upon fish hooked on longlines that are otherwise too large, too deep or too fast for them to catch (Gilman *et al*., 2006b; Hamer, Childerhouse and Gales, 2012). The same depredation behaviours that benefit marine mammals may, however, result in accidental hooking or entanglement, which can lead to asphyxiation, impaired foraging, or other injuries (Cassoff *et al*., 2011). Moreover, these depredation behaviours and the resulting economic losses may encourage fishers to kill the animals involved in order to minimize future losses (Gilman *et al*., 2006b, Hamer, Childerhouse and Gales, 2012). In addition to depredation, hooking, entanglement and fisher antagonism, interactions may also take the form of vessel strikes. The prevalence of each type of interaction varies by species and area, but one study of endangered North Atlantic right whales (*Eubalaena glacialis*) found that, at the population level, vessel strikes and entanglement were the first- and second-most important threats to survival (Vanderlaan, Smedbol and Taggart, 2011). Citing a previous study of 38 humpback whales stranded in the mid-Atlantic waters of the United States of America between 1985 and 1992, it was stated that 16 percent had injuries consistent with entanglement. However, this study also concluded that pelagic longline gear posed the second-lowest entanglement threat of the six gear types examined (only hagfish traps posed a lower threat) (Vanderlaan, Smedbol and Taggart, 2011). Moreover, in a study of 45 right and humpback whale entanglements in the North Atlantic between 1993 and 2002, none was attributed to longline gear (Johnson *et al*., 2005). These studies help to place interactions between marine mammals and pelagic longline fisheries in context and suggest that the interactions described in this chapter comprise only a small portion of the overall risk to these populations.

Odontocetes that intentionally interact with longline gear for the purpose of depredation are believed to orient toward distinctive sounds made by fishing vessels and their equipment. There are also some reports of odontocetes actively trailing longline vessels (Gilman *et al*., 2006b). Both odontocetes and sharks can seriously damage catch before it can be brought to the vessel, but the patterns of depredation are different. Lacking the razor-sharp teeth of sharks, odontocetes are said to tear and shred prey rather than make clean bites as sharks do (Hamer, Childerhouse and Gales, 2012; Plate 10). They are also known to be able to remove certain sections of the fish or even pluck it from the hook (Gilman *et al*., 2006b). It has been suggested that it is the use of sonar that allows marine mammals to so skilfully avoid the longline hook

and to depredate at night (McPherson and Nishida, 2010). Despite these important differences between odontocete and shark depredation, marine mammals may be more often blamed for depredation losses simply because they are more visible at the surface. Similarly, while smaller odontocetes are believed to be attracted to longline gear for the purpose of eating bait, it has been suggested that their role in bait predation may be overestimated relative to that of other fishes and squids (Hamer, Childerhouse and Gales, 2012).

PLATE 10

Damage to tuna caused by depredating toothed whales typically shows either scraping toothmarks (left) or precise removal of the entire body behind the gills.



Courtesy of D. Hamer.

Given that odontocetes are often attracted to the captured fish rather than the bait, mitigation measures involving baits or hooks that are effective in reducing interactions for other species may not be effective for marine mammals. On the other hand, the attractiveness of longline gear to cetaceans is in many ways similar to the attractiveness of any fishing gear that ensnares fish and provides an easy opportunity for feeding (e.g. gillnets). This suggests that mitigation measures that work well for one type of fishing-gear–marine-mammal interaction may also work for another. This type of multigear review was conducted by Werner *et al.* (2006) for 66 studies of bycatch reduction techniques across a number of taxa. In this review half (33) of the techniques were aimed at reducing marine mammal interactions (including seals), and 21 of these were based on acoustics (pingers, acoustic harassment devices, passive acoustic deterrents or predator sounds). Although no studies of acoustics in longline fisheries were reviewed, a number of trials of acoustics in gillnet fisheries appeared effective (Werner *et al.*, 2006), and the preliminary results of trials of acoustic dissuasion devices for longlines are reported in McPherson and Nishida (2010). In addition to highlighting the potential for solutions to work across gear types, the potential for the same technique to be effective for cetaceans, pinnipeds and even seabirds was noted (Werner *et al.*, 2006). Nevertheless, given the high capacity for learning in marine mammals (e.g. relative to sharks), an important issue in the development of mitigation measures, particularly acoustic methods, will be the potential for what is initially a deterrent to become a signal that a feeding opportunity is available (Gilman *et al.*, 2006b).

5.1.3 Species risk profiles and international conservation initiatives

As discussed above there is no international agreement with regard to whether the scope of the IWC extends beyond the 13 species of great whales, of which 2 species (sperm whale and southern right whale) have been documented to interact with pelagic longline fisheries (Hamer, Childerhouse and Gales, 2012). However, other international

conservation treaties list both great whales and smaller cetaceans in classifications that call for special management measures. The most relevant for fisheries management are CITES, the CMS, and the IUCN Red List.

All of the great whales, except for the west Greenland population of minke whale (*Balaenoptera acutorostrata*), are listed on CITES Appendix I, which entails a general prohibition on commercial international trade in these species. In addition, another 75 cetacean species are listed on either Appendices I or II, which applies trade controls to all countries that are parties to CITES and do not take out a specific reservation to the listing.[27] Of the cetacean species listed on CITES Appendix I, two are known to interact with pelagic longline fisheries (sperm whale and southern right whale). The remainder of the cetaceans known to interact with longline fisheries (Table 24) are listed on CITES Appendix II. Of the two pinniped species known to interact with pelagic longline gear, the Hawaiian monk seal is listed on CITES Appendix I and the New Zealand fur seal is listed on CITES Appendix II.

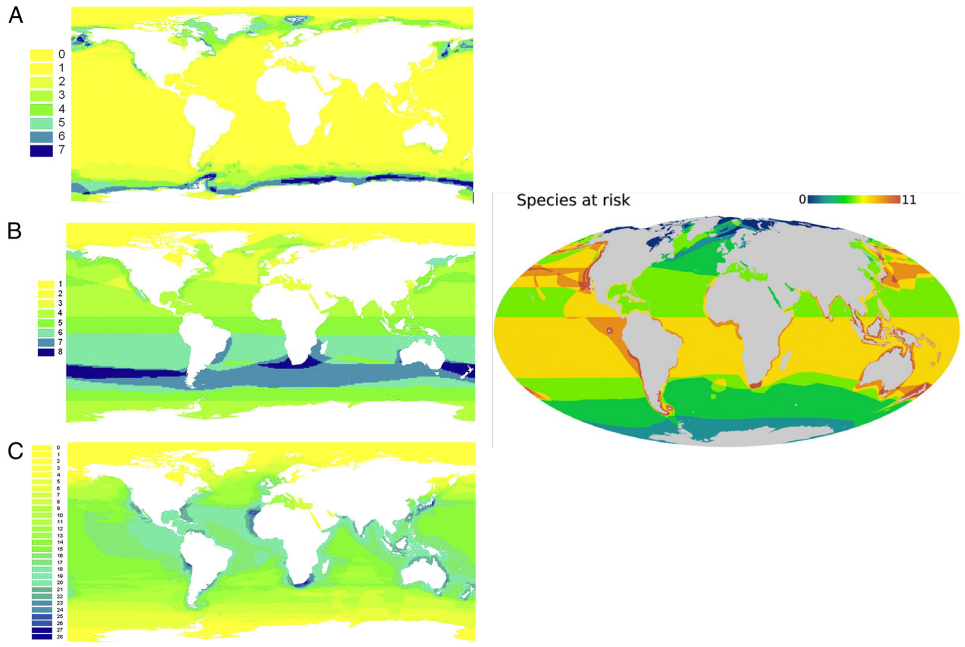
The CMS lists species that are either "threatened with extinction" (Appendix I) or able to "significantly benefit from international cooperation" (Appendix II; CMS, 2012). Cetaceans comprise 15 of the species listed on CMS Appendix I and 44 of the species listed on CMS Appendix II. Of the cetacean species known to interact with pelagic longline gear (Table 24), 1 is listed on CMS Appendix I only, 5 on CMS Appendix II only, 3 on both CMS Appendices, and 1 is not listed (false killer whale). Neither the Hawaiian monk seal nor the New Zealand fur seal is listed on the CMS.

Most of the cetaceans known to interact with pelagic longline gear are listed by the IUCN Red List as either "data deficient" or "least concern" (Table 24). The only exception is the sperm whale, which was listed as "vulnerable" owing to the ongoing depleted state of the population (IUCN, 2013a). The Hawaiian monk seal is listed as "critically endangered" and the New Zealand fur seal is listed as "least concern". The different designations for these species in the three systems (CITES, CMS and the IUCN Red List) probably reflect differences between classification systems based on threats that are trade-related, rooted in migratory behaviours, and/or based on documented population declines regardless of trade or migration patterns, respectively. Nevertheless, it is clear from the number of marine mammals species listed in each system that there is considerable conservation concern associated with this group.

Two recent papers by researchers at the National Autonomous University of Mexico have mapped marine mammal species richness in the global ocean and assessed extinction risks by species and area (Pompa, Ehrlich and Ceballos, 2011; Davidson *et al*., 2012). The first study found differing patterns of species richness by taxonomic group, with pinniped species richness concentrated at the poles, especially near Antarctica, baleen whales showing high species richness in the vicinity of 30°S, and the highly speciose odontocetes concentrated near tropical coasts (Figure 38, left panel). The pattern of overlaps between the odontocetes and in the world's tropical and temperate tuna fisheries, in combination with their different feeding behaviours, helps to explain why pelagic longline interactions occur most frequently with odontocetes. Extinction risk was found to be concentrated mainly in coastal areas, with the major pelagic longline fishing grounds (with the exception of the Eastern and North Pacific) located in areas of medium risk for most marine mammal species (Davidson *et al*., 2012; Figure 38, right panel). Applying its own model, this study found an additional 15 marine mammal species to be threatened with extinction (IUCN Red List categories of "critically endangered", "endangered" or "threatened") in addition to the 32 already listed as such by the IUCN Red List. However, none of these is a species identified above as interacting with pelagic longlines. One of the most interesting findings of Davidson *et al*. (2012) was that life-history traits have a stronger influence on

[27] See: www.cites.org/eng/resources/species.html; also see footnotes to Section 2.1.3.

FIGURE 38
Geographic patterns of species richness in a) pinnipeds, b) baleen whales and c) odontocetes (toothed whales)



Note: Number of species shown along the margin (left); and global distribution of marine mammal species at risk of extinction with warmer colours signifying higher numbers of species at risk (right).
Sources: Reprinted for non-commercial and educational use from Pompa, Ehrlich and Ceballos (2011) and Davidson et al. (2012) ©2011, 2012 National Academy of Sciences.

extinction risk than environmental and human impact variables. This led the authors to call for more research into basic biology, particularly for elusive and deep-dwelling species such as beaked whales.

5.2 MARINE MAMMAL INTERACTIONS BY AREA

As discussed above, information on marine mammal interactions with longline fisheries is limited not only by under-reporting, as is the case for other taxa discussed in this paper, but also by the more cryptic nature of the interactions (Hamer and Childerhouse, 2012). In particular, the presence of marine mammals is not always apparent and some impacts may be attributed to other causes (e.g. entanglement may be considered gear breakage, or loss of target species may be attributed to sharks or environmental conditions). For these reasons, the following discussions of interactions in each area of the global ocean are limited and uneven.

5.2.1 Atlantic

Several studies document interaction rates between marine mammals and pelagic longlines in the Atlantic. These studies complement those discussed above by Johnson *et al.* (2005), Cassoff *et al.* (2011) and Vanderlaan, Smedbol and Taggart (2011), which describe interactions between marine mammals, mainly large whales, and fishing gear, rarely including longlines, based on post-mortem entanglement evidence from the North Atlantic (see Section 5.1.2).

In a study of Spain-based longline fleets, both observer and logsheet data were used to derive depredation rates of 1–9 percent of the total number of sets per vessel for 8 vessels. However, in sets for which depredation occurred, between 2 and 100 percent of the catch was lost, with more than 25 percent lost on two-thirds of these occasions. Based on bite marks, most depredation was attributed to false killer whales. However,

this species was considerably less frequently sighted, probably because it remained at depth. Cetacean sighting rates were much higher off the Azores than off Brazil, with the most commonly sighted species being the common dolphin, usually associated with high catches of swordfish, and the spotted dolphin, usually associated with high catches of shortfin mako sharks (Hernandez-Milian *et al*., 2008).

A recent analysis of observer data from a number of pelagic longline fleets operating in the western Mediterranean from 2000 to 2009 focused on hooking rates in the most frequently encountered species, Risso's dolphin (n=33 of 57 marine mammal individuals observed; Macías López *et al*., 2012). Over the ten years of the study, a total of 2 587 sets were observed, and marine mammals were hooked on 2 percent of the observed sets for an overall interaction (i.e. catch) rate of 0.011 marine mammals per 1 000 hooks. The highest interaction rates were found in a small, experimental surface longline fishery targeting swordfish, but the Japanese bluefin tuna longline fleet also had elevated interaction rates (Table 25). Characteristics common to both fleets that may determine interaction rates with Risso's dolphin include a long mainline, gear fished for at least 24 hours (i.e. including night-time soak), use of squid bait, and hooks fished at depths of 50–100 m.

TABLE 25
Hooking rates (number of cetaceans caught per 1 000 hooks) in the western Mediterranean by fleet and species, 2000–09

Fleet	Common dolphin	Striped dolphin	Risso's dolphin	Long-finned pilot whale	Total (including unidentified marine mammals)
LLALB	0.0020	0.0040	0	0	0.0060
LLJAP	0	0	0.0246	0.0038	0.0303
LLHB	0.0015	0.0016	0.0028	0.0006	0.0077
LLHBexp	0	0	0.0588	0	0.0588
LLPB	0	0	0	0	0
LLSP	0	0	0.0087	0	0.0108
LLAM	0	0.0030	0.0120	0	0.0150

Note: LLALB = drifting surface longline targeting albacore; LLJAP = Japanese surface longline targeting bluefin tuna; LLHB = home-based surface longline targeting swordfish; LLHBexp = experimental home-based longline targeting swordfish; LLPB = bottom longline; LLSP = drifting semi-pelagic longline targeting swordfish; LLAM = American drifting surface longline targeting swordfish.
Source: Macías López *et al*. (2012).

The ICCAT list of marine mammals that interact with longline fisheries includes six species: fin whale (*Balaenoptera physalus*), long-finned pilot whale, Risso's dolphin, striped dolphin, bottlenose dolphin and goosebeaked whale (*Ziphius cavirostris*). These species were included in a comprehensive PSA exercise for Atlantic tuna fisheries, and, as a taxonomic group, marine mammals were found to have the highest average intrinsic vulnerability to population decline. However, their susceptibility scores were extremely low owing to the infrequency of their interactions with longline fisheries, and they were thus not included in the final risk ranking (Arrizabalaga *et al*., 2011).

United States Atlantic longline fisheries reported a total of 37 interactions with marine mammals for 2011 involving: spotted, bottlenose and Risso's dolphins; and pilot, false killer and pygmy sperm whales (*Kogia breviceps*). Twelve of these individuals were dead, one was alive and the remainder were seriously injured, mainly owing to gear ingestion (Garrison and Stokes, 2012). The most frequent interactions were with pilot whales, which, when extrapolated to the entire United States Atlantic longline fishery, were estimated at 350 (95 percent confidence interval of 200–643) individual interactions for 2011, a figure close to historic high levels (Garrison and Stokes, 2012).

5.2.2 Eastern Pacific Ocean

The EPO is known as the most active area of interactions between marine mammals and tuna fisheries worldwide, but these interactions are known almost exclusively from the purse seine fishery (Hall and Roman, 2013). The IATTC's AIDCP is explicitly focused on reducing dolphin mortalities in the purse seine fishery, and recently adopted requirements for national longline observer coverage do not call for recording of interactions with marine mammals (IATTC, 2011a; see Section 1.2.1). Ecological risk assessments conducted by the IATTC have included three dolphin species but have assessed susceptibility to purse seine fisheries only (IATTC, 2012a).

It is not known whether the IATTC's lack of data on marine mammal interactions with longline fisheries derives from its historical lack of observer coverage of the longline fishery, a dearth of interactions, a diversion of attention from the relatively small risk posed by the longline fishery as compared with the purse seine fishery, or a combination of these factors. A review of United States marine mammal stock assessments for the EPO (mainly Washington, Oregon and California) suggests that longline fishery interactions are a minor component of human-caused mortality and serious injury to marine mammals in the region (NOAA, 2013c). Estimates for marine mammal bycatch in United States fisheries dating from the late 1990s suggest that 23 percent of the United States cetacean bycatch was taken by fishing gear other than gillnets or trawls (i.e. potentially by longlines and other gear types), and of the total (all gear types) only 7 percent was taken in the Pacific excluding Alaska (Read, Drinker and Northridge, 2006). These results from the well-monitored United States fisheries suggest that the scale of marine mammal bycatch in longline fisheries in the EPO is small.

5.2.3 Western and Central Pacific Ocean

The only known study of the threats to marine mammals from pelagic longline fisheries was conducted in the form of an ecological risk assessment of the WCPO longline fisheries for 15 species of baleen whales, toothed whales, seals and sea lions using WCPFC observer data (Kirby and Hobday, 2007). This assessment found that most of the marine mammals had medium- or high-risk rankings (depending on which of three formulae were applied) with the exception of the New Zealand sea lion (*Phocarctos hookeri*), which had a low or medium risk. Two geographic regions of interactions were highlighted: around the Hawaiian archipelago, and off the South Island of New Zealand (Kirby and Hobday, 2007). However, these areas may have been identified because they host longline fisheries with observer programmes that record marine mammals interactions, rather than because they have higher interaction rates than other (unobserved) longline fisheries in the WCPO. Nevertheless, the remainder of this discussion focuses on these two regions.

The pelagic longline fishery based in Hawaii has used observer data to monitor marine mammal interactions for many years. A summary of encounters for 2004–08 (Table 26) indicates that deep-set longlines have recorded a greater number of interactions than have shallow-set longlines, but this may be attributed to the much larger number of sets observed in the deep-set fishery (n=20 724 versus n=6 228). Another difference between the two fisheries is that the deep-set fishery occurs about 400 km to the south of Hawaii and, thus, interacts more with tropical species than the more northerly shallow-set fishery that interacts with more temperate species. Three species are believed to engage in depredation behaviour – bottlenose dolphin, short-finned pilot whale and false killer whale – with the latter most frequently implicated. This species is also the only one for which the interaction rates have exceeded thresholds set by the United States Marine Mammal Protection Act, which has prompted further research and management to mitigate impacts (see Section 5.3). Modelling of a variety of factors including area, time, vessel, habitat, fishing gear, and

set characteristics was undertaken in order to explore covariates that could be useful in predicting and avoiding depredation by false killer whales, but the only significant factor identified by the model was a reduction in repeat depredation if the subsequent set occurred more than 100 km from the previous set location. It should be noted that the time interval between sets could not be included in the model owing to a lack of contrast in the data (Forney *et al*., 2011).

TABLE 26
Marine mammal interactions (classified as deaths, serious injuries, non-serious injuries) in the Hawaii shallow- and deep-set pelagic longline fishery, 2004–08

Species	Deep-set longline fishery				Shallow-set longline fishery			
	Deaths	*Serious injuries*	*Non-serious injuries*	*Subtotal*	*Dead*	*Serious injuries*	*Non-serious injuries*	*Subtotal*
Blackfish	0	6	0	6	0	1	0	1
Risso's dolphin	1	4	0	5	2	8	0	10
Short-finned pilot whale	0	6	2	8	0	0	0	0
Blainville's beaked whale	0	0	1	1	0	0	0	0
Humpback whale	0	0	1	1	0	1	1	2
False killer whale	2	14	3	19	0	0	1	1
Pan-tropical spotted dolphin	1	0	0	1	0	0	0	0
Striped dolphin	1	0	0	1	0	1	0	1
Bottlenose dolphin	0	1	0	1	0	4	0	4
Unidentified cetacean	0	4	1	5	0	0	1	1
Unidentified beaked whale	0	1	0	1	0	0	0	0
Bryde's whale	0	0	0	0	0	0	1	1
Pygmy sperm whale	0	0	0	0	0	0	1	1

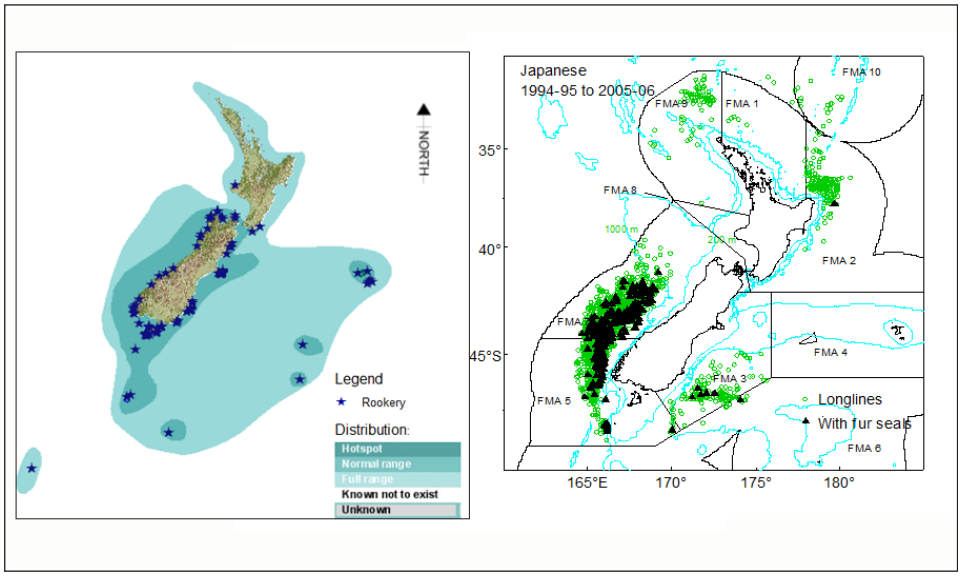
Note: Blackfish: false killer whale or short-finned pilot whale; short-finned pilot whale: *Globicephala macrorhynchus*; Blainville's beaked whale: *Mesoplodon densirostris*; humpback whale: *Megaptera novaeangliae*; sperm whale: *Physeter macrocephalus*; pan-tropical spotted dolphin: *Stenella attenuate*; Bryde's whale: *Balaenoptera edeni*.
Source: Forney (2010).

Application of the WCPFC ecological risk assessment methodology to the waters of New Zealand produced a risk ranking for two species of Otariidae (fur seals and sea lions) that was low relative to the 26 species of seabirds assessed (Waugh *et al*., 2008). Although the productivity of the two mammal species was low, their vulnerability to longline fisheries was considered low as well, resulting in a low overall risk (Waugh *et al*., 2008).

The primary concern in New Zealand waters with regard to marine mammals and pelagic longline fisheries is the interaction of the New Zealand fur seal (*Arctocephalus forsteri*) with the fishery for southern bluefin tuna (Abraham and Thompson, 2011). Most reported interactions (94 percent) are with the Japanese fishery off the west coast of the South Island. This is to some extent a reflection of observer coverage in this fishery and the fact that this fishery is conducted in close proximity to important fur-seal breeding colony and haul-out sites (Figure 39; Baird, 2008, 2011). Interaction (hooking) rates are reported as ranging from 0.010 to 0.112 fur seals per 1 000 hooks in the period 1995–2006 for both domestic and Japanese pelagic longline fisheries. Most observed interactions are mouth hookings, although foul hooking may also occur, and

FIGURE 39

New Zealand fur seal at-sea distribution and location of breeding colonies as of 2010 and observed Japanese longline sets with and without fur seal interactions, 1994–2006



Sources: Left panel: Baird (2011); right panel: Baird (2008).

most fur seals (98 percent) are reported to be released alive with the hook and part of the leader still in place (Baird, 2008). Although they were included in the WCPFC ecological risk assessment, interactions with New Zealand sea lions and whales are reportedly very rare (Abraham and Thompson, 2011). Killer whale depredation has been reported from New Zealand longline fisheries off the North and South Islands (Visser, 2000).

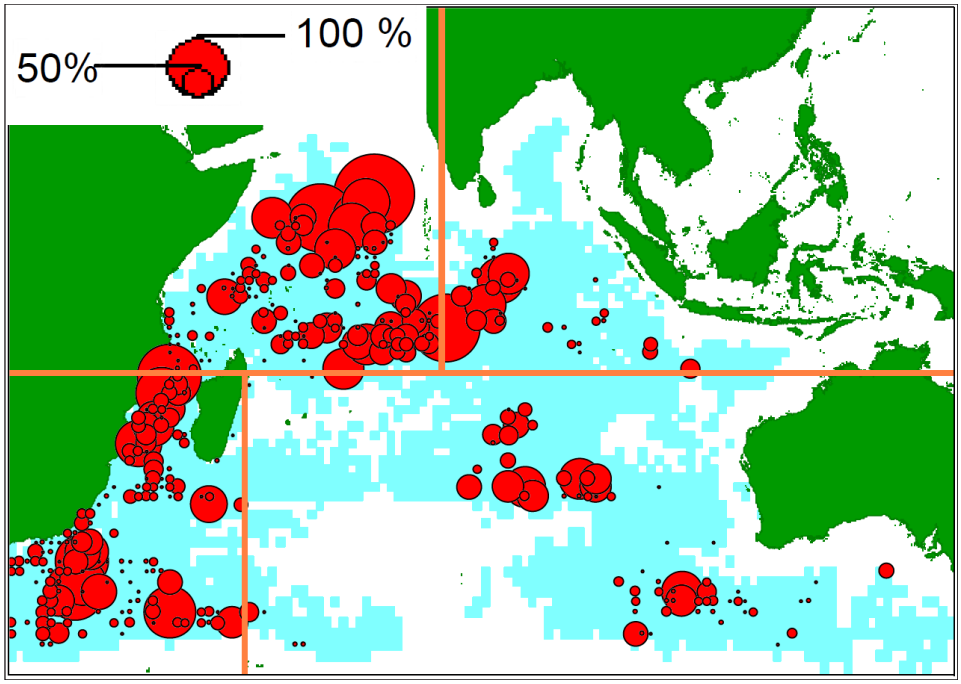
5.2.4 Indian Ocean

Concerns regarding marine mammal interactions with fishing gear in the Indian Ocean are most often articulated with regard to the extensive coastal gillnet fisheries. Nevertheless, the results of recent surveys have identified that interactions with longline fisheries most commonly involve the false killer and short-finned pilot whales, and less frequently, the killer whale (IOTC, 2007). Other species with documented but less frequent interactions include Risso's dolphin (off Réunion); humpback whale (off Madagascar); and false killer whale, spinner dolphin (*Stella longirostris*) and melon-headed whale (*Peponocephala electra*, off Mayotte) (Kiszka *et al.*, 2008; Kiszka *et al.*, 2010). Data reported from the Japanese commercial longline fleet in the period 2000–05 indicated that the largest depredation losses occurred in the bigeye tuna fishing grounds in the northwest Indian Ocean, the yellowfin tuna grounds between Mozambique and Madagascar, and the southern Bluefin tuna grounds south of South Africa (Figure 40; Nishida and Shiba, 2006).

More so than in other regions, concerns regarding marine mammal interactions with longline fisheries in the Indian Ocean have focused on depredation. The IOTC is the only t-RFMO to have initiated a research programme dedicated specifically to depredation issues. This programme began in 1999, involved data collection by 7 countries spanning 5 years, and was summarized in a workshop and report in 2007 (IOTC, 2007). While this programme highlighted issues associated with marine mammal and shark predation in the Indian Ocean, it is not clear that depredation rates in this region are higher than in other areas. For example, a summary of data collected

FIGURE 40

Average annual depredation rates by both marine mammals and sharks on all species of hooked fish as reported by Japanese commercial longliners in the Indian Ocean, 2000–05



Note: The size of the circle represents the percentage of the catch that was damaged.
Source: Nishida and Shiba (2006).

from Japanese research and training vessel cruises from 1977 to 1981 indicated higher interaction rates in the Pacific than in the Indian Ocean (Nishida and Tanio, 2001). Other data compiled for the IOTC study suggested that interaction rates in other oceans are comparable. It should be noted that some of these studies were not able to distinguish accurately between marine mammal and shark depredation as historical species identification capabilities were limited or unreliable (IOTC, 2007).

The IOTC programme seems to have focused exclusively on depredation rates as there is no information given regarding hooking or entanglement of marine mammals. While some data are available for the total depredation rates upon all species (regardless of whether they were target species), and some data were presented in units of hooks or sets fished, most data were presented in terms of the number of target species damaged versus the number of target species caught. These rates ranged from 4 to 56 percent for cetaceans alone, but were usually near the low end of this range; depredation rates by sharks, when given, were often similar. The workshop concluded that the fraction of the total catch that suffers from depredation is less than 5 percent on average. However, on those sets where depredation occurs, a large portion of the catch can be lost (up to 100 percent). Even with the extensive data set compiled for the workshop, the delineation of patterns of depredation was complicated by findings of both temporal and spatial variability as well as different depredation rates for different predator–prey species combinations.

5.3 MANAGEMENT MEASURES AND THEIR EFFECTIVENESS

None of the t-RFMOs has adopted any management measures that pertain to the issue of operational interactions between marine mammals and longline fisheries. It is not clear whether a lack of management has led to a lack of data, or vice versa, but data holdings for longline interactions with marine mammals appear to be very

limited. In general, the t-RFMOs have taken one of two approaches to reporting these interactions: collecting data through longline observer programmes; or requesting fishers to record interactions.

The first approach of collecting data through longline observer programmes has the benefit of standardized data collection but the drawback of generally low sample coverage. In the case of the WCPFC, where the longline observer programme is administered centrally as a Regional Observer Programme (ROP),[28] observers are trained to collect data on 16 marine mammal "species of special interest", including both baleen and toothed whales. They are also asked to document landings, sightings, bait stealing and depredation in narrative text fields in their trip report (SPC/FFA, 2007). These data, which given the emphasis of the WCPFC ROP on purse seine fisheries are mainly expected to represent purse seine fisheries, are maintained by the WCPFC ROP but they are not routinely analysed or reported. In the case of other t-RFMOs that do not have regional longline observer programmes, and instead rely on national observer programmes for coverage (i.e. ICCAT, CCSBT and, most recently, IATTC and IOTC) the extent to which marine mammal interactions are reported depends on the requirements of each national observer programme. In some cases, such as the United States of America, the data requirements are comprehensive (e.g. including the point of hooking or entanglement, mammal condition and behaviour, biological measurements, other species present, and whether attempts were made by fishers to avoid the interaction [Cotter, 2010]) and these data are provided to ICCAT. At the other end of the spectrum are those national observer programmes whose requirements with regard to marine mammals are minimal and/or not well documented, and which do not provide these data to the relevant t-RFMO, as well as those with little or no longline observer coverage.

A second approach to documenting longline fishery interactions with marine mammals through logsheet recording is illustrated by the research programme described above for the IOTC (see Section 5.2.4). In support of this programme, Japan requested its commercial longline fishers to voluntarily record depredation events, and 480 vessels did so for more than 20 000 sets over a five-year period (Nishida and Shiba, 2006). While the size of this data set is impressive, there are a number of shortcomings associated with the data, including no data for sets without depredation, difficulties in relating depredation data to fishing effort (i.e. because the depredation forms were separate from the logsheets), and the potential for depredation to be attributed to the wrong species (e.g. sharks or a different species of mammal) all of which have implications for mitigation measures. Moreover, the IOTC noted that some members refused to submit the depredation data to the secretariat on the grounds that they were confidential (IOTC, 2007).

For all of these reasons, the majority of information sources included in this report originate from national management programmes. The largest national management programmes for marine-mammal–longline fishery interactions are in countries with well-developed marine resource management programmes and where national longline fisheries have existed or still do exist in close proximity to protected species. Several examples are discussed below.

Under the United States Marine Mammal Protection Act, populations are assessed every 1–3 years to determine whether interactions with fisheries are likely to be causing adverse population effects. The most recent case of action being taken in response to such assessments is the situation with regard to false killer whales in the waters around Hawaii. In this case, it was determined that interaction rates for the false killer whale exceeded the threshold (called "potential biological removal" or PBR) for adverse

[28] For many years before the establishment of the ROP, a consistent longline observer programme was operated by the Forum Fisheries Agency and the Secretariat of the Pacific Community.

effects, and thus a requirement for a management plan (called a "take reduction plan" or TRP) to reduce both mortalities and serious injuries was triggered (Forney *et al*., 2011). The goal of TRPs is to reduce mortality and serious injuries to below the PBR within six months, and within five years to reduce mortalities and serious injuries to near-zero levels. For false killer whales around Hawaii, the TRP required eight management actions: use of specified circle hooks; use of specified branch line materials to keep the animal on the hook and allow the branch line to be cut near the hook; establishment of a year-round closed area for longline fishing around the Main Hawaiian Islands; a triggered closure for an area south of the Main Hawaiian Islands if deep-set longline fisheries exceed a specified number of interactions in a given year; and four measures to improve fishers' responses when an interaction occurs (NOAA, 2012c). The false killer whale TRP took effect at the end of December 2012 (NOAA, 2012d). In January 2013, a first interaction occurred with the deep-set longline fisheries and, under the terms of the TRP, if two interactions occur within one year the southern exclusion area will be closed for the remainder of the year (Gutierrez, 2013).

Elsewhere in the United States of America, there have been nine other marine mammal take reduction teams convened, and two of these have involved longline fisheries (NOAA, 2013d). The first was the Atlantic Offshore Cetacean Take Reduction Team, convened in May 1996 to consider mitigation measures for right whales, humpback whales, sperm whales, beaked whales, pilot whales, common dolphins, bottlenose dolphins and spotted dolphins in the Atlantic pelagic driftnet, pelagic longline and pair trawl fisheries. However, by 2001, two of the fisheries no longer existed and the longline fishery had been substantially modified to reduce interactions with sea turtles and billfishes; therefore, the team was disbanded. The second team involving longline fisheries is the Pelagic Longline Take Reduction Team covering long-finned and short-finned pilot whales in the Atlantic pelagic longline fishery. This TRP was implemented in June 2009 and is still operational. It has three requirements: (i) posting a marine mammal handling and release placard; (ii) a limit of 20 nautical miles for longline mainlines in certain broad-scale areas of the Atlantic coast; and (iii) special observer and research requirements in "hotspot" areas (NOAA, 2009). Data on marine mammal interactions are collected by observers and assessed annually for trends and against historical levels (Garrison and Stokes, 2012). Another United States-based example of the mitigation of longline impacts on protected marine mammal species pre-dates the TRPs and involves the Hawaiian monk seal. In October 1991, in response to 13 unusual seal wounds thought to have resulted from interactions with the Hawaii pelagic longline fishery, the National Marine Fisheries Service (NMFS) established a protected species zone extending 50 nautical miles around the Northwest Hawaiian Islands and its corridors, and since then no interactions have been observed (NOAA, 2012e).

In the Southwest Pacific, both Australia and New Zealand manage longline interactions with marine mammals primarily through ongoing fishery monitoring programmes. Both countries require that interactions with protected species be recorded by longline fishers in logbooks and by observers when present. In Australia, four species have been recorded in total from these two data sets including short-finned pilot whale, melon-headed whale, humpback whale and Australian fur seal, as well as unidentified whales and seals. For the period 2008–2012, a total of 22 interactions were been recorded in longline logbooks and by observers, but the number of interactions reported for 2012 was zero. These data are evaluated by Australia's Department of Sustainability, Environment, Water, Population and Communities on a quarterly basis but, unlike for sea turtles, there does not appear to be a fixed threshold for evaluating whether interaction rates are unacceptably high (Patterson, Sahlqvist and Larcombe, 2013). In New Zealand longline fisheries, the marine mammal interaction of greatest concern is the New Zealand fur seal (see Section 5.2.3). However, this species is

listed by the Department of Conservation as "not threatened", and the population is increasing in New Zealand waters (Brouwer and Griggs 2009). The most recent data on interactions suggest that an estimated 20 fur seals are hooked each year in the pelagic longline fishery and all of these are released alive. A Code of Best Practice for the Mitigation of the Effects of Fishing in New Zealand Pelagic Longline Fisheries has been developed by pelagic longline fishers and calls for cutting the branch line at the closest possible point to the animal for all marine mammals (Brouwer and Griggs, 2009). The New Zealand government has distributed line cutters to fishers to facilitate such release and has not set any limit on the number of interactions (A. Hore, personal communication, July 2013).

5.4 OTHER MITIGATION METHODS

As the preceding examples have shown, existing management measures for longline fishery interactions with marine mammals involve a combination of closed areas, monitored interaction thresholds, and safe release. However, all of these measures are reactive in that actions are triggered as a result of interactions rather than aimed at preventing them in the first place. A large number of studies have been conducted to test various methods for discouraging marine mammals from approaching longline gear, and these studies are the subject of this section. The following subsections describe the main types of methods, i.e. special longline hooks, physical barriers, and acoustic mitigation methods, which are still in the development phase but may have potential management application in the future. Some information on other mitigation techniques is presented in Hamer, Childerhouse and Gales (2012), but these appear to be in the earliest stages of development and are not discussed here.

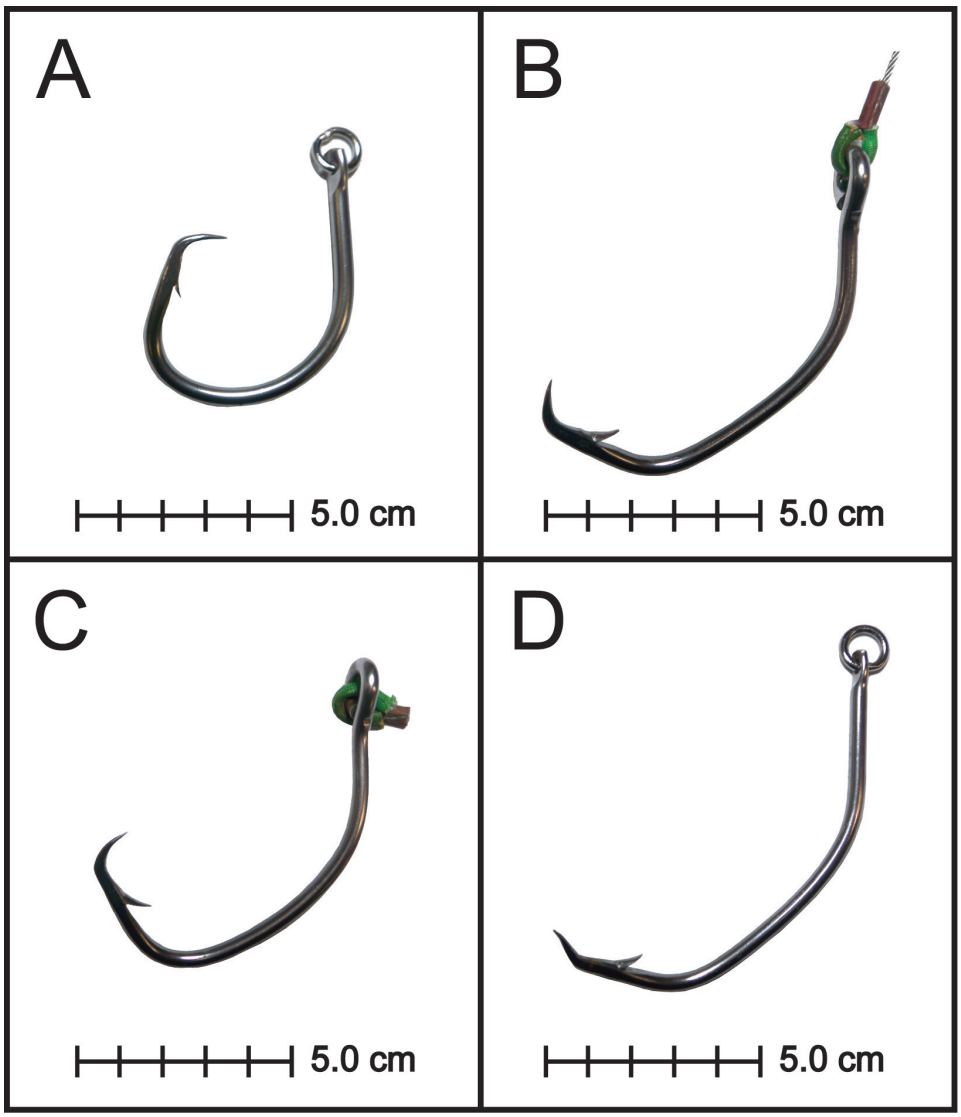
5.4.1 Weak hooks

Spurred by a desire to develop better mitigation techniques for interactions between the Hawaii-based deep-set pelagic longline fishery and false killer whales, a study was conducted in 2010 to test "weak" hooks. These hooks are designed to straighten more easily than other hooks and, thus, were hypothesized to effectively release large non-target species while maintaining catch rates of target species. It should be noted, however, that weak hooks would not be expected to have any effect on depredation rates. Weak hooks have been previously tested in the Atlantic to minimize pelagic longline interactions with pilot whales (Bayse and Kerstetter, 2010). In that study, there was no difference between the catch rates for the target yellowfin tuna and swordfish species on weak or strong hooks, which indicates that target species catch rates can be maintained. However, interactions with marine mammals were too few to evaluate whether they were successfully released by the weak hooks. In the Gulf of Mexico, weak hooks were trialled as a means of decreasing the selectivity of gear for larger Atlantic Bluefin tuna when targeting yellowfin tuna. That study found that catches of bluefin tuna were statistically significantly reduced by 56 percent when using the weak hooks while yellowfin catch rates were maintained (Bigelow *et al*., 2012).

Prior to the false killer whale interaction issue, the deep-set Hawaii longline fishery had already voluntarily transitioned from primarily using Japanese tuna hooks in the early 2000s to using circle hooks, or a combination of circle hooks and other hooks, by the late 2000s (i.e. it is only the shallow-set Hawaii longline fishery for which circle hooks are mandated). As circle hooks are weaker than the traditional hooks, this voluntary change to circle hooks in the deep-set fishery was estimated to result in a weakening of hooks by at least 30 percent (i.e. comparison based on the strongest circle hook). In weak hook trials conducted in Hawaii, the 15/0 circle hooks used represented a weakening of 46 percent (for the 4.5 mm gauge) and 64 percent (for the 4.0 mm gauge) as compared with a Japanese tuna hook. Of more than 300 000 hooks deployed, 76 were found to have been straightened and 70 of these were weak hooks

(Figure 41). One of the strong hooks was observed to be straightened by a false killer whale estimated to be more than 1 000 kg. Catch rates of the target species – bigeye

FIGURE 41
Examples of an unfished and straightened hooks from trials of weak hooks in the deep-set Hawaii longline fishery



Note: (A) Lateral view of a control 15/0 circle hook (4.5 mm diameter wire). (B) Control hook straightened by a false killer whale. (C) Weak hook with a 131 cm FL retained bigeye tuna. (D) A weak hook straightened by an unknown animal.
Source: Bigelow et al. (2012).

tuna – were maintained on the weak hooks, although it was noted that in seasons other than the trial season larger bigeye tuna are caught, and whether these sizes of bigeye tuna would have been retained on the weak hooks is unknown. It is probable that the main factors in determining whether hooks are straightened are the direction and force of the pull, which is in turn determined by whether the hook is attached to the branch line with a ring or non-ring, the location of hooking and the force applied to the mainline and branch line during haulback (Bigelow *et al.*, 2012). Under new regulations designed to minimize interactions with false killer whales as of December 2012, the deep-set Hawaii longline fishery is now required to use circle hooks with a maximum

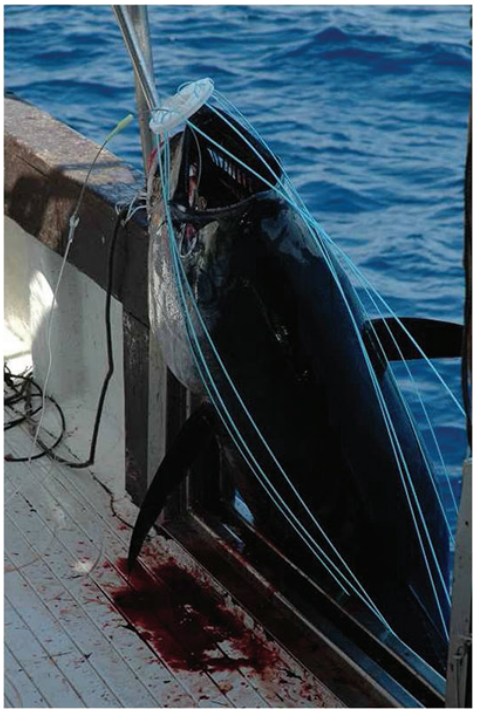
wire diameter of 4.5 mm and an offset of 10 degrees or less. In addition, monofilament leaders and branch lines in this fishery must have a minimum diameter of 2 mm, and leaders or branch lines made from any other materials must have (an equivalent) minimum breaking strength of 181.4 kg. The purpose of the latter requirement is to ensure that the hook is the weakest component of the terminal tackle and thus straightens before any other element of the branch line fails (leaving the terminal tackle attached and risking post-release mortality(NOAA, 2012d).

5.4.2 Physical barriers

Another means of discouraging marine mammal interactions with pelagic longline gear takes the form of physical barriers that drop into place and protect hooked fish from marine mammal depredation. By reducing the occurrence of depredation, it is thought that adverse effects on marine mammals (i.e. hooking, entanglement) will also be reduced. Moreno *et al*. (2008) report success with net-sleeve configurations in the demersal longline fishery preventing sperm whale interactions, but Hamer, Childerhouse and Gales (2012) note that the principles of application would be different in pelagic longline fisheries. In particular, for demersal longline fisheries, the threat of depredation occurs only during haulback, and the net sleeve is put in place by water resistance as the line is hauled. In contrast, for pelagic longlines, the barrier device would need to deploy immediately after hooking to protect the target fish and block the hook throughout the soak (Rabearisoa *et al*., 2012). The need for a triggering

PLATE 11

Three physical barrier devices tested in trials in the Seychelles and Fiji. The "spider" (left) consists of eight polyester legs maintained by a 100 mm diameter plastic disc through which the branch line is inserted. The spider is triggered by a beta pin released when a hooked fish pulls on the branch line. The "sock" (top right) is a conical net made of fibreglass mosquito netting or propylene fibre net that is rolled up and then triggered with the same beta pin when a hooked fish pulls on the branch line. The cage device (bottom right) consists of a functional deterrent structure that will either physically or psychologically deter a depredating odontocete.



Courtesy of N. Rabearisoa (left, top right) and D. Hamer (bottom right).
Sources: Rabearisoa et al. (2012); Hamer and Childerhouse (2012).

mechanism, an expandable physical barrier, and a configuration that is both inexpensive and time-efficient to attach to a branch line are challenges for this type of device (Hamer, Childerhouse and Gales, 2012). In addition to net-sleeve or sheath approaches, some devices have been designed based on the theory that marine mammals avoid entangled gear. Using this principle, streamers made of plastic tubes, monofilament or wires have been proposed, and some designs have incorporated metallic elements in an attempt to disrupt marine mammal echolocation patterns (McPherson and Nishida, 2010).

Recent experiments on pelagic longlines in the Indian Ocean trialled streamer and sheath designs described, respectively, as "spiders" and "socks" (Rabearisoa *et al*., 2012; Plate 11). Spiders were found to deploy correctly more often than socks (87 percent versus 69 percent), and were less likely to become entangled with the longline gear (4 percent versus 18 percent). However, neither spiders nor socks protected hooked fish any better than the control treatment of no protection, and it was noted that neither device offered effective protection for billfish as neither could deploy over and around the bill. The cost of the devices is not given, but the deployment time increased by 4 times for the spiders and by 12 times for the socks. In addition, both devices increased haulback time considerably owing to water resistance and entanglement problems. The authors also note that any lessening of depredation rates due to the devices may disappear as cetaceans become habituated to them (Rabearisoa *et al*., 2012). It is thus clear that there are many issues still to be overcome before physical barrier methods are ready for broad scale deployment.

Hamer and Childerhouse (2012) also report on South Pacific trials of spider-like devices with free hanging lines ("chains") and encircling hoops ("cages"). Low interaction rates hindered statistical interpretation of the data; however, results were consistent with either a deterrent effect for marine mammal depredation, an attraction effect for target species, or both. More significantly, no incidents of design failure or material fatigue were recorded, and setting and hauling times were minimally affected (although an extra crew member was required).

5.4.3 Acoustic methods

Among bycatch mitigation measures for marine mammals, by far the greatest attention has been focused on acoustic methods. For several decades, longline fishers have improvised techniques for scaring marine mammals away from fishing gear by using metallic percussion (e.g. banging the vessel's stern or a pipe placed in the water) (P. Miyake, personal communication, August 2013). More recent technologies and techniques have been designed and tested for nets (Werner *et al*., 2006; see Section 5.1.2) and would require overcoming significant practical and design issues if they are to be applied to longline gear that may be more than 100 km in length (McPherson *et al*., 2008; FAO 2013b). Moreover, the purpose of the signal is different in different fisheries. For example, fixed or towed gear applications typically require an alert signal to be broadcast to the marine mammal, so it can avoid collision, encirclement or entrapment (Werner *et al*., 2006). Such alert signals, typically called acoustic deterrent devices (ADDs), or "pingers", use sounds of less than 150 dB (Shapiro *et al*., 2009). In contrast, depredating marine mammals may need to be actively deterred using acoustic harassment devices (AHDs). These devices are designed to intimidate or cause pain, using sounds of greater than 170 dB (Shapiro *et al*., 2009). One drawback, especially with ADDs, is that they may become a "dinner bell" (i.e. attractant) for individuals that hear them often (Caretta and Barlow, 2011). A related issue is that the effective deterrence of either type of device may diminish through time owing to habituation (Mooney, Pacini and Nachtigall, 2009).

The ADDs and AHDs introduced above are considered to be two forms of direct acoustic mitigation because they seek to influence the marine mammals' behaviour. Another form of direct acoustic mitigation related to ADDs is echolocation disruption

devices (EDDs). These are designed to confuse and disorient the marine mammals so that they are unable to accurately locate the hooked fish and depredate it (Hamer, Childerhouse and Gales, 2012). Other studies have explored the potential for masking the sound signature of fishing vessels (McPherson and Nishida, 2010) and playback of predator (e.g. killer whales) noises to induce a flight response from potential prey (e.g. pilot whales; Dahlheim, 1988). Although a number of trials of these direct techniques have been conducted, successful application in longline fisheries has proved difficult (Hamer, Childerhouse and Gales, 2012). It is not clear whether this is due to technology issues, habituation or simply a lack of sufficient experimentation (e.g. Nowacek *et al*., 2007). For example, one study on a captive animal showed that acoustic deterrence initially reduced echolocation performance from a baseline of 95 percent to 47 percent but it subsequently improved to 85 percent either with behavioural adaptation or a decrease in the device's emitted sound pressure over time (Mooney, Pacini and Nachtigall, 2009). In addition to the uncertainties about effectiveness, some of these techniques are currently constrained from wider application by the large sizes of the batteries and transponders required (Hamer, Childerhouse and Gales, 2012) and by concerns regarding adverse effects on marine mammals (Nowacek *et al*., 2007; Gilman, 2011).

Another approach to acoustic mitigation techniques is indirect. Rather than aiming to change the behaviour of marine mammals, these techniques are designed to allow the fishery to avoid marine mammals. One such technique involves using passive listening devices to detect and avoid areas based on identification of species-specific whistles sounded by marine mammals during depredation events (McPherson and Nishida, 2010). Although there has been some limited success with such techniques in experimental trials, results are often inconclusive (Hamer, Childerhouse and Gales, 2012).

5.5 CONCLUSIONS REGARDING MARINE MAMMAL INTERACTIONS

Marine mammals interacting with longline fisheries are taxonomically diverse and, while mainly involving toothed whales and dolphins (odontocetes), also include species ranging from baleen whales such as the southern right whale to pinnipeds such as fur seals. All longline interactions with marine mammals can be considered undesirable from the point of view of the fishery, but may be either positive (i.e. depredation provides feeding opportunities) or negative (i.e. the risk of hooking or entanglement) from the point of the view of the marine mammal. While finding ways of avoiding depredation has been a key motivation for research on mitigating marine mammal and longline interactions, it is likely that cetaceans are often blamed for depredation actually undertaken by sharks, simply because they are more visible at the sea surface.

Of the 10 cetacean species most commonly interacting with pelagic longline fisheries, 2 are listed on CITES Appendix I (sperm whale and southern right whale) and the remainder are listed on CITES Appendix II. Nine of these species are listed on one or both of the CMS appendices. In contrast, only the sperm whale, and the Hawaiian monk seal and New Zealand fur seals (both of which have localized interactions), have IUCN Red List threatened status. These different designations probably reflect differences between classification systems based on threats that are trade-related, rooted in migratory behaviours, and/or based on documented population declines, respectively.

Global mapping has documented that cetacean species diversity is concentrated in tropical coastal areas often coinciding with prime tuna longline fishing grounds. Studies of interaction and depredation rates in the Atlantic suggest that dolphin interactions are problematic for Spain-based fleets, whereas in the western Atlantic the United States pelagic longline fisheries have frequent interactions with pilot whales. In the EPO, marine mammal interactions with longlines are minor compared with

the region's purse seine fisheries, as well as infrequent compared with United States fisheries on all coasts. The western Pacific pelagic longline fisheries' interactions with marine mammals are poorly characterized except in areas with robust observer programmes. The most problematic issue encountered in this fishery concerns false killer whale interactions off Hawaii, which have led to a TRP under the United States Marine Mammal Protection Act. In the Indian Ocean, reports of depredation damage to target species prompted the t-RFMO (the IOTC) to undertake a focused, multiyear research programme. Despite heightened concerns, and a finding that up to 100 percent of the catch may be lost in some depredation events, rates of occurrence were found to be similar to those in other oceans, and less than 5 percent on average.

None of the t-RFMOs has adopted management measures that pertain to mitigating the level of operational interactions and consequential impacts on marine mammals and longline fisheries. The fact that there are few t-RFMO data holdings may be either the cause or the product of this. In contrast, some countries with national observer programmes have robust monitoring of interactions, and in some cases management systems for imposing mitigation measures if interactions rise above a specified threshold (e.g. false killer whales and pilot whales in the United States of America). In other cases, interactions remain at low levels that are considered to be appropriately managed through safe release guidelines and periodic reviews of monitoring data (e.g. fur seals in New Zealand).

Mitigation technologies and techniques often need to be designed with the particular characteristics of marine mammals (as opposed to sharks) and longlines (as opposed to fixed or towed gear) in mind. While some methods are primarily intended to prevent depredation, this is believed to also lower the probability of hooking and entanglement. Weak hooks take advantage of size and power differences to straighten and release very large animals while still retaining most smaller target species, but they are not expected to reduce depredation. Physical barriers such as streamers, sleeves or cage devices either prevent access to caught fish or perhaps simulate gear tangles and thus psychologically deter depredation. One of the major issues to be overcome in association with these barrier devices is the additional time and/or cost required to deploy them. Development of acoustic techniques is complicated by the extensive area covered by a single longline deployment and by shortfalls in sufficiently powerful, compact and cost-effective technologies. With the advanced capacity for learning in most mammals, acoustic mitigation measures must also guard against behavioural adaption or habituation, or be used in conjunction with other mitigation techniques. Despite these challenges, reducing marine mammal interactions, particularly in the form of depredation, will provide a direct economic benefit to fishers. Therefore, this provides an ideal opportunity to encourage their participation in the development and implementation of effective mitigation measures to reduce hooking and mortality impacts.

6. Other bony fishes

This paper has defined the target species of longline fisheries for tuna and tuna-like species as the 51 species of the family Scombridae (mackerels, Spanish mackerels, bonitos and tunas), and the 13 species of the families Istiophoridae and Xiphiidae (billfishes). It has then discussed four major groups of bycatch taxa: elasmobranchs, sea turtles, seabirds and marine mammals. This chapter discusses the other bony fishes (i.e. besides the Scombridae and billfishes) that are caught in association with pelagic longline fisheries. The scope of longline fisheries covered in this paper (see Section 1.1) is particularly relevant to this chapter. In particular, longline catches of other bony fishes may vary considerably based on vessel size and distance from shore, but often these details are not available in the literature and data. Therefore, particular caution should be exercised in generalizing findings from any one fishery to longline fisheries as a whole.

6.1 INTERACTIONS WITH LONGLINE GEAR BY AREA

Information on the catches of other fishes in pelagic longline fisheries fishing for tuna and tuna-like species is extremely limited, and most available information is qualitative based on presence/absence records. Although there may be exceptions for some species in some areas, t-RFMOs generally do not require that catches of other fish be reported by flag States in annual statistical submissions. However, some flag States do report this information on a voluntary basis, and there are likely to be some national data holdings that are not reported to the t-RFMOs.

The proximate reasons for the lack of reporting requirements are likely to be that these other fish species are not targeted; are of low commercial value; and are often discarded, used as food for the crew and/or sold in local markets as mixed catch. Related to this there is also a relatively high potential for misidentification and a lack of concern for species with fast growth and short reproductive cycles. The conundrum is that, without sufficient data, it is difficult to determine the need for management, and without management there is no stimulus to improve catch and biological data.

This data-poor situation for other bony fishes is worse than, but materially similar to, that of the other bycatch taxa discussed in this report. The primary difference between other bony fishes on the one hand, and elasmobranchs, sea turtles, seabirds and marine mammals on the other, is the amount of research effort that has focused on the latter. As illustrated by the studies cited in the previous chapters, a large number of review articles have attempted to assemble data from national observer programmes, research surveys, and even experimental studies, and to analyse these data using innovative meta-analysis techniques. While some of the same data sources may contain information on other bony fishes, no comparable analytical effort has yet been undertaken.

As discussed in the following sections, for pelagic longline fisheries targeting tuna and billfishes, the most commonly caught bony fishes other than the Scombrids and billfishes are dolphinfish (*Coryphaena* spp.), opah (*Lampris guttatus)*, oilfish (*Ruvettus pretiosus*), escolar (*Lepidocybium flavobrunneum*), and ocean sunfish (*Mola* spp.). These species are distributed worldwide, and are short-lived and fast-growing (Table 27).

TABLE 27
Bony fishes other than Scombrids and billfishes that interact with pelagic longline fisheries targeting tuna worldwide

	Species	Common name	Distribution	Maximum length
	*Coryphaena* spp.	dolphinfish	Tropical and subtropical waters of the Atlantic, Indian and Pacific	210 cm (TL)
	*Lampris guttatus*	opah	Worldwide in tropical to temperate waters	200 cm (TL)
	*Ruvettus pretiosus*	oilfish	Circumtropical and temperate seas of the world	300 cm (TL)
	*Lepidocybium flavobrunneum*	escolar	Tropical and temperate seas of the world, but probably not occurring in the northern Indian Ocean	200 cm (SL)
	*Mola* spp.	ocean sunfish	Warm and temperate zones of all oceans	333 cm (TL)

Notes: Distribution and maximum length information from Froese and Pauly (2011). TL = total length; SL = standard length.
Photo credits: NOAA Fisheries.

6.1.1 Atlantic Ocean

While ICCAT (2007a) has published a list of bycatch species caught by longline gear of which a portion are bony fishes not classified as Scombrids or billfishes, this list indicates only the presence or absence of these fishes in longline catch records, and thus does not imply that any of them are caught or killed in significant quantities. Although there are 44 entries, many of the listings are not species-specific, and if the maximum number of species represented by each listing is tallied, there are potentially 659 species (Table 28). Most of these species have not been assessed for the IUCN Red List, but of those that have, nine species are classified in threatened categories: the common

sea bream is endangered; and one triggerfish, one cod, two eels and four triggerfish are vulnerable. It is unlikely that these species are caught in large numbers in pelagic longline fisheries but it not possible to know the scale and frequency of the interactions from available ICCAT data.

TABLE 28
Bony fish bycatch species, other than Scombrids or billfishes, listed by ICCAT as present in longline catches

Scientific name	Common name	IUCN Red List category	No. of species
Alepisaurus brevirostris	Shortnose lancetfish	–	1
Alepisaurus ferox	Longnose lancetfish	LC	1
Aphanopus carbo	Black scabbardfish	–	1
Balistidae	Triggerfish	1 spp. VU, 6 spp. LC	~40
Brama brama	Atlantic pomfret	–	1
Campogramma glaycos	Vadigo	–	1
Caranx hippos	Crevalle jack	–	1
Coryphaena equiselis	Pompano dolphin fish	LC	1
Coryphaena hippurus	Dolphinfish (mahi-mahi, dorado)	LC	1
Cubiceps spp.	Bigeye cigarfish	LC	10
Elagatis bipinnulata	Rainbow runner	–	1
Epinephelus spp.	Grouper	2 spp. LC, 1 sp. DD	99
Gadus morhua	Cod	VU	1
Gempylus serpens	Snake mackerel	–	1
Lampris guttatus	Opah	–	1
Lepidocybium flavobrunneum	Escolar (oilfish)	–	1
Lobotes surinamensis	Tripletail	–	1
Lophius americanus	Goosefish	–	1
Lophius piscatorius	Monk fish	–	1
Luvarus imperialis	Luvar	–	1
Macrouridae	Rat-tail	16 spp. DD, 12 spp. LC	1
Mola mola	Ocean sunfish	–	1
Mola spp.	Sunfish	–	1
Molva dypterygia	Blue ling	–	1
Nesiarchus nasutus	Black gemfish	–	1
Ophichthidae	Eel	13 spp. DD, 37 spp. LC, 2 spp. VU, 1 sp. NT	>300
Polyprion americanus	Stone bass	DD	1
Pomatomus saltatrix	Bluefish	–	1
Ranzania laevis	Slender mora (sunfish)	–	1
Rachycentron canadum	Cobia	–	1
Remora remora	Remora	–	1
Remora spp.		2 spp. LC	5
Ruvettus pretiosus	Oilfish	–	1
Sciaenops ocellatus	Red drum	–	1
Seriola dumerilii	Greater amberjack	–	1
Seriola spp.	Amberjack	1 sp. LC	9
Pagrus pagrus	Common sea bream	EN	1
Sphyraena barracuda	Barracuda	–	1
Taractes asper	Rough pomfret	–	1
Taractichthys steindachneri	Sickle pomfret	–	1

Scientific name	Common name	IUCN Red List category	No. of species
Taractichthys longipinnis	Big scale pomfret	–	1
Tetraodontidae	Puffer	4 spp. VU, 9 spp. DD, 32 spp. LC	~120
Trachipterus arcticus	Deal fish	–	1
Trichiuridae	Snake mackerel	1 sp. LC, 1 sp. DD	~40

Notes: Each entry's scientific name has been matched with IUCN Red List categories for the taxon (IUCN, 2013a) and the exact (if known) or approximate number of species in each taxon is shown. IUCN Red List categories are: DD = data deficient; LC = least concern; NT = near threatened; VU = vulnerable; and EN = endangered. IUCN Red List "threatened" categories are VU, EN and CR (critically endangered).
Source: ICCAT (2007a).

A recent ICCAT project designed to coordinate bycatch information and develop a database to store it found very little data available for other bony fishes within the ICCAT data holdings (Cotter, 2010). Because catch and effort data submission for these species is voluntary, most of the existing data for these species originate in observer data sets. As described in Section 1.2.1, although there are a number of national observer programmes operating in the ICCAT convention area, the quantity of data from these programmes reported to ICCAT has to date been minimal, and ICCAT has yet to conduct stock assessments for any of the species listed in Table 28.

Although quantitative information on the catch of other bony fishes in the longline fisheries managed by ICCAT is lacking, a number of sources indicate that dolphinfish are a major component of the catch in some parts of the convention area (Mahon and Oxenford, 1999; Macías *et al*., 2012). The only known stock assessment for this species was conducted for the southeastern United States of America by NOAA, where it was noted that recreational catches are two to ten times higher than commercial catches. While stock status based on the 2000 assessment remains uncertain, it appears that overfishing is not occurring and that the stock is not overfished (Cass-Calay and Ortiz, 2009).

In an 11-year study of dolphinfish bycatch in Spanish Mediterranean pelagic longline fisheries, important differences were found in catch rates and sizes of dolphinfish between fleets. Albacore tuna fleets had the highest catch rates (3.7 dolphinfish per 1 000 hooks), whereas swordfish targeting fleets had slightly lower catch rates (0.9–1.2 per 1 000 hooks), and bluefin tuna targeting fleets had near zero catch rates comparable with demersal longlines. The size of dolphinfish caught was found to correlate with the depth of the set and the size of the hooks, such that deeper sets with larger hooks, as in the bluefin tuna targeting fleet, caught larger dolphinfish (Macías *et al*., 2012).

6.1.2 Eastern Pacific Ocean

As characterized by Andraka *et al*. (2013) the small-scale (artisanal) and large-scale pelagic longline fisheries of the EPO can be separated into those targeting tunas and billfishes and those targeting dolphinfish (also referred to as dorado or mahi-mahi). The gear and vessel characteristics of the fleets vary from country to country, but within each country the characteristic features of dolphinfish targeting operations as compared with tuna and billfish targeting operations are the use of J hooks rather than tuna hooks, and trip lengths that are shorter by about 40 percent. While the dolphinfish-targeting longlines reportedly catch almost exclusively their target species, those targeting tuna and billfish also catch dolphinfish. Oilfish is the only other bony fish besides the Scombridae and billfishes that is reported to be commonly caught in these fisheries (Andraka *et al*., 2013).

Longline observer programmes in the EPO are in their infancy (see Section 1.2.1), and in contrast to other t-RFMOs, information on catches of other fishes in the EPO is based on logsheet reports. The IATTC has estimated longline catches of

carangids (yellowtail [*Seriola lalandi*]), rainbow runner, and jack mackerel (*Trachurus symmetricus*), dolphinfish and unidentified other fishes since 1982; figures since 1995 are shown in Table 29. Dolphinfish catches peaked in 2001 at just under 16 000 tonnes but appear to have varied between 2 500-4 500 tonnes in recent years. The IATTC reports that most catches of dolphinfish are unloaded in Central and South American ports, but there is some uncertainty about which gear types are used to catch them (IATTC, 2012a). No estimates for other non-Scombrid and billfish species of bony fishes are provided.

TABLE 29
Estimated catches of carangids, dolphinfish and unidentified other fishes in the Eastern Pacific Ocean by longline fisheries, 1995–2011

Year	Carangids (yellowtail, rainbow runner and jack mackerel)	Dolphinfish (*Coryphaena* spp.)	Other
		(tonnes)	
1995	–	39	210
1996	–	43	456
1997	–	6 866	848
1998	–	2 528	1 340
1999	–	6 284	975
2000	4	3 537	1 490
2001	18	15 941	1 726
2002	15	9 464	1 914
2003	54	5 301	4 681
2004	1	3 986	671
2005	–	3 854	558
2006	–	3 404	262
2007	6	2 980	2 001
2008	5	4 423	616
2009	10	4 238	1 412
2010	8	1 245	1 587
2011	7	403	74

Source: IATTC (2012a).

6.1.3 Western and Central Pacific Ocean

The scientific data to be provided annually to the WCPFC by its members does not include any catch or effort data for bony fishes besides the Scombrids and billfishes (WCPFC, 2013b). Therefore, all information on bycatch of these fishes available to the WCPFC is derived from observer programmes. The WCPFC's observer programme is arguably more comprehensive and better coordinated than those of the other t-RFMOs, but it still appears not to meet coverage requirements of 5 percent in some areas (see Section 1.2.1).

Analysis of observer data for bycatch species is not conducted on a routine basis. However, a comprehensive report on non-target species was prepared in 2010 for a meeting of the Joint Tuna RFMOs (Table 30). All taxa of finfish caught by longliners and recorded by observers were tallied for a 16-year period and ranked according to aggregate catch weight, and the non-Scombrid and non-billfish bony fishes comprised 10.7 percent of the total longline catch (SPC-OFP, 2010). When the data were partitioned by type of longline fishery, opah ranked highest of the taxa shown in Table 30 in deep-set and albacore longline fisheries; and ocean sunfish, followed by dolphinfish and oilfish, ranked highest in shallow-set longline fisheries.

TABLE 30
Major bony fish species other than Scombrids and billfishes caught by longliners in the Western Central Pacific Ocean, 1994–2009 (aggregated)

Rank	Taxa	Scientific name	Observed catch	
			Tonnes	*%*
8	Opah	*Lampris guttatus*	1 330	2.8
11	Escolars	*Gempylidae*	805	1.7
14	Dolphinfish	*Coryphaena hippurus*	565	1.2
15	Ocean sunfish	*Mola mola*	499	1.1
18	Pomfrets	*Bramidae*	346	0.7
21	Lancetfishes	*Alepisaurus* spp.	264	0.6
22	Butterfly kingfish	*Gasterochisma melampus*	223	0.5
23	Shark sucker	*Remora* spp.	213	0.5
24	Oilfish	*Ruvettus pretiosus*	207	0.4
30	Other fish	*Osteichthys*	590	1.2
				10.7

Note: Rankings are based on the top 30 finfish including tunas, billfishes, sharks and rays, which are not shown here.
Source: SPC-OFP (2010).

Nominal catch rates for these species varied by fishery and area, but no clear time trends were observed. Discard rates across the different longline fisheries were 4–18 percent for dolphinfish, 3–50 percent for opah, 23–73 percent for oilfish, and 48–98 percent for ocean sunfish. It was estimated that, overall, 46 percent of the landed bony fishes other than Scombrids and billfishes are dead when hauled to the vessel (SPC-OFP, 2010).

Small island developing States have identified dolphinfish, rainbow runner (which is mainly caught in the purse seine fishery) and wahoo (*Acanthocybium solandri*, a Scombrid) as particularly important to sustainable livelihoods (WCPFC Resolution 2005-03). Recent concerns regarding the impact of WCPFC tuna fishing on key food-fish stocks have been addressed primarily in terms of analysis of purse seine bycatch (Pilling *et al*., 2013). In WCPFC longline fisheries, these three species together comprise about 2.1 percent, and on average about 60 tonnes per year (SPC-OFP, 2010), as compared with the purse seine fishery where catches of these three species are on average about 3 200 tonnes per year (Pilling *et al*., 2013).

In the eastern portion of the WCPO, the pelagic longline fleet based in Hawaii lands dolphinfish, opah, pomfrets and oilfish, and discards longnose lancetfish and snake mackerel (Gilman *et al*., 2012). Using data from this fishery, a preliminary analysis of the stock status of opah was recently conducted after observing a decline in standardized catch rates from 1999 to 2004 without a subsequent recovery (NOAA, 2012f). This analysis estimated that catches of foreign longline vessels operating around Hawaii are likely to be higher than those reported by the Hawaii-based fleet but exact catch quantities are unknown. It was noted that the major change in catch rates corresponded with a change in bait type from saury prior to 2000 to saury, sardines or both after 2000. Although no direct linkage between catch rates and bait types was found, it is possible that the change in bait types was part of a broader change in operations in the fishery, and some other aspect of that change may have influenced catch rates.

6.1.4 Indian Ocean

Under IOTC catch and effort reporting requirements in effect as of 2012 for longliners, the catch of bony fishes other than Scombridae and billfishes should be recorded as a single amount (IOTC Resolutions 12/03 and 13/03). Given this requirement it can be expected that national catch records submitted to the IOTC will not contain taxa-

specific catch quantities for other bony fishes. Moreover, as discussed in Section 1.2.1, the IOTC observer programme was established only in 2010 and has to date received data for only a small number of trips. Therefore, while the observer programme is expected to have generated some data on catches of other bony fishes, the quantity of data is small and unlikely to provide meaningful insights into status and trends of these species.

A recent review of bycatch in IOTC fisheries described three countries that have provided characterizations of longline catch and bycatch to the IOTC Working Party on Bycatch based on research cruises (Ardill, Itano and Gillett, 2013). The first of these was an experimental longline cruise conducted by Spain in 2010. The catch of other bony fishes was reportedly composed of lancetfishes and ocean sunfish. The second was a longline cruise conducted off Réunion in 2007, which found that 38 percent of the catch was non-target species, and that of these 37 percent were dolphinfish, i.e. 14 percent of the total catch in number. This study also documented that the number of dolphinfish recorded in logbooks was slightly less than the number recorded by observers but comparable with the recording rates for target species. Other non-target taxa showed much higher discrepancies between the two data sources. The study also concluded that species such as pomfrets, lancetfish, oilfish and escolar were discarded dead in most cases (Bach *et al*., 2008). The third study profiled South African longline fisheries. It was stated that, in this fishery, live billfishes are required to be released and that oilfish and escolar probably constituted more than 70 percent of the "other" bycatch, with dolphinfish accounting for 10 percent (Ardill, Itano and Gillett, 2013).

6.2 MANAGEMENT AND MITIGATION MEASURES

As described in the preceding regional characterizations, there are few data on the catches of other bony fishes and, thus, little basis for implementing management measures. Some t-RFMOs have adopted resolutions calling for reduction of dead discards, either as measures calling for live release (e.g. IATTC, 2012a) or through catch retention plans (e.g. WCPFC CMM 2009-02). However, these measures are all aimed at purse seine fisheries and at reducing discards of small target tuna species. For longline fisheries, it appears that management concerns for other bony fishes are not sufficient to warrant management measures for these species *per se*, although measures aimed at managing longline fisheries as a whole would be expected to influence catches of other bony fishes to some extent.

Given the importance of dolphinfish to local economies and sustainable livelihoods in the Caribbean, the migratory nature of the species, and the variety of fleets that catch it, a regional approach to assessment and management has been advocated for some time (Mahon and Oxenford, 1999). Dolphinfish in the Caribbean probably serve as a model for this and other species of bony fishes in other regions that are caught incidentally in tuna longline fisheries. Reasons why such species require further attention include: (i) a tendency to aggregate, which makes them prone to depletion; (ii) marketable or usable size is smaller than the size at maturity; (iii) high interannual variability suggests a close link between recruitment and environmental conditions; and (iv) data-poor and/or uncertain fishery statistics hamper monitoring of stock status (Mahon and Oxenford, 1999). Despite this rationale, bony fishes lack the commercial value of tuna and billfish target species, as well as the inherent vulnerability of taxa such as elasmobranchs, sea turtles, seabirds and marine mammals. As a result, non-target bony fishes have languished without either effective monitoring or management for some time. Where possible, particularly given the importance of these species to some local communities, basic fishery monitoring systems should be expanded to include these species, and simple reviews of stock status should be periodically conducted.

6.3 CONCLUSIONS REGARDING OTHER BONY FISHES

In addition to the 64 species of Scombrids and billfishes, there are at least 650 species of other bony fishes that may be caught in association with pelagic longline fisheries targeting tuna and tuna-like species. Catch data for these species are usually limited owing to a combination of: a lack of reporting requirements; low commercial value and frequent discarding; a supposed low vulnerability to fishing pressure; and/or the difficulties of species identification. As a result, most information is qualitative or based on short-term, single-fleet surveys.

Based on a review of information for the Atlantic, Eastern and Western Central Pacific and Indian Oceans, the most commonly encountered non-target bony fishes are dolphinfish, opah, oilfish, escolar and ocean sunfish. Most of these species have not been assessed for the IUCN Red List, and the small number that have been assigned IUCN Red List "threatened" status are unlikely to interact with pelagic longline fisheries in large numbers. Dolphinfish appears to be the greatest concern in the Atlantic, with the only stock assessment having been conducted there (southeastern United States of America), the Caribbean countries calling for regional management, and a recent analysis of bycatch by Spanish Mediterranean fleets. Dolphinfish also appears to be the non-target bony fish species recorded in the greatest quantities in the IATTC convention area with catches of 2 500–4 500 tonnes annually in recent years. In the WCPO, at least ten taxa of non-Scombrid and billfishes bony fishes comprise 10.2 percent of the longline catch. Dolphinfish are usually retained, opah and oilfish are sometimes retained, and ocean sunfish are usually discarded. A sudden decline in catch rates in opah in 2000 around Hawaii prompted analysis of the potential factors driving this shift, but the reasons for the decline remain unclear. A number of disparate studies in the Indian Ocean suggest that, in addition to the five key bony fishes listed above, lancetfishes and pomfrets are also commonly caught. Nevertheless, IOTC data holdings remain extremely limited owing to a lack of longline catch reporting requirements for these species and delays in effectively implementing an agreed longline observer programme.

There are growing concerns about the value of bycatch species such as dolphinfish to local food supplies and a desire to ensure that commercial fisheries do not affect the livelihoods of artisanal fishers. However, the main focus of these concerns is the purse seine fishery, and the relative catch in longline fisheries appears to be low. Despite their naturally high productivity, several characteristics of the key non-target bony fishes reviewed here make them vulnerable (i.e. aggregation, high interannual variability, and exploitation prior to maturity). Moreover, owing to poor data quality and lack of stock assessments, it is unclear if current levels of interactions with tuna longline fisheries are placing either particular bony fish populations, or ecosystem structure or functions, at risk. Therefore, despite the lack of a commercial or pressing conservation mandate to manage these resources, more attention should be paid to improving fishery statistics and to conducting periodic monitoring of stock status indicators.

7. Synthesis of status and prospects for bycatch mitigation in longline fisheries

The diversity of pelagic longline gear designs and fishing methods, the variety of habitats they are deployed in, the thousands of marine species they may interact with, and the different mechanisms and behaviours that govern those interactions, provide an almost overwhelming array of topics to be addressed in any discussion of bycatch mitigation. In order to organize the large quantity of information, this paper has taken a taxonomic approach and discussed elasmobranchs, sea turtles, seabirds, marine mammals and other fishes in separate chapters. However, on any given pelagic longline set, interactions will occur simultaneously across taxa, and factors that mitigate impacts for some species may exacerbate impacts for others.

This section attempts to draw together the preceding chapters by discussing common themes in approaches to effective bycatch mitigation in longline fisheries. As with any meta-analysis or case study approach, it is critical to only compare like with like. It is thus tempting, given the myriad methods and experimental designs reviewed, and the range of results produced, to conclude that almost all findings are case-specific and cannot be generalized. At the other end of the spectrum, it is also important to avoid inappropriate extrapolation and oversimplification in the desire to identify commonalities. Finding the pathways towards effective bycatch mitigation solutions will involve striking a balance between these two extremes, and building on existing information to identify new fishery-specific solutions. In doing so, there will be both scientific and technical issues, as well as policy issues, that require further exploration. These issues are highlighted in the following sections.

7.1 SCIENTIFIC AND TECHNICAL ISSUES IN BYCATCH MITIGATION

7.1.1 Cross-taxa effects

Many of the mitigation studies reviewed in this paper have been motivated by a desire to mitigate the impacts of longline fisheries on a particular type of organism, e.g. circle hooks for sea turtles (Section 3.4.1), night setting for seabirds (Section 4.3.2.3), and weak hooks or physical barriers for marine mammals (Sections 5.4.1 and 5.4.2). However, each of these mitigation techniques may have either positive or negative effects on catch rates for other bycatch taxa or for target species.

For example, in the first of these three specific examples, concerns have been expressed in some fisheries that, while large circle hooks may reduce sea turtle hooking rates, they may also increase catch rates for sharks, including the threatened oceanic whitetip shark (Gilman, *et al.*, 2012). In fact, one study of fisher attitudes in Ecuador found that increased shark catch rates may be a strong factor in the uptake of circle hooks by some fishers (Mizrahi, 2012). On the positive side, it has been suggested that, in addition to protecting sea turtles, wider circle hooks may reduce seabird bycatch by up to 80 percent (Gilman, 2011). In the second example, setting pelagic longlines at night, in order to avoid the times when most seabirds are actively foraging, has proved effective, especially on dark nights. This type of measure is also expected to mitigate sea turtle interactions and improve tuna catch rates (Gilman *et al.*, 2012). However, restricting the time of line setting in this way could have major repercussions for the

economic performance of longline fisheries that traditionally set during daylight hours (i.e. most tuna fisheries; Beverly, Chapman and Sokimi, 2003). In the final example, the results of weak-hook trials designed to allow "self-release" of large marine mammals indicated that the design worked as intended towards marine mammals but maintained catch rates of target species (Bigelow *et al*., 2012). This finding appears to have been an important part of the decision to implement weak hooks in the fishery (NOAA, 2012d).

Although some studies have taken a holistic approach to evaluation of mitigation measures, others have focused only on the results for a single species group and have not addressed impacts to other bycatch taxa or to target species. This may be due to the experimental design of the study, and/or because commercial operations were not or could not be replicated. While such limitations are expected to continue, it would be useful if future studies explicitly discussed impacts on other bycatch and target taxa and stated whether there are any conclusive findings for these taxa (e.g. stating that the experimental design may not cater for all taxa or that some taxa that were catered for may, owing to low catch rates, not be eligible for statistical analysis). This kind of clear presentation of results will not only better guide future research but also provide greater clarity in the development of international guidance documents, and national regulations and plans of action.

RECOMMENDATION 1
Cross-taxa effects

Future studies of mitigation measures should consider effects across the range of potential bycatch and target taxa, if possible, and clearly state which taxa the study addressed and whether findings are or are not conclusive for these taxa.

7.1.2 Combinations of mitigation measures

Another dimension of bycatch mitigation studies that has yet to be fully explored is how combinations of measures work together. This situation may occur by design in a fishery with single-taxa bycatch issues, e.g. wide circle hooks in combination with fish bait to mitigate sea turtle interactions (Section 3.4.2), or streamer lines and weighted branch lines to mitigate seabird interactions (Sections 4.3.2.1 and 4.3.2.2). More probably, it may occur where one mitigation measure is implemented for one taxa and another measure for another taxa (e.g. circle hooks for sea turtles in combination with nylon leaders for sharks, Section 2.4.2). It is often not possible at the time of initial testing to foresee which other mitigation measures may be operating in tandem, and testing multiple factors may complicate interpretation of results.

In the first example (i.e. circle hooks and fish bait for sea turtles), hook type and bait type were one of the earliest combinations to be studied together with the first papers published in the early 2000s. The reason for this may have been that fishers select certain types of hooks in part due to the ease of baiting (Watson *et al*., 2005). In this case, it was quickly identified that finfish baits lowered sea turtle interaction rates on both J and circle hooks. However, despite this finding being replicated in a number of other fisheries (Gilman, 2011), this combination is not yet required by any of the t-RFMOs (Section 3.3). In the second example as well, there is some evidence that weighted branch lines and paired streamers work together to both sink hooks faster and to better protect sinking hooks from diving seabirds, respectively (Dietrich, Melvin and Conquest, 2008), but more demonstration trials may be needed to promote these techniques to fishers. In the third example (i.e. circle hooks for sea turtles and nylon leaders for sharks), studies have found that shark catch rates on nylon leaders are likely to be more similar to catch rates on wire leaders when circle hooks are used

because the circle hook is more likely to embed in the corner of the jaw and the animal is unable to bite through the leader and escape regardless of leader material (Ward *et al*., 2008; Afonso *et al*., 2012; Gilman, Owens and Kraft, 2013). As a result, the effectiveness of either measure alone cannot be expected to be maintained when they are used in combination. As well as better definition of hook-type properties (Sections 2.4.2 and 3.4.1), other common combinations of mitigation measures that require further study are the operational factors of hook depth, soak time and time of the set. Not only are there expected inter-relationships between these factors that need to be tested in specific fisheries, there is an emerging understanding that even within species groups such as sharks there are important differences in diel behaviours (Musyl *et al*., 2011; Bromhead *et al*., 2012) that could affect interaction rates.

Some studies have done better than others in documenting all of the operational conditions under which the mitigation measure was tested. Nevertheless, better documentation of gear and operating characteristics in all studies, whether intentionally varied (e.g. leader type, hook type), held constant (e.g. time of set), or unable to control (e.g. wind), will assist in generalizing the results. Given what is being discovered regarding differences in the effect of leader type and hook type on catch rates in similar fisheries (Bromhead, Rice and Harley, 2013; Gilman, Owens and Kraft, 2013), it is likely that fishery-specific trials will always be desirable to confirm effectiveness but may not always be practical. Therefore, making maximum use of the results of existing studies is essential.

RECOMMENDATION 2
Combinations of mitigation measures

Future studies of mitigation measures should better document which potentially significant explanatory variables (e.g. gear and operational conditions) were controlled for in experiments, and explicitly accounted for in the analysis, in order to correctly report single factor effects and assist in appropriately generalizing to other fisheries.

7.1.3 Economic and safety considerations

Only some of the studies reviewed in this paper explicitly considered how the implementation of the mitigation technique (or techniques) would affect the profitability of the fishing trip. As discussed above, impacts on the catch rates of target species are one component of this kind of assessment, but prolonged set and hauling times, as found in some of the marine mammal hook barrier methods (Section 5.4.2), or high rates of fouling of longline floats and streamer (tori) lines for seabirds (Section 4.3.2.1), would also affect economic performance. Moreover, high risk of crew injury is an indirect cost faced by fishing operations. Therefore, crew safety when using safe release methods (e.g. for sharks; Section 2.4.4) and equipment such as weighted branch lines (Section 4.3.2.2) will be an important determinant in the readiness with which fleets will adopt new techniques.

In recent studies of physical barriers to marine mammal depredation of hooked fish, deployment time increased four- to twelve-fold and retrieval time was also prolonged owing to entanglement and water resistance (Rabearisoa *et al*., 2012). A similar study avoided these operational impacts but required an additional crew member (Hamer and Childerhouse, 2012). Although it is likely that both approaches will entail additional costs, these costs may be compensated by the enhanced value of catch that has not been damaged by depredation. In contrast, fouling of streamer (tori) lines with standard pelagic longline floats will represent a cost (Melvin, Guy and Read, 2010), in time if not in broken gear, which is only offset by the non-monetary value of reduced seabird bycatch. With regard to impacts on crew safety, although these impacts are usually

acknowledged (Løkkeborg, 2011), there appear to be few studies that have addressed the additional risks of injury posed by different forms of shark handling, branch-line weighting or other mitigation devices.

Although some of the remaining obstacles to implementation of bycatch mitigation techniques are scientific in nature, it is likely that larger gains will be made by directing future efforts toward residual economic and safety issues. These issues may be difficult to explore in a research environment as they will relate to actual conditions on board commercial vessels. Therefore, given that the theoretical effectiveness of many techniques is already well documented, the focus of bycatch mitigation efforts would usefully shift towards demonstration trials on board commercial vessels conducting normal fishing operations. Such demonstration trials would be a useful way of both gathering data on economic and safety performance and introducing new techniques to fishers. Such trials would also be expected to lead to further refinements that improve effectiveness, reduce costs/risks, or both.

RECOMMENDATION 3
Economic and safety considerations

In order to overcome economic and safety concerns associated with mitigation measures, the focus should shift towards demonstration trials on board commercial vessels conducting normal fishing operations.

7.1.4 Understanding biological mechanisms

In addition to further scientific study of bycatch mitigation measures across taxa, in combinations, and as they relate to economics and safety, it will also be useful to continue basic biological research into why and how mitigation measures are effective. Over time, fishers have identified effective ways of catching target species, and science is now, in an accelerated manner, attempting to discover ways of not catching bycatch species. This discovery is in many cases aided by advances in physiology and sensory biology, as demonstrated by studies of the shark-repellent effects of magnets and electropositive metals (Section 2.4.3), sea turtle photosensitivities and preferences by species (Section 3.4.5), and research into the role of echolocation in the behaviour of depredating marine mammals (Sections 5.4.2 and 5.4.3). These studies have the potential to both improve existing mitigation concepts (e.g. refining methods for particular species or areas) as well as discover new techniques. Ecological studies of preferred habitats and their temporal and spatial overlap with fishing grounds may also provide new opportunities for mitigation (Sections 3.4.4 and 4.2).

In the case of elasmobranchs, the vast number of species that may interact with pelagic longline fisheries, and their expected differences in sensory capabilities and feeding behaviour and ecology, suggests that a single approach will not be effective in all cases (Jordan *et al*., 2013). However, as more becomes known about species-specific differences in prey location, e.g. hammerheads rely heavily on electrosensitivity, whereas makos preferentially use vision, and blue sharks use olfaction (Hutchinson *et al*., 2012), the opportunities for effective mitigation will increase. For sea turtles, sensory-system research for use in pelagic longline mitigation (Crognale *et al*., 2008; Southwood *et al*., 2008) appears to have slowed owing to a recent focus on circle hooks and finfish baits as best practice techniques. Nevertheless, sea turtle sensory research in net-based fisheries continues (Wang, Fisler and Swimmer, 2010) and may discover transferable techniques. In order to deter marine mammals from depredation, mesh structures that deploy over hooked fish and mimic entanglement are being tested (Hamer, Childerhouse and Gales, 2012). However, it is not clear whether these structures are perceived visually or through sonar, and incorporating metallic elements

to further increase sonar disruption is being pursued (McPherson and Nishida, 2010). Further studies of the role of habituation in marine mammal responses to acoustic deterrents are likely to be pivotal to the success or failure of these techniques (Mooney et al. 2009). In terms of understanding habitats, most habitat-based mitigation measures implemented to date are based on static boundaries (Sections 3.3, 4.3.1 and 5.3), and past efforts to develop more dynamic tools mainly for sea turtles (Howell et al. 2008, Braun-McNeill et al. 2008) have recognized the complexity of the task.

Given the urgency of mitigating some bycatch interactions, sensory and habitat research may seem to represent too indirect an approach. However, in parallel with improving uptake of existing mitigation techniques, it is critical to understand the biological mechanisms that make certain mitigation measures succeed where others fail. Deeper insights into these mechanisms hold the key to improving effectiveness and reducing costs. While habitat-based measures such as closures have had limited success in longline fisheries thus far (Senko *et al*., 2013), rather than discouraging further habitat research, additional studies should be conducted both to improve habitat-based mitigation measures, as well as to help understand variability in gear- or set-based mitigation measures.

RECOMMENDATION 4
Understanding biological mechanisms

Further research into bycatch organisms' sensory biology and habitats should be conducted as a basis for improving existing mitigation measures as well as to generate ideas for new, more effective techniques.

7.1.5 Handling and post-release mortality

Most studies reviewed in this paper have focused on reducing bycatch interactions. This focus on relative (i.e. as compared to without mitigation) interaction rates does not address the total mortality rates arising from bycatch interactions with longline fisheries. In particular, totality mortality rates will depend on not only the hooking rates, the hauling mortality rates and the handling mortality, but also on the post-release mortality rates. This issue is perhaps most important for sharks (Section 2.4.4), and less important for sea turtles, seabirds and marine mammals, whose survival rates are more easily read from their condition at the point of release (Sections 3.1.2, 4.1.1 and 5.1.2). However, in addition to acute mortality resulting from trauma, there is also the possibility of delayed mortality resulting from chronic injuries such as embedded hooks or entanglement wounds or infections (Sections 3.1.2, 4.1.2.5 and 5.1.2).

Research has shown that the survival of sharks interacting with pelagic longline fisheries varies by species (Musyl *et al*., 2011), size and sex (Coelho *et al*., 2012), as well as according to the handling practices applied in the fishery (Musyl *et al*., 2009, Campana *et al*., 2009). Therefore, the effectiveness of no-retention mitigation measures, such as those for the oceanic whitetip shark, may vary considerably between fleets and may not result in sufficient mortality reduction for this species (Clarke, 2013). Concerns have also been raised about the long-term consequences of embedded hooks and trailing branch lines left in place upon release of sea turtles from longline vessels (Parga, 2012).

The extent to which handling mortality, which can be substantially affected by human behaviour, determines overall mortality rates as opposed to hooking mortality, which is more influenced by factors such as hooking rates, gear configurations and soak time, should be the subject of further research. In addition, post-release mortality warrants further attention in order to quantify the total mortality resulting from the encounter. Given that there is a range of potential mortality rates spanning from zero

to unmitigated levels, it is not necessarily the case that current best practice mitigation will achieve the mortality reduction necessary to maintain populations of bycatch at sustainable levels. Total mortality rates need to be quantified in order to understand whether the mortality rates achieved under best practice mitigation require further reduction.

RECOMMENDATION 5
Handling and post-release mortality

For some bycatch taxa, particularly sharks and sea turtles, handling and post-release mortality should be investigated in order to quantify the total mortality rates to populations resulting from longline fishery interactions.

7.1.6 Non-fishery impacts

Just as it is necessary to estimate the total mortality associated with pelagic longline fisheries in order to understand the impact of this fishery on each population, so it is also necessary to estimate the total mortality to the population from all sources in order to understand whether the impacts are sustainable. For almost every species considered in this study, impacts from pelagic longline fisheries comprise only part of the total population impact. For elasmobranchs and other bony fishes, other fisheries such as purse seines, trawls and gillnets also affect populations caught by longline fisheries. Sea turtles are affected by interactions with other fisheries as well as climate change, human take of meat and eggs for consumption, coastal development and pollution (Section 3.1.1). Seabirds are also taken by fisheries other than longlines, as well as affected by human disturbance, predation, climate change and pollution (Section 4.1.3). Longline fisheries' threats to marine mammals (primarily hooking) are noteworthy but minor compared with those associated with vessel strikes and entanglement (mainly in gear other than longlines [Section 5.1.2]).

Despite the fact that research into the relative severity of various impacts to these organisms is highly uncertain owing to lack of data, some information on comparative threats is available. Analysis of observer data for elasmobranchs designated as key species by the WCPFC found that the longline fishery catches more than ten times as many of the key shark species as does the purse seine fishery (Lawson, 2011). Global studies of both sea turtles and seabirds suggest that the primary threat to these species is fisheries bycatch (Wallace *et al*., 2011; Croxall *et al*., 2012), although in the case of most populations of sea turtles, longline fisheries were shown to have lower overall impact scores than fisheries using nets and trawls (Wallace *et al*., 2013). Marine mammal interaction rates with longline fisheries are low compared with other bycatch species (Arrizabalaga *et al*., 2011), but localized interaction rates may be high enough to cause concern for species with a high risk of extinction, such as the false killer whale off Hawaii (NOAA, 2012c). Other bony fishes caught by longline fisheries are also affected by other gear types either as target species (e.g. dolphinfish [Mahon and Oxenford, 1999]) or as bycatch (e.g. opah [NOAA, 2012e]).

This discussion is not intended to diminish the importance of longline fishery impacts on these populations. However, focusing this paper on pelagic longline fisheries only tends to distract from the need to develop holistic solutions to mitigating population impacts to sustainable levels. To some extent, the t-RFMOs perpetuate single-issue solutions by conducting assessments and adopting mitigation measures for fishery impacts only, without attempting to quantify other sources of mortality such as total take and habitat disturbance. It may be that a mechanism with a broader remit (e.g. the CMS) is better placed to develop these holistic solutions. Nevertheless, in the short-term, t-RFMOs can improve their own management systems by compiling data

from members on the full range of fishery and non-fishery impacts and by making a start on full spectrum impact assessments.

RECOMMENDATION 6
Non-fishery impacts

Although fishery interactions, and in many cases longline fishery interactions, are often the primary source of impacts to bycatch populations, non-fishery impacts arising from other take, habitat modification, marine pollution, etc. should be assessed as part of a holistic approach to impact mitigation.

7.2 POLICY ISSUES IN BYCATCH MITIGATION

7.2.1 Need for comprehensive management

Throughout the preceding chapters, findings have been caveated by deficiencies in existing data. Most of the t-RFMOs hold only a fraction of the potentially available shark interaction data and almost no data on sea turtles, seabird or marine mammal interactions. In addition, the vast majority of the data held by the t-RFMOs are not in the public domain, thereby limiting access for analysis. With the exception of some shark populations for which ecological risk assessments or stock assessments have been conducted, it would be difficult to argue that t-RFMOs are currently managing, or even assessing or monitoring, most bycatch taxa.

There are multiple reasons for this situation. In some cases, t-RFMOs face debates among members concerning whether the t-RFMO has the mandate to manage bycatch organisms. In most t-RFMOs, the responsibility for collection of bycatch information has been delegated to observer programmes, many of which fail to achieve effective or required coverage rates. Compounding these issues, minimal requirements for submission of bycatch data are not complied with, and shortcomings in t-RFMO compliance systems (e.g. Koehler, 2013) allow these non-compliances to continue.

Under the current scenario, there are nevertheless several ways that bycatch assessments are undertaken. The t-RFMOs have sponsored fishery-specific ecological risk assessments and population-based stock assessments for a number of shark species (blue, mako and porbeagle in the Atlantic; oceanic whitetip, silky and blue sharks in the WCPO; and silky sharks in the EPO). In some cases, these assessments have struggled to overcome data gaps. Although ICCAT and the IOTC are currently undertaking assessments for sea turtles, in addition to also being hampered by data gaps, these assessments are expected to address only fishery-related threats. Thus, they will not be able to draw conclusions about what mitigation measures for fisheries might be necessary to reduce total threats to sea turtles to sustainable levels. For sea turtles and seabirds, the only studies that have addressed threats to these populations holistically have been meta-analyses based mainly on limited public domain data (e.g. Wallace *et al*., 2013; Croxall *et al*., 2012). While such studies are useful, they are conducted outside the fishery management framework and, thus, it is unclear how or if they will be used to frame management measures for fisheries.

Given the status of bycatch populations and the continuing multifaceted threats to their sustainability, the need for comprehensive assessment and management is clear. The question is thus whether the t-RFMOs will accept this responsibility, or whether other organizations should take on a leadership role. On one side of the argument will be those who state that t-RFMOs do not have the mandate and/or the resources to manage bycatch. On the other side, however, is the argument that fisheries appear to constitute the greatest threat to elasmobranchs, sea turtles and seabirds, and thus these threats are most appropriately mitigated through fisheries management via the

t-RFMOs. In the absence of a clear way forward on this major issue, the t-RFMOs' Kobe Bycatch Working Group is directing its efforts toward harmonizing observer data collection fields for bycatch, and the production and dissemination of identification guides. These initiatives are valuable but leave unanswered the critical question of how, when and by whom comprehensive management of bycatch populations will be initiated.

RECOMMENDATION 7
Need for comprehensive management

Whether or not t-RFMOs accept the leading role for comprehensive management of bycatch populations, they should: (i) strengthen their bycatch data collection and assessment programmes to provide scientifically robust data to holistically address the status of bycatch populations; and (ii) adopt and effectively implement mitigation and management measures that can protect bycatch populations as a whole.

7.2.2 Dissemination and implementation strategies

Most of the studies reviewed in this paper describe the development and experimental testing of mitigation measures. Comparatively few describe the implementation and effectiveness of these measures in commercial pelagic longline fisheries. This may be in part because scientists have focused their efforts on research rather than on policy or compliance issues. It is also no doubt due to the fact that few of the mitigation techniques described in this paper have yet been formally implemented, and for even fewer are performance data available in the public domain.

Successful transfer of mitigation measures from an experimental setting to commercial fisheries involves two components: (i) introduction and refinement of techniques; and (ii) follow-through for continued, effective use. In the first phase, as highlighted by Cox *et al*. (2007), the information flow should not be one-way, but should encourage feedback from fishers in order to create a situation of genuine collaboration. Other elements that are likely to promote successful uptake include allowing time for new techniques to be phased into use, and providing low-cost or no-cost mitigation equipment. For pelagic longline fisheries, many of the techniques described in the preceding chapters have been developed in conjunction with commercial fishers (e.g. streamer lines [Section 4.3.2.1], and double line weighting, [Section 4.4.4]). Although these techniques have yet to achieve full-scale implementation, their origins in fishers' innovations should facilitate this process. Other examples of facilitating uptake of new mitigation techniques are based in economics. For example, a programme offering new circle hooks for old J hooks in at least nine Eastern Pacific countries encouraged mitigation of sea turtle interactions by removing the gear cost associated with the switch (IATTC, 2012a; Andraka *et al*., 2013). Similar economically motivated approaches may involve offering market-based incentives for products that either meet fishery certification standards (e.g. Marine Stewardship Council certification) or industry-based codes of conduct (e.g. ISSF Conservation Measures [ISSF, 2013]). It should be noted that the effectiveness of these certification and code-of-conduct approaches in ameliorating bycatch issues will depend on the extent to which the requirements of these programmes conform to internationally accepted best-practice mitigation standards and the improvements necessary in each particular fishery. In contrast to these voluntary approaches, some of the mitigation measures for mammals (e.g. weak hooks [Section 5.4.1]) and sharks (e.g. no-retention [Section 2.3.2]) have been implemented through regulations (i.e. national and t-RFMOs). The weak hooks technique was subject to a United States government federal rule-making process and testing in the fishery of concern before implementation in December 2012 (NOAA,

2012d), whereas the no-retention measures had no prior consultation with fishers and lack guidelines for shark handling (Clarke, 2013).

Once mitigation techniques have been introduced, in the case of voluntary measures, or mandated, in the case of regulatory decisions, there is still the need to ensure they are continuously and effectively used. This follow-through can take the form of compliance monitoring, incentives, or a combination of the two. Compliance monitoring is the more straightforward way of assessing implementation rates and overall effectiveness, but it requires adequate resources to be successful. Observer programmes can provide excellent data, but many have failed thus far to meet required coverage levels (Section 1.2.1). Recent developments in the field of electronic monitoring may be able to assist in this regard. However, it should be noted that the coverage of the data collected through electronic monitoring may be expanded at the expense of its resolution (e.g. electronic monitoring may reveal the number of seabirds hooked but perhaps not their species) when compared with observer programmes.

Incentives provide an alternative or complementary means of implementing effective mitigation. Again, given the small number of mitigation measures that have been implemented thus far for pelagic longline fisheries worldwide, there are few examples of successful incentives profiled in this paper. Perhaps the prime example for future implementation would be the physical barriers to prevent depredation of hooked fish by marine mammals (Section 5.4.2). As depredation costs associated with damage to target species, bait loss and gear breakage are estimated at more than US$400 per 1 000 hooks, avoiding such losses would provide a major incentive to use marine mammal interaction mitigation techniques, even if those techniques come at a cost. For sea turtles and seabirds, interaction costs include loss of bait and gear, as well as the opportunity cost of lost target species. Therefore, while arguably less than the costs of marine mammal depredation, there are likely to be economic incentives to mitigating sea turtle and seabird interactions as well. There is probably the least amount of economic incentive to avoid shark interactions because in most areas there are lucrative markets for shark products, in particular shark fins, and thus the only cost to catching sharks is the opportunity cost of not catching a target species. Aside from these types of economic incentives, and as noted by Cox *et al*. (2007), in any profitable fishery with individual licence limits on bycatch or fishery-wide bycatch thresholds, there is an inherent incentive to comply in order to avoid licence revocation or fishery closure.

Effective implementation of bycatch mitigation measures can be challenging when fishers resist the measures and either refuse to implement them voluntarily or refuse to comply with regulations. A programme of work designed under the Global Environment Facility's Areas Beyond National Jurisdiction project aims to achieve implementation of agreed seabird mitigation measures by 40 percent of the longline vessels fishing in the ICCAT and IOTC areas within five years. The approach involves conducting at least two sea trials demonstrating the measures in regional commercial fishing fleets, as well as initial informational workshops and pre-cruise coordination meetings, post-cruise feedback workshops, and fleet and t-RFMO results dissemination meetings. The outcomes of this project may serve as a useful model for the implementation of other bycatch mitigation measures in other t-RFMOs.

RECOMMENDATION 8
Dissemination and implementation strategies

Researchers and fisheries managers should look beyond developing and testing mitigation techniques to devising effective implementation strategies involving elements of fisher collaboration, compliance monitoring, and incentives.

7.2.3 Benefits and limitations of gear modification

Selecting the most appropriate bycatch mitigation measure requires weighing up both technical and policy considerations. Options that appear to be optimal from a technical and theoretical perspective may have associated cost or operational issues that impede their implementation and limit their overall effectiveness. In contrast, options that are inexpensive and unobtrusive may not sufficiently reduce interaction rates. Moreover, options that have proved successful in some fisheries may fail when implemented under other circumstances. All of these factors, plus the wide diversity of longline fisheries and the lack of compliance and monitoring data, make it difficult to draw conclusions in this paper regarding which type of bycatch mitigation works best.

A recent study posed this question by comparing gear modifications, closures, bycatch limits and buy-outs for three bycatch species caught by a variety of fisheries. Although it was concluded that gear modifications were the most promising and most commonly used techniques, effectiveness varied from case to case (Senko *et al*., 2013). Gear modification techniques also dominated this paper (Sections 2.4, 3.4, 4.3, 4.4 and 5.4) including hooks, baits, branch lines, deterrents, repellents, streamer lines, line shooters, hook pods and hook sleeves. Many of these techniques remain under development, but circle hooks and streamer lines have been implemented in a number of fisheries (Sections 3.4.1, 5.4.1 and 4.3.2.1). Some t-RFMOs have adopted lists of approved gear modification measures for sea turtles and seabirds in high-risk areas or fisheries and require that a minimum number of these measures be applied (Sections 3.3 and 4.3.3). Although examples of closures in longline fisheries are more limited, these have been invoked for elasmobranchs in the form of "shark sanctuaries" (Section 2.3.4), for sea turtles in past closures of Atlantic fishing grounds (Section 3.3) and to protect the Hawaiian monk seal (Section 5.3). Bycatch limits, which can be considered a form of triggered closure, were illustrated by the United States approach to false killer whale bycatch in the Hawaii longline fishery (Section 5.3), sea turtles in the Hawaii longline fishery (Section 3.3), as well as other management examples from the United States of America (Atlantic) and Australia (Section 3.3). Similar approaches could be pursued for sharks by adopting reference points and/or harvest control rules to maintain catches within sustainable levels. No examples of buy-outs were reviewed for this paper.

As discussed above (Section 7.2.2), one approach to mitigation involves introducing techniques, either through education, incentives or regulation, and then using compliance monitoring and/or more incentives to ensure continued implementation. This approach can be considered a kind of input control, i.e. managing by specifying what should be done to reduce bycatch. An alternative approach would be based on applying output controls, i.e. managing by specifying what reduced bycatch interaction rates must be achieved. This approach is illustrated by the bycatch limits that have been set for the Hawaii-based longline fishery (Sections 3.3 and 5.3). Output approaches require high levels of compliance monitoring, but more directly enlist the ingenuity of fishers in devising effective bycatch strategies. While these strategies may lead to reduced bycatch, it is not always clear what exactly fishers do to operate within these limits (e.g. Gilman, Dalzell and Martin, 2006; Howell *et al*., 2008), and thus opportunities for transferring effective techniques to other fisheries may be limited.

This review has focused heavily on bycatch mitigation involving gear modifications. This is likely to be the result of a number of factors including: (i) the large number of research studies that have addressed these topics; (ii) gear modification techniques do not necessarily require high levels of compliance monitoring (e.g. as compared with bycatch limits); and (iii) fishers may prefer this approach as it avoids fishery closures (Senko *et al*., 2013). This favouring of gear modification approaches to bycatch mitigation in longline fisheries may distract from the potential for using other types of measures such as closures, limits or buy-outs. Although these approaches are more extreme in their alteration of fishing operations, they may be necessary in some cases,

particularly when bycatch interaction rates from mitigated fishing activities cannot be reduced below sustainable levels. The lack of examples in this review does not mean that closures, limits or buy-outs are inappropriate for longline fisheries; rather, for now their advantages and disadvantages are more clearly illustrated from experience in other types of fisheries (see Senko *et al*., 2013). It should also be noted that these approaches are not mutually exclusive, and combinations can and should be explored in the search for optimal solutions.

RECOMMENDATION 9
Benefits and limitations of gear modification

Gear modification is often a favoured approach to bycatch mitigation, but when interaction rates cannot be reduced below sustainable levels, other management approaches such as closures, limits or buy-outs should be considered.

7.3 The future of bycatch mitigation

The compilation of material in this paper is intended to serve as a useful reference for the current state of bycatch and its management in longline fisheries. Much progress has been made in the past 20 years in understanding the status of bycatch populations, developing technologies that can mitigate interactions, and beginning to manage high-risk bycatch–longline fishery interactions. However, it is not clear what further gains can be had from meta-analyses of available but unrepresentative bycatch data; from technical advances in mitigation techniques that are not implemented or enforced; or from narrowly focused, species-specific management for a seemingly increasing list of threatened species.

In several years' time, it will be necessary to update this paper with the latest status and trends in longline bycatch. It is hoped that by that time reporting requirements for interactions and mortalities will have generated robust and representative data sets, held centrally and transparently. Mitigation measures will have been widely implemented, meaningfully monitored and routinely assessed for effectiveness. Finally, the overall trajectories of the bycatch populations affected by longline fishing will have been quantitatively evaluated and found to be improving. On each of these three fronts, the past era has seen technical advances such that it is now understood, in many cases, how the problems can be solved. The future challenge, and the key to achieving these outcomes, is to shift efforts toward governance, finance and outreach mechanisms to allow the technical solutions to be implemented effectively.

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This publication is the third in a series on bycatch in global tuna fisheries. Dealing with longline fisheries, its scope is defined taxonomically to comprise only non-tuna and non-tuna-like species, i.e., elasmobranchs, sea turtles, seabirds, marine mammals and other (non-target) bony fishes.

The diversity of pelagic longline gear designs and fishing methods, the variety of habitats they are deployed in, the thousands of marine species they may interact with, and the different mechanisms and behaviours that govern those interactions provide an array of topics to be addressed in any discussion of bycatch mitigation. Scientific and technical issues in mitigation including effects across taxa, effects of combinations of measures, economic and safety considerations, underlying biological mechanisms, handling and post-release mortality, and non-fishery impacts must all be addressed. In addition, it is also necessary to consider issues such as who takes the lead for ensuring mitigation is sufficient for the population as a whole, how to devise effective mitigation implementation strategies, and whether gear modification should be used in concert with more sweeping measures.

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