FAO Fisheries and Aquaculture Report No. 1032 FIRF/R1032 (En)
ISSN 2070-6987

Report of the

FOURTH FAO EXPERT ADVISORY PANEL FOR THE ASSESSMENT OF PROPOSALS TO AMEND APPENDICES I AND II OF CITES CONCERNING COMMERCIALLY-EXPLOITED AQUATIC SPECIES

Rome, 3–8 December 2012

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FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS
Rome, 2013

E-ISBN 978-92-5-107463-3 (PDF)

Preparation of this document

This is the report of the Fourth FAO Expert Advisory Panel for the Assessment of Proposals to Amend Appendices I and II of CITES Concerning Commercially-exploited Aquatic Species, held at FAO headquarters from 3 to 8 December 2012.

The meeting of the Panel was funded by FAO Regular Programme and by the Governments of Germany, Japan and New Zealand.

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FAO. 2013.
Report of the fourth FAO Expert Advisory Panel for the Assessment of Proposals to Amend Appendices I and II of CITES Concerning Commercially-exploited Aquatic Species, Rome, 3–8 December 2012.
FAO Fisheries and Aquaculture Report No. R1032. Rome, FAO. 161 pp.

Abstract

The fourth FAO Expert Advisory Panel for the Assessment of Proposals to Amend Appendices I and II of CITES Concerning Commercially-exploited Aquatic Species was held at FAO headquarters from 3 to 8 December 2012. The Panel was convened in response to the agreement by the twenty-fifth session of the FAO Committee on Fisheries (COFI) on the terms of reference for an expert advisory panel for assessment of proposals to the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), and to the endorsement of the twenty-sixth session of COFI to convene the Panel for relevant proposals to future CITES Conference of the Parties.

The objectives of the Panel were to:

- assess each proposal from a scientific perspective in accordance with the CITES biological listing criteria (Resolution Conf. 9.24 [Rev. CoP13];
- comment, as appropriate, on technical aspects of the proposal in relation to biology, ecology, trade and management issues, as well as, to the extent possible, the likely effectiveness for conservation.

The Panel considered the following seven proposals submitted to the sixteenth Conference of the Parties to CITES:

- CoP16 Prop. 42. Proposal to include *Carcharhinus longimanus* (oceanic whitetip shark) in Appendix II in accordance with Article II paragraph 2(a).
- CoP16 Prop. 43. Inclusion of *Sphyrna lewini* in Appendix II in accordance with Article II 2(a) and inclusion of *S. mokarran* and *S. zygaena* in Appendix II in accordance with Article II 2(b).
- CoP16 Prop. 44. Inclusion of *Lamna nasus* (Bonnaterre, 1788) in Appendix II in accordance with Article II 2(a).
- CoP16 Prop. 45. Transfer of *Pristis microdon* from Appendix II to Appendix I of CITES in accordance with Article II, paragraph 1.
- CoP16 Prop. 46. Inclusion of the genus *Manta* in Appendix II in accordance with Article II paragraph 2(a)
- CoP16 Prop. 47. Inclusion of the ceja river stingray (*Paratrygon aiereba*) in Appendix II in accordance with Article II paragraph 2(a).
- CoP16 Prop. 48. Inclusion of the freshwater stingrays *Potamotrygon motoro* and *P. schroederi* in Appendix II in accordance with Article II paragraph 2(a).

Contents

Abbreviations and acronyms

CCAMLR	Commission for the Conservation of Antarctic Marine Living Resources
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
CMS	Convention on the Conservation of Migratory Sharks
COFI	FAO Committee on Fisheries
CPUE	catch per unit of effort
DW	disc width
EC	European Commission
EEZ	exclusive economic zone
GFCM	General Fisheries Commission for the Mediterranean
IATTC	Inter-American Tropical Tuna Commission
ICCAT	International Commission for the Conservation of Atlantic Tunas
ICES	International Council for the Exploration of the Sea
IFS	Introduction from the Sea (provisions of CITES)
IOTC	Indian Ocean Tuna Commission
IPOA-Sharks	International Plan of Action for Conservation and Management of Sharks
IUCN	International Union for Conservation of Nature
IUU	illegal, unreported and unregulated (fishing)
NAFO	Northwest Atlantic Fisheries Organization
NDF	non-detriment finding
NEAFC	North East Atlantic Fisheries Commission
NPOA-Sharks	National Plan of Action for Conservation and Management of Sharks
QSCP	Queensland Shark Control Program
RFMO	regional fisheries management organization
SCRS	Standing Committee on Research and Statistics
SRFC	Sub-Regional Fisheries Commission
SEAFDEC	Southeast Asian Fisheries Development Center
SEAFO	Southeast Atlantic Fisheries Organization
TAC	total allowable catch
TL	total length
WCO	World Customs Organization
WCPFC	Western and Central Pacific Fisheries Commission
WPEB	Working Party on Ecosystems and Bycatch

Introduction

BACKGROUND AND PURPOSE OF THE EXPERT ADVISORY PANEL

1. The fourth FAO Expert Advisory Panel for the Assessment of Proposals to Amend Appendices I and II of CITES Concerning Commercially-exploited Aquatic Species was held in response to the agreement by the Twenty-fifth Session of the FAO Committee on Fisheries (COFI), February 2003, on the Terms of Reference for an expert advisory panel for assessment of proposals to the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), and to the endorsement of the Twenty-sixth Session of COFI to convene the Panel for relevant proposals to future CITES Conference of the Parties.

2. The FAO Panel also falls within the agreement between CITES and FAO, as elaborated in the Memorandum of Understanding between the two organizations, for FAO to carry out a scientific and technical review of all relevant proposals for amendment of Appendices I and II. The results of this review are to be taken into account by the CITES Secretariat when communicating their recommendations on the proposals to the Parties to CITES.

3. The Terms of Reference agreed to at the Twenty-fifth Session of COFI are attached to this report as Appendix A. In accordance with those Terms of Reference, the Panel was established by the FAO Secretariat, according to its standard rules and procedures and observing the principle of equitable geographical representation, drawing from a roster of recognized experts. The task of the Panel was to:

- assess each proposal from a scientific perspective in accordance with the CITES biological listing criteria, taking account of the recommendations on the criteria made to CITES by FAO;
- comment, as appropriate, on technical aspects of the proposal in relation to biology, ecology, trade and management issues, as well as, to the extent possible, the likely effectiveness for conservation.

THE PANEL MEETING

4. The Panel met in Rome from 3 to 8 December 2012, hosted by FAO with funding from the FAO regular programme and the Governments of Germany, Japan and New Zealand. The agenda adopted for the meeting is included as Appendix B.

5. The Panel consisted of a core group made of 8 members and 15 specialists on the species being considered and aspects of fisheries management and international trade relevant to that species. In addition, two invited observers attended the 2012 Panel, one from the CITES Secretariat and one from the Secretariat of the International Commission for the Conservation of Atlantic Tunas (ICCAT). The list of participants to the meeting is included as Appendix C.

6. The meeting was opened by Mr Árni Mathiesen, Assistant Director-General, FAO Fisheries and Aquaculture Department, who welcomed the participants and provided some background information to the convening of the meeting of the Advisory Panel and the importance of its task.

7. Arne Bjorge was elected Chair of the Panel. Marcelo Vasconcellos, FAO, was elected rapporteur.

8. The agenda of the meeting was adopted as tabled.

9. Johanne Fischer, FAO Senior Fisheries Officer, made a presentation on the Terms of Reference of the FAO Expert Advisory Panel and on the FAO interpretation of the CITES criteria for the inclusion of commercially-exploited aquatic species in the CITES Appendices.

10. Proponents of the seven proposals for listing on CITES Appendices were invited to present the proposals in person to the Panel and to answer any questions of clarification by Panel participants. For this purpose, the proponents were represented by the following individuals:

- Colombia and the United States of America by Mr Vladimir Puentes Granada, Mr Felipe Osorio Viera and Mr Dwayne Meadows (CoP16 Prop. 42 – Oceanic whitetip shark);
- Brazil by Mr Fabio Hazin (CoP16 Prop. 43 – Scalloped hammerhead shark, great hammerhead shark and smooth hammerhead shark);
- Australia by Ms Nicole Phillips (CoP16 Prop. 45 – Freshwater sawfish);
- Ecuador by Mr Fabio Hazin and Dr Gustavo Iturralde (voice over Internet Protocol link) (CoP16 Prop. 46 – Genus *Manta*);
- The European Union (Member Organization) by Dr Elsa Nickel and Ms Sarah Fowler (CoP16 Prop. 44 – Porbeagle shark).

11. Marcelo Vasconcellos presented the preliminary assessment to the Panel.

12. Voice over Internet Protocol interview of Charlie Lim.

OUTCOME OF THE MEETING

1. Evaluation of the proposals

13. The Panel considered the following seven proposals submitted to the CITES sixteenth Conference of the Parties:

CoP16 Proposal 42. Proposal to include *Carcharhinus longimanus* (oceanic whitetip shark) in Appendix II in accordance with Article II paragraph 2(a). The proposal includes an annotation stating that "the entry into effect of the inclusion of *Carcharhinus longimanus* in CITES Appendix II will be delayed by 18 months to enable Parties to resolve the related technical and administrative issues".

CoP16 Proposal 43. Proposal to include *Sphyrna lewini* in Appendix II in accordance with Article II 2(a) and inclusion of *S. mokarran* and *S. zygaena* in Appendix II in accordance with Article II 2(b). The proposal includes an annotation stating that "the entry into effect of the inclusion of these species in CITES Appendix II will be delayed by 18 months to enable Parties to resolve the related technical and administrative issues".

CoP16 Proposal 44. Proposal to include *Lamna nasus* (Bonnaterre, 1788) in Appendix II in accordance with Article II 2(a). The proposal includes an annotation stating that "the entry into effect of the inclusion of *Lamna nasus* in CITES Appendix II will be delayed by 18 months to enable Parties to resolve related technical and administrative issues".

CoP16 Proposal 45. Proposal to transfer *Pristis microdon* from Appendix II to Appendix I of CITES in accordance with Article II, paragraph 1.

CoP16 Proposal 46. Proposal to include the genus *Manta* in Appendix II in accordance with Article II paragraph 2(a).

CoP16 Proposal 47. Proposal to include the ceja river stingray *Paratrygon aiereba* in Appendix II in accordance with Article II paragraph 2(a). The proposal includes an annotation stating that "the entry into effect of the inclusion of *Paratrygon aiereba* in CITES Appendix II will be delayed by 18 months to enable Parties to resolve the related technical and administrative issues".

CoP16 Proposal 48. Proposal to include the freshwater stingrays *Potamotrygon motoro* and *P. schroederi* in Appendix II in accordance with Article II paragraph 2(a). The proposal includes an annotation stating that "the entry into effect of the inclusion of *Potamotrygon motoro* and *Potamotrygon schroederi* in CITES Appendix II will be delayed by 18 months to enable Parties to resolve the related technical and administrative issues".

2. General comments and observations

2.1. Comments from Members and Organizations received by the FAO Secretariat

14. In accordance with the Terms of Reference for the Panel, FAO Members and regional fishery management organizations were notified of the proposals submitted that dealt with commercially exploited aquatic species and were informed that FAO would be convening the Expert Advisory Panel. They were invited to send any comments or relevant information to the FAO Secretariat, for consideration by the Panel. Comments in writing were provided by the Inter-American Tropical Tuna Commission (IATTC), the North East Atlantic Fisheries Commission (NEAFC) and the Western and Central Pacific Fisheries Commission (WCPFC). Information from three other regional fishery bodies was brought to the meeting by staff of these organizations who were invited as experts; i.e. the International Council for the Exploration of the Seas (ICES), the Southeast Asian Fisheries Development Center (SEAFDEC) Policy and from the Sub-Regional Fisheries Commission (SRFC). Finally, a staff member from the International Commission for the Conservation of Atlantic Tunas (ICCAT) attended the first four days of the meeting as Invited Observer and offered relevant information from his organization during this time.

2.2. Interpretation of the Annex 2a Criteria for inclusion of species in Appendix II in accordance with Article II, paragraph 2(a) of the Convention

15. The Panel applied the CITES Res. Conf. 9.24 (Rev. CoP15) criteria interpreted in accordance with FAO's initial advice to CITES on criteria suitable for commercially-exploited aquatic species and as applied in the second and third Meetings of the Expert Advisory Panel in 2007 and 2009. Document CoP14 Inf. 64, prepared by the FAO Secretariat and submitted to the fourteenth Conference of the Parties to CITES in 2007, also provides an explanation of the interpretation of the Annex 2a criteria for inclusion of species in Appendix II as applied by the Panel.

16. The Panel also noted the conclusions of the "Workshop to review the application of CITES criterion Annex 2 a B to commercially-exploited aquatic species" (FAO, 2011), which confirmed the view expressed in FAO (2007) and in CoP14 Inf. 64 that the same definitions, explanations and guidelines in Annex 5 of the Res. Conf. 9.24 (Rev. CoP15), including the decline criteria, apply both for Criterion A and for Criterion B of Annex 2 a.

17. The Panel was informed about the recommendations of the CITES Animals Committee and Steering Committee in 2012 (SC62 Doc. 39, see Appendix D) regarding the application of Annex 2a criterion B and the introductory text to commercially-exploited aquatic species, in particular the following*:* "The Animals Committee finds that there are diverse approaches to the application of Annex 2a criterion B in Resolution Conf. 9.24 (Rev. CoP15). The Animals Committee finds that it is not possible to provide guidance preferring or favouring one approach over another. The Animals Committee recommends that Parties, when applying Annex 2a criterion B when drafting or submitting proposals to amend the CITES Appendices, explain their approach to that criterion, and how the taxon qualifies for the proposed amendment."

2.3. General comments by the Panel on the proposals

18. The Panel welcomed the presentations by representatives of the proponents of the seven proposals at the beginning of its meeting. Both the presentations of the proposals and the opportunity to ask questions of clarification to the representatives of the proponents after initial Panel discussions greatly improved the information available to the Panel and its ability to make informed assessments of the proposals.

19. In relation to the proposals, the Panel noted that the quality of the data and the information varied, some being particularly poor. Some proposals used tables to present indices of productivity and decline, and in some cases information was presented in such a way that it could be relatively easily reviewed and assessed. Nonetheless, the comments from previous Panels are still relevant for

several proposals: presentation of reliable indices, quantitative wherever possible, is central to determining whether species meet criteria for inclusion in the Appendices, and the basis for such indices should be presented clearly and concisely. Even where information is difficult to quantify, all efforts should be made to present the information in a form that can be objectively assessed.

20. Most of the proposals relied to some extent on sources that are unpublished or difficult to access. Assessment of proposals would be facilitated if proponents provided copies of all source documents (in pdf format or other) along with listing proposals. The Panel gratefully acknowledges those proponents who provided copies of source materials during the Panel meeting.

21. Assessing proposals against the listing criteria requires an assessment of the importance of international trade in driving exploitation and in affecting species status. Little information on the relative importance of international trade in driving exploitation was presented in some proposals. This is often due in part to the lack of information on this subject, resulting from the lack of species-level tariff codes for many species in trade (see below).

22. Accurate recording of international trade in sharks is seriously hampered by the absence of any species-specific reporting mechanism. In 2009, the Panel had suggested that CITES Parties and FAO encourage the World Customs Organization (WCO) to establish specific headings within the standardized tariff classification of the Harmonized System to record trade in sharks and their products at the species level. In this context, FAO reported that it has submitted a proposal to the WCO for the inclusion of a large number of shark product codes (see Appendix E).

23. As requested by the Thirty-second Session of COFI in 2012, the Panel has made efforts to improve the comments on the technical aspects of the proposals and their likely effectiveness for conservation, based on the inputs from experts on trade, management and implementation issues. However, the Panel noted that the technical aspects involved in the implementation of CITES listings are context-specific and need to be considered on a case-by-case basis. To improve knowledge on these technical aspects, the Panel recommended the implementation of more empirical studies on the impacts and factors influencing the successful implementation of CITES listings of commercially exploited aquatic species.

2.4. For consideration in reading the reports

24. As was done in the previous Advisory Panels, in considering trends in abundance reported in the proposals, the Panel attempted to evaluate the reliability of each source of information. This was done by assigning a score between zero (no value) and five (highly reliable) to each item of information used to demonstrate population trends. The criteria used to assign a score are included in Table 1.

TABLE 1

Criteria used by the FAO Expert Advisory Panels to assign a measure of the reliability of information derived from different sources for use as indices of abundance

Reliability index of population abundance information	Source of data or information
5	Statistically designed, fishery-independent survey of abundance
4	Consistent and/or standardized catch-per-unit-of-effort (CPUE) data from the fishery
3	Unstandardized CPUE data from the fishery; scientifically designed, structured interviews; well-specified and consistent anecdotal information on major changes from representative samples of stakeholders.
2	Catch or trade data without information on effort
1	Confirmed visual observations; anecdotal impressions
0	Information that does not meet any of the above, or equivalent, criteria; flawed analysis or interpretation of trends

Notes: A score of 0 indicates that the information was not considered reliable, while a score of 5 indicates that it was considered highly reliable. Any information on abundance allocated a non-zero value was considered useful. These scores could be adjusted up or down in any particular case, depending on the length of the time series and the amount of information available on the sources and methods.

Sources: FAO (2004, 2007, 2010).

25. For future evaluations, the Panel recommended that the reliability index in Table 1 also considers the scientific quality of the references used, giving higher reliability to sources that have been subjected to a robust peer review.

REFERENCES

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FAO Expert Advisory Panel assessment report: oceanic whitetip shark - CoP16 Proposal 42 -

Species:

Carcharhinus longimanus – oceanic whitetip shark.

Proposal:

Inclusion of *Carcharhinus longimanus* in Appendix II in accordance with Article II paragraph 2(a) of the Convention and satisfying Criterion A in Annex 2a of Resolution Conf. 9.24 (Rev. CoP15).

Basis for proposal:

The proposal indicates that this low-productivity species has undergone declines of 60–70 percent in the Northwest and Central Atlantic Ocean, and up to a tenfold decline in abundance in the Western and Central Pacific Ocean. The species has been overexploited, primarily because the fins are highly valued in international trade and it is a common bycatch in global pelagic fisheries. According to the proposal, the species is likely to become threatened with extinction unless international trade is monitored and regulated.

ASSESSMENT SUMMARY

CITES biological listing criteria

Both the current FAO Expert Panel and the previous one (FAO, 2010) concluded that, based on the available evidence, oceanic whitetip shark, *Carcharhinus longimanus*, meets the biological criteria for listing in CITES Appendix II. Importantly, new information from the first-ever full-stock assessment conducted (in 2012) for oceanic whitetip for the Western and Central Pacific area corroborated and reinforced this conclusion. There are three time series for the Indian Ocean, all of which decline, with one meeting the Appendix II decline criterion.

There is a paucity of quantitative data with which to determine global trends in this widely distributed tropical oceanic shark. Most of the available indices are based on fishery catch per unit of effort (CPUE). Two regional studies provide long time series (45–50 years) that show historical extents of decline conforming to the Appendix II decline criterion, and a short (10 years) recent time series in one area that also shows a historical extent of decline consistent with the Appendix II decline criterion. Information from other areas is very limited and difficult to interpret.

Comments on technical aspects of the proposal:

Biology and ecology: The Panel agreed with the 2009 Panel’s conclusion that oceanic whitetip is a species with low productivity. There were no other biological or ecological vulnerability or modifying factors that would alter the conclusions regarding biological listing criteria.

Trade: Fins for this species are in demand and of high value in the world market, and there is evidence that international trade is driving retention of bycatch. While this species is generally not targeted but taken as bycatch in fisheries targeting other species, the Panel noted that a large proportion of individuals captured as bycatch could be released alive.

Fisheries management: Retaining bycatch for international trade in high-seas tuna fisheries constitutes an important risk factor for oceanic whitetip, although the risk may have been mitigated to some extent by the introduction of regulations related to sharks. Nine regional fisheries management organizations (RFMOs) and some countries have introduced shark finning regulations, while some countries have banned the retention of shark catch. In principle, these regulations could reduce mortality or at least improve monitoring of shark catches but compliance with these management measures is likely to be variable. More recently, three of the tuna RFMOs have adopted bans on the retention of oceanic whitetips that will have a positive impact on the stock recovery if they are implemented effectively.

Likely effectiveness of a CITES listing for the conservation of the species: The benefits of an Appendix II listing of oceanic whitetip shark would depend on its effective implementation. As most harvest is expected to be from international waters, the CITES requirements for Introduction from the Sea (IFS) and for non-detriment findings (NDFs), if implemented effectively, could contribute to developing better assessments of the species status in the Indian Ocean, where mandatory reporting of oceanic whitetip is not required. It would also provide an additional control to ensure that products entering international trade are derived from legal and sustainable fisheries. Furthermore, a CITES Appendix II listing, if implemented effectively, could also act as a complementary measure for regulations implemented by fisheries management authorities; in particular, where RFMOs have adopted measures prohibiting retention of oceanic whitetip.

DETAILED PANEL ASSESSMENT

1. Scientific assessment in accordance with CITES biological listing criteria

1.1 Biological aspects

The following summary review of the biological aspects of *C. longimanus* is mostly based on the previous FAO Ad Hoc Panel report on the species (FAO, 2010).

1.1.1 Population assessed

Oceanic whitetip shark is a circumglobal, oceanic shark of tropical and subtropical waters, usually found between latitudes 35° N and 30° S and at temperatures warmer than 20° C (Compagno, 1984; Fowler *et al*., 2005). It is normally found offshore in oceanic waters, or near oceanic islands. The species primarily occurs in surface waters at less than 100 m depth, based on unpublished pop-up satellite tag observations off Hawaii and western Gulf of Mexico (Musyl, unpublished, cited in Burgess *et al*., 2005, and Carlson and Gulak, 2012) and on observations of decreasing catch rate between 80 and 280 m (Nakano *et al*., 1997, cited in Bonfil, Clarke and Nakano 2008).

There have been few studies of the population structure of this species. Kohler, Casey and Turner (1998, p. 49) summarize the results of tagging 542 individuals between 1962 and 1993 in the Atlantic Ocean. Six individuals were recaptured, with a maximum distance travelled of 2 270 km (1 226 nm) and a maximum movement of 32 km/day (17.5 nm/day). One oceanic whitetip shark tagged in the western Gulf of Mexico moved a straight-line distance of 238 km southeast to the edge of the continental shelf about 300 km north of the Yucatan Peninsula (Carlson and Gulak, 2012). Studies of population structuring have been identified as a priority in the Pacific because of different CPUE trends between the Eastern and Western Pacific (IATTC, 2007a).

1.1.2 Productivity level

Life history characteristics of oceanic whitetip are mainly associated with low productivity (Table 1). Information on life history characteristics associated with productivity level is available from the Southwest Atlantic (Lessa, Marcante and Paglerani, 1999) and the Western Pacific (Seki *et al*., 1998). This information has been used to derive rate of increase and generation time estimates (Smith, Au and Show, 1998; Cortes, 2002, 2008). The growth rate (as indexed by the von Bertalanffy K parameter), natural mortality and the intrinsic rate of population increase are all consistent with low productivity while the age at maturity and generation time indicate medium productivity (or low to medium). The Panel agreed that the most reliable of these indicate that the overall productivity level is low.

1.1.3 Anthropogenic sources of mortality

Fishing is believed to be the only anthropogenic source of mortality for oceanic whitetip.

Oceanic whitetip shark is a common bycatch species in tuna fisheries in all oceans, being reported in tuna and swordfish longline fisheries in the Atlantic and in tuna longline and purse seine fisheries in the Pacific and Indian Oceans (proposal). However, catches are often unreported, which makes it difficult to quantify accurately the numbers taken annually. For example, the average reported catch to FAO from 2000 to 2010 was only 335 tonnes per year (FAO FishStat), originating from Brazil, China and Portugal. Clarke (2008a) estimated that on average between 2 906 and 7 109 tonnes of oceanic whitetip would have been needed to be taken annually from the Atlantic to supply the international fin trade in the mid-2000s. Lawson (2011) estimated an average annual bycatch of about 129 000 oceanic whitetip sharks (approximately equivalent to 4 730 tonnes/year, using a conversion factor from Clarke *et al*. [2006a]) in tuna fisheries in the Western and Central Pacific between 1995 and 2010. Rice and Harley (2012) conducted the first full stock assessment for oceanic whitetip (in the Western and Central Pacific region) that used catch estimates ranging from 60 000 to 230 000 individuals (about 2 200–8 400 tonnes).

There is evidence that oceanic whitetip is taken in some small-scale, directed fisheries for sharks. Bonfil and Abdallah (2004) reported catch of oceanic whitetip from such fisheries in the Gulf of Aden and the Pacific coast of Central America. Some targeted longline fisheries for sharks have also occurred in Papua New Guinea (Rice and Harley, 2012) and are likely to occur in other areas of the Western Central Pacific. In Maldives, in the Indian Ocean, Anderson, Adam and Saleem (2011) described the existence of a shark longline fishery targeted to reef and pelagic sharks, including the oceanic whitetip. This fishery reached a peak in 1998–2000 and subsequently declined owing to poor catches and low economic returns. The fishery closed in 2010 with a national ban on shark fisheries.

The level of oceanic whitetip catch in these directed fisheries is likely to be minor compared with the bycatch in tuna fisheries.

1.1.4 Population status and trends

Population size

No estimates of global population abundance are available. The only available stock assessment indicates that the median estimate of biomass in the Western Central Pacific in 2010 was 7 295 tonnes (Rice and Harley, 2012), which would be equivalent to population numbers of the order of 200 000 individuals.

Area of distribution

No estimate of the global area of distribution is available, but the species is circumglobal in oceanic waters and so can be considered to have a very large area of distribution.

Population trend

Time series of abundance indices from several parts of the range are available (Table 2).

Northwest Atlantic

The only information on population trends in the Northwest Atlantic is for United States fisheries, and therefore it may or may not be reflective of patterns for the entire area.

Baum and Myers (2004) compared longline CPUE from research surveys in 1954–1957 (“the 1950s”) with those from observed commercial longline sets in 1995–1999 (“the 1990s”) in the Gulf of Mexico (Figure 1). A severe decline in oceanic whitetip CPUE was estimated, equivalent to a 99 percent extent of decline; 3 individuals were taken in 275 sets in the 1990s compared with 397 individuals in 170 sets in the 1950s. This study was subsequently severely criticised by several authors (see below) and was ultimately corrected to an 88 percent extent of decline on the basis of analyses in Driggers *et al*. (2011) (Table 2).

Baum *et al*. (2003) estimated an extent of decline of 70 percent in CPUE based on logbook records in the Northwest Atlantic pelagic longline fishery between 1992 and 2000 (Figure 2), and indicated that declining CPUE trends had been observed in almost all subareas of the fishery area (Figure 3). The exception was a substantial increase in CPUE in Subarea 5, the United States mid-Atlantic (Cape Hatteras to Cape Cod).

The methods and results of Baum *et al*. (2003) and Baum and Myers (2004) were critiqued by Burgess *et al.* (2005), who agreed that the abundance of large pelagic sharks had declined but presented arguments that the population declines were probably less severe than indicated by these studies. Of particular relevance to oceanic whitetip, Burgess *et al*. (2005) noted that the change from steel to monofilament leaders between the 1950s and 1990s could have reduced the catchability of all large sharks, and the increase in the average depth of sets during the same period could have reduced the catchability of the surface-living oceanic whitetip. In responding to the critique, Baum, Kehler and Myers (2005) indicated that their model had in part addressed the change in depth of sets, but agreed that the change in catchability resulting from a change in the material used in leaders needed further study. They noted that subtle changes in the methods of setting gear could have large effects on catch rates, and that for some species of large sharks, catch rates on monofilament were higher than on steel leaders. Nonetheless, Baum, Kehler and Myers (2005) believed that their estimated decline rates were robust.

Subsequently, Driggers *et al*. (2011) conducted a study on the effects of different leader materials on the CPUE of oceanic sharks and determined that with equivalent methods but using a wire leader, the catch rates of Baum and Myers (2004) for the recent period would have been 0.55 rather than 0.02 (as estimated by Baum and Myers [2004] using nylon leaders). Comparing the recent 0.55 value with the Baum *et al*. (2003) value of 4.62 for the 1950s gave an estimated extent of decline of 88 percent.

Cortes, Brown and Beerkircher (2007) found less severe declines, but in a shorter period (1992–2003/05) than those cited above. Declines of 57 percent in logbook CPUE from the commercial longline fishery, and of 9 percent in observer CPUE from the same fleet, were provided. Observer CPUE is generally considered to be more reliable than logbook CPUE.

Western and Central Pacific

Ward and Myers (2005) compared longline CPUE from research surveys in 1951–1958 (“the 1950s”) (880 sets) with those from commercial longline fisheries with observers on board in 1999–2002 (“the 1990s”) (505 sets) (Figure 4). They estimated a tenfold decrease in CPUE over the period. The authors attempted to ensure comparability of the methods used between the two periods and outlined sources of uncertainty in making the comparison. The distribution of sampling in the two periods was different although areas overlapped.

Polacheck (2006) provided evidence that declines in longline CPUE of large pelagic fishes over long periods may overestimate population declines. This was shown to occur for large pelagic species other than sharks, in cases where detailed stock assessments are available to compare with CPUE trends.

Matsunaga and Nakano (1999) provided information on longline CPUE changes between 1967–70 (“the 1960s”) and 1992–95 (“the 1990s”) in four contiguous subareas of the Central Pacific. For the later period, they provided information that had been corrected for differences in depths sampled compared with the earlier period, as well as uncorrected information (Table 3). The uncorrected data showed declines in all four subareas, ranging from 5 percent to 53 percent, while the corrected data show declines in two subareas and increases in two subareas. They noted that further standardization of data sets is required to clarify the extent of change.

Walsh, Bigelow and Sender (2009) compared observer data on commercial longline sets in 1995–2000 and 2001–2006, and showed a 76 percent extent of decline in nominal CPUE in deep sets (median depth of deepest hook 248 m) and a 53 percent decline in shallow sets (median depth of deepest hook 60 m) (deep and shallow sets also differed in gear configuration and bait). More weight should be given to the information from shallow sets given the shallow-living habits of this species. The authors noted that area differences may have affected the estimated trends.

The only full stock assessment of oceanic whitetip in any part of its distribution has recently been conducted for the Western and Central Pacific by Rice and Harley (2012). This assessment was based on standardized CPUE indices from all fisheries covered by observers between 1995 and 2010 and underwent rigorous peer review by expert participants associated with the Western and Central Pacific Fisheries Commission (WCPFC). According to model outputs, the median estimate of the current biomass is approximately 7.3 percent of the total unexploited biomass (Figure 5; 95 percent interval 3.4–19.2 percent). The spawning biomass was estimated to have declined on average by 86 percent since 1995 (95 percent intervals 54.2–91.3 percent). Walsh and Clarke (2011) estimated similar rates of decline in CPUE for the Hawaii-based longline fishery over the same period. The mean nominal CPUE was estimated to have decreased by 91.6 percent, from 0.428 sharks/1 000 hooks in 1995 to 0.036 sharks/1 000 hooks in 2010. The standardized CPUE showed the same general decline trend (Figure 6). The annual mean nominal CPUE in the deep- and shallow-set was estimated to have decreased by 91.5 percent and 89.6 percent, respectively, over the same period.

Clarke *et al*. (2012) analysed the WCPFC long-term record of species-specific catches of sharks collected by onboard observers from 1995 to 2010. Standardized catch rates of longline fleets declined significantly for oceanic whitetip sharks: annual values decreased by 90 percent from 1996 to 2009 and uncertainty in the estimates was low (Figure 7). The authors noted congruent declines to near-zero catch rates in other data sets from Japan and Hawaii over the same period (Clarke *et al*. 2011; Walsh and Clarke 2011) and considered that the significantly smaller sizes of sharks found (Clarke *et al*. 2011, Clarke *et al*. 2012) confirmed the depleted state of the oceanic whitetip population in the Western and Central Pacific Ocean.

Both Walsh and Clarke (2011) and Rice and Harley (2012) concurred that population declines in the region started before 1995. Thus, the estimated declines are highly likely to be underestimates of the historical extent of decline of oceanic whitetip. A comparison of the CPUE from shark surveys conducted in the Central Pacific between 1952 and 1995 (2.7/1 000 hooks; Strasbur [1958], cited in Walsh and Clarke [2011]) with a mean CPUE in 1995–2000 (0.35–1.22/1 000 hooks; Walsh and Clarke [2011]), indicates a twofold to sevenfold decrease in CPUE prior to 1995. However, such estimates must be treated with caution because of possible differences in fishing gear and practices (Driggers *et al*. 2011) and other factors affecting catch rates.

Eastern Pacific

Background information for the design of a shark research programme for the IATTC (IATTC, 2007b) indicates that the purse seine CPUE on floating objects of oceanic whitetip experienced an extent of decline greater than 95 percent in the Eastern Pacific between 1994 and 2006 (Figure 8). This is based on an unstandardized index using observer data from 100 percent of sets during the relatively short period that fish aggregating devices have been used (details in Roman-Verdesoto and Orozco-Zoller [2005]).

Southwest and Equatorial Atlantic

Unstandardized CPUE observations are available from several papers on this species, and these may provide a basis for comparing abundance levels in different periods. Domingo (2004) recorded average catch rates of 0.006 (1998–2003) while Domingo *et al.* (2007) found average catch rates of about 0.025 individuals per hook in 2003–06. In the Equatorial Southwest Atlantic, oceanic whitetips were reported as the second-most abundant shark, outnumbered only by blue shark in research surveys between 1992 and 1997 (Lessa, Marcante and Paglerani, 1999). However, data from observers in the Uruguayan surface longline fleet in the South and Equatorial Atlantic did not confirm this; the highest CPUE recorded did not exceed 0.491 individuals/1 000 hooks for the 2003–06 period with only 63 oceanic whitetips caught on 2 279 169 hooks (Domingo *et al.*, 2007). Castro and Mejuto (1995) recorded a catch rate in this area of 0.26 per 1 000 hooks in the mid-1990s, and Domingo (2004) and Domingo *et al.* (2007) recorded catch rates of 0.09 (2003) and 0.08 (2003–06), respectively. Amandè *et al*. (2010) described bycatch of the European purse seine tuna fishery, and the *Carcharhinus longimanus* were occasionally taken as bycatch.

Hazin *et al.* (2007) noted that the total catch of oceanic whitetip showed a continuous decline over the six-year period from 2000 to 2005, from about 640 tonnes to 80 tonnes. The Spanish longline fleet increased its effort in the South Atlantic in the early 1990 to mid-1990s and expansion of fishing activities by southern coastal countries, such as Brazil and Uruguay, also contributed to increased effort in this period (SCRS, 2009). Fishing effort in this area may have subsequently decreased in recent years (ICCAT, 2012).

Indian Ocean

Recent papers reported in the proposal provide some additional information on the decline of oceanic whitetip in the Indian Ocean. Anderson, Adam and Saleem (2011) estimated that oceanic whitetip contributed 3.5 percent of the shark catch in the longline fishery for sharks off the northern Maldives in the period 2000–04. The fishery was carried out by small dhonis, using shark longlines with an average of 141 hooks. The average CPUE of whitetip in the period was 0.20 individuals per dhoni (or approximately 0.14 sharks/100 hooks). In comparison, data from a shark longline survey conducted in the same area in 1987–88 indicated that oceanic whitetips represented 29 percent of the shark catch (Anderson and Waheed, 1990). The average CPUE was 48.7 sharks/1 000 hooks. Applying the percentage of whitetips in the catch to the total CPUE, it is estimated that the CPUE of whitetip in this period was about 1.41 individuals/100 hooks. This would represent a 90 percent decline in abundance between 1987–88 and 2000–04. Such a level of decline would be consistent with the decrease in the proportion of oceanic whitetip in the catch (from 29 percent to 3.5 percent) and also with anecdotal information reporting a marked decrease in sightings of oceanic whitetip sharks off northern and central Maldives (Anderson, Adam and Saleem, 2011).

Commenting on the study of Anderson, Adam and Saleem (2011), the IOTC Working Party on Ecosystems and Bycatch (WPEB) noted that “data collected on shark abundance represents a consistent time series for the periods 1987–1988 and 2000–2004, collected with similar longline gear, and that the data was showing a declining trend in oceanic whitetip shark abundance, which is a potential indicator of overall stock depletion. The WPEB further noted that it could be related to localised effects, however this was deemed unlikely as oceanic whitetip sharks are wide-ranging and abundance trends from long-term research conducted by the former Soviet Union between the 1960s and 1980s indicate a similar decline of oceanic whitetip sharks, and that sightings of this species in Maldives and Réunion islands is now quite uncommon” (IOTC-WPEB07, 2011).

Yokawa and Semba (2012) analysed the trend in CPUE of oceanic whitetip in Japanese tuna longline fisheries in the Indian Ocean from 2000 to 2010. The data showed low values in 2000 and 2001 (attributed to extremely low catches), and a gradual decreasing trend from 2003 to 2009 (Figure 9). The authors interpreted the decline in CPUE of about 40 percent in the period as an indication of decrease in abundance of the population. Hiraoka and Yokawa (2012) stated that the Japanese longliners, one of major longline fleets in the Indian Ocean, were redeployed to the tropical tuna fishing grounds in this period, whereas up to the mid-1990s they had mainly operated in southern bluefin tuna fishing grounds, which are out of the main distribution area of oceanic whitetip shark. This means that oceanic whitetip shark in the Indian Ocean were subjected to lower fishing effort by Japanese longliners compared with other oceans in the period before the 1990s.

Standardized CPUE was calculated using set records of the Spanish longline fishery targeting swordfish in the period 1998–2011 (Ramos-Cartelle *et al.*, 2012). The historical trend showed large fluctuations and the fit to the data indicated that the CPUE showed a general decreasing trend in 1998–2007 followed by an increase thereafter. Overall, the magnitude of the decrease in CPUE was estimated to be about 25–30 percent over this period.

Other indices

Baum and Myers (2004) observed a 35 percent decline in average weight of individuals taken (from 86.4 kg to 56.1 kg) comparing longline catches in the 1950s with those in the 1990s. Ward and Myers (2005) observed a 50 percent decline in average weight of individuals taken, from approximately 40 kg in the 1950s to approximately 20 kg in the 1990s (Figure 10). They noted that the decline in biomass, considering the concurrent declines in abundance (80 percent) and average weight (50 percent), would have been substantial.

1.2 Assessment relative to quantitative criteria

1.2.1 Small population

As no global population estimate is available, it is not possible to assess oceanic whitetip against this criterion. However, the species is widely distributed and (based on the stock assessment for the Western and Central Pacific and the estimates of numbers of individuals involved in the fin trade, see section 1.1.3), the species is likely to number at least in the hundreds of thousands.

1.2.2 Restricted distribution

As a species occurring circumglobally in tropical and subtropical waters, oceanic whitetip cannot be characterized as a species with a restricted distribution.

1.2.3 Decline

Under the CITES criteria for commercially-exploited aquatic species (Conf. Res. 9.24 Rev. CoP15), a decline to 15–20 percent of the historical baseline for a low productivity species might justify consideration for Appendix I. For listing on Appendix II, being “near” this level might justify consideration, “near” for a low productivity species being 20–30 percent of the historical abundance level (15–20 percent + 5–10 percent). For a medium productivity species, the Appendix I level would be 10–15 percent of the baseline, the Appendix II (“near”) level 15–25 percent. FAO (2001) advised that in examining historical extent of decline, the longest time horizon possible should be examined. Based on the available life history information (Table 1), FAO (2010) considered oceanic whitetip a low productivity species. The 20–30 percent Appendix II decline threshold was therefore adopted for the species. No new information was presented in the current proposal to support a different interpretation. Consequently, the analyses conducted by FAO (2010) remain valid and are reproduced below, with additional new data on decline for the Western and Central Pacific and the Indian Ocean.

No overall population decline index is available for comparison with the guidelines. Stock abundance indices in the individual areas are considered below.

In the Northwest Atlantic (Gulf of Mexico), Baum and Myers (2004) estimated an extent of decline of more than 99 percent in approximately 40 years. Correcting this with recent information on leader materials gives an extent of decline of 88 percent. Recent rates of decline for the Northwest Atlantic are provided by Baum *et al*. (2003) (70 percent 1992–2000), and Cortes, Brown and Beerkircher (2007) (57 percent 1992–2005 for logbook data, 9 percent 1992–2003 for observer data, with more weight to the latter). This historical extent of decline is consistent with an Appendix II listing, if it portrays population abundance accurately. The long time series of Baum and Myers (2004), as corrected by Driggers *et al*. (2011), should be interpreted in light of the evidence of Polacheck (2006) that long-term CPUE series may overestimate population declines of large pelagic fishes.

In the Western and Central Pacific, the most reliable estimate of the extent of decline comes from the only full stock assessment conducted for this species; namely, the assessment undertaken by Rice and Harley (2012) which was peer-reviewed by the WCPFC. They estimated a decline in spawning biomass of 86 percent between 1995 and 2010 (15 years), with declines of unknown magnitude having already occurred prior to the assessment period. As this is the only full stock assessment for this species, it should receive the greatest weight, at least for this region. Underscoring the estimated extent of decline of 86 percent, the fishing mortality in 2010 was estimated to be 6.5 times the optimal (F_{MSY}) level, and the biomass in 2010 was estimated to be 15.3 percent of the optimal (B_{MSY}) level.

The longest time horizon for this region is provided by Ward and Myers (2005), who indicated a historical extent of decline of 90 percent over a period of approximately 40 years. Again, this should be interpreted in the context of the evidence of Polacheck (2006) that long CPUE series may overestimate abundance declines in large pelagic species. Matsunaga and Nakano (1999) indicated consistent declines in four subareas, but not to Appendix II levels, from the late 1960s to the early 1990s (approximately 34 years) using uncorrected data, and a mixed pattern of declines and increases using corrected data. This paper indicated that further standardization would be required in order to interpret the data fully.

A recent rate of decline of 76 percent (deep sets) or 53 percent (shallow sets, more appropriate information for this species) over an approximately ten-year period (1995–2000 vs 2004–06) was provided by Walsh, Bigelow and Sender (2009). Using a slightly longer time series (15 years), Walsh and Clarke (2011) demonstrated higher rates of decline in deep and shallow sets (91.5 percent and 89.6 percent, respectively).

Of the available data for the Western and/or Central Pacific, only those presented by Matsunaga and Nakano (1999) did not show a decline to the Appendix II level. All others were consistent with an Appendix II listing for a low (or even medium) productivity species.

In the Eastern Pacific, the longest time series available is 13 years (1994–2006) (IATTC, 2007b) and indicates a substantial decline of more than 95 percent. The information appears to be robust but is surprising considering the long history of longline exploitation prior to the beginning of this time series, and the relatively low removals by this fishery. This decline would be consistent with an Appendix II decline level.

In the South Atlantic, observations of relative CPUEs suggest a decline in the Southeast Atlantic, and there is conflicting information in the Southwest Atlantic. These unstandardized observations do not appear adequate to support a decision based on the decline criterion.

In the Indian Ocean, a decline in CPUE of 90 percent between 1987–1988 and 2000–04 (approximately 15 years) was inferred on the basis of unstandardized data from different sources (Anderson and Waheed, 1990; Anderson, Adam and Saleem, 2011). Semba and Yokawa (2011) estimated a recent rate of decline of 40 percent over 6 years (2003–09). The two sources of data indicate a continuous decline in the population since the late 1980s. The overall level of decline is uncertain owing to the quality of the data (either too short or with low reliability and possibly confounded by fleet movements during this period). Standardized CPUE was calculated using set records of the Spanish longline fishery targeting swordfish over the period from 1998–2011 (Ramos-Cartelle *et al*., 2012). The historical trend showed large fluctuations; overall, the magnitude of the decrease in CPUE was estimated to be about 25–30 percent over this period.

According to the CITES guidelines (Res. Conf. 9.24 Rev. CoP15) a population could also qualify for an Appendix II listing if the recent rate of decline would drive a population down within approximately a ten-year period from the current population level to the historical extent of decline guideline (i.e. 20 percent of baseline abundance).

Applying the recent rate of decline of 8 percent per year estimated from Semba and Yokawa (2011), it is inferred that population will be at 26 percent of the baseline in 10 years, which is close to the decline criterion. Considering that the population in the Indian Ocean had probably declined to some extent before the period examined by Semba and Yokawa (2011), it can be argued that the recent rate of decline also supports an Appendix II listing for the species.

Were trends due to natural fluctuations?

There is no indication in the sources available that declines were due to natural fluctuations.

2. Comments on technical aspects in relation to trade, management and implementation issues

2.1 Trade aspects

The proposal does not include any new information of trade aspects of the species. The text below is reproduced from FAO (2010), as it contains the most relevant information about trade in the species.

Oceanic whitetip is exploited in many parts of its range, primarily as bycatch in oceanic longline fisheries targeting large pelagic species (tunas, swordfishes and others). In most areas, oceanic whitetip makes up a relatively small proportion of longline catches, and catch rates are relatively low, but total global catch may be substantial. Clarke *et al*. (2006a) (Figure 11) estimated total annual catches of oceanic whitetip, based on trade data from the China, Hong Kong SAR fin market, at 200 000–1 200 000 individuals or 22 000–42 000 tonnes.

Meat and skins may be used, and may be traded on a small scale, but the principal product in trade is fins. Oceanic whitetip meat from longline bycatch has been marketed in Europe, North America and Asia (Rose, 1996; Vannuccini, 1999). Skins may be used for leather products in the United States of America and Mexico (Rose, 1996).

Market preferences for fins of shark species are variable, but oceanic whitetip are a preferred species in many fin markets and make up part of the “first choice” category in the China, Hong Kong SAR fin market (Vannuccini, 1999). Oceanic whitetip fins reportedly command high prices in the China, Hong Kong SAR market (US$45–85/kg, proposal).

Trade statistics for oceanic whitetip fins are not available, as this species (as most other shark species) does not have its own customs code under systems currently in international use (Harmonized Tariff Schedule). Recent work on quantities of fins of different shark species transiting the China, Hong Kong SAR fin market has provided information on the relative importance of oceanic whitetip fins in trade.

The China, Hong Kong SAR market has represented a substantial proportion of the global trade in shark fins: 65–80 percent in 1980–90, 50–65 percent in 1991–1995, 44–59 percent in 1996–2000, 30–50 percent following 2000 (Clarke, 2008). The decline in China, Hong Kong SAR’s share of world trade is attributed to increasing trade through mainland China, where statistics are difficult to obtain (Clarke, Milner-Gulland and Cemare, 2007). Despite the estimated decline over time in the share of the world trade transiting China, Hong Kong SAR, total imports to China, Hong Kong SAR increased in the 1990s (Figure 12), suggesting that total world trade in shark fins was increasing in this period. Shark fins are a traditional luxury or celebration commodity in China, and a recent trend of rising incomes in China is considered a key driver of increasing demand for shark fins (Clarke, Milner-Gulland and Cemare, 2007).

Fins of oceanic whitetip made up 1.8 percent of fins traded in the China, Hong Kong SAR market (Clarke *et al*. 2006b, Table 5) between November 2002 and February 2004.

In summary, it seems clear that oceanic whitetip fins are an important product in the international fin trade, although a relatively small component of the overall trade. This species appears not to be targeted in fisheries for trade, but is taken as bycatch in fisheries targeting other species. Ease of processing and storage of dried fins facilitates trade, and the products command relatively high prices in trade.

2.2 Fisheries management aspects

Oceanic whitetip sharks are mainly caught as bycatch in oceanic fisheries for tunas and swordfish. There are very few examples of directed fisheries and of specific national regulations controlling its use. Because fins have a high value in international markets, there is a high incentive for finning oceanic whitetip shark. Consequently, management measures adopted for controlling shark finning in oceanic fisheries are particularly relevant for the conservation of the species. Regulations related to finning have been adopted by several of the countries that have the highest reported shark catches, including countries of the European Union (Member Organization), the United States of America, Brazil, Mexico, Taiwan Province of China, Argentina, Sri Lanka, New Zealand, Nigeria, Canada and Australia (Camhi *et al*., 2009; Fischer *et al*., 2012). Indonesia, one of the most important shark fishing nations, has not adopted finning regulations in its exclusive economic zone (EEZ), but would be subjected to the finning regulations in the jurisdictional areas of the RFMOs in which that country participates. In the United States of America, area closures and gear restrictions, implemented under the Fishery Ecosystem Plan for Pacific Pelagic Fisheries of the Western Pacific Region, are used to minimize the bycatch of sharks in the United States Pacific islands (WPFMC 2008). Other countries have banned fisheries for sharks in their EEZs, or defined fisheries exclusion areas for sharks, including Colombia, French Polynesia, Palau, Maldives, Honduras, Bahamas, Tokelau, the Marshall Islands, Costa Rica, Ecuador, Mauritania and Guinea-Bissau (proposal; Techera, 2012).

Measures have also been adopted by most of the RFMOs managing fisheries that also catch pelagic sharks: ICCAT, General Fisheries Commission for the Mediterranean (GFCM), Indian Ocean Tuna Commission (IOTC), IATTC, Northwest Atlantic Fisheries Organization (NAFO), Southeast Atlantic Fisheries Organization (SEAFO), WCPFC, Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR) and NEAFC (Fischer *et al*., 2012). Three tuna RFMOs have adopted specific management measures prohibiting the retention of oceanic whitetip sharks. ICCAT has prohibited the retention, transshipping, landing, storing and selling of oceanic whitetip sharks in fisheries managed by it (Recommendation 08 of 2010). The IATTC has recently adopted a similar measure for the fisheries that it manages (Resolution C-11-10; in effect from 1 January 2012). Even more recently, the WCPFC has adopted a non-retention measure for oceanic whitetip (Conservation and Management Measure 2011-04; in effect from 1 January 2013).

Measures prohibiting retention of oceanic whitetip, if adequately enforced, could reduce the bycatch mortality of oceanic whitetip because the species has a low post-capture mortality (Musyl *et al*., 2011); therefore, a large proportion of individuals caught and released alive would survive. For example, the finning regulations introduced in 2001 in the United States Hawaii-based longline fishery have acted to reduce mortality on oceanic whitetip and other large shark species (Walsh, Bigelow and Sender, 2009). In 1995–2000, prior to the ban, the fins were taken from a large proportion of captured oceanic whitetip with the remaining carcass being discarded (72.3 percent in deep sets and 52.7 percent from shallow sets), as was the case with other large sharks (Walsh, Bigelow and Sender, 2009, Table 3). In 2004–06, following the implementation of the new regulations, almost all sharks were released, although some were dead on release. Minimum mortality estimates declined substantially as a result of the finning regulations, from 81.9 percent to 25.6 percent in deep sets and from 61.3 percent to 9.1 percent in shallow sets (Walsh, Bigelow and Sender, 2009, Table 3).

Aside from this example, there is little information on the level of compliance with the various fisheries management measures for sharks, including oceanic whitetip. Compliance is likely to be variable among countries and regions.

2.3 Implementation issues

2.3.1 Introduction from the sea

Given that oceanic whitetip is a species of the open ocean, rather than of continental shelves, and therefore primarily occurs in the marine environment not under the jurisdiction of any State, introduction from the sea (i.e. transport of captured specimens from international waters to areas under national jurisdiction) would be expected to occur often, at least in fisheries regulated by RFMOs that allow the landing of the species. As noted above, ICCAT, IATTC and WCPFC have recently prohibited the capture and landing of oceanic whitetip. However, this is not the case for the Indian Ocean, where the available evidence indicates considerable risk to the stock status at current effort levels (IOTC, 2011).

Under CITES, such transport of specimens listed on Appendix II would require a certificate from the State to whose jurisdiction the specimens are brought, including an NDF and a legal acquisition finding. Exactly how these certification processes would be carried out is still a matter of debate within CITES. In matters related to listed shark species, it is imperative that RFMOs and the CITES Secretariat work closely together. The same applies to the CITES Secretariat and national scientific and management authorities.

2.3.2 Basis for findings: legally obtained, non-detrimental

Export permits for Appendix II species must be accompanied by a certificate attesting that the specimens were legally obtained. There appear to be few specific regulations on harvest of oceanic whitetip, including recent measures adopted by ICCAT, IATTC and WCPFC prohibiting the retention, transshipping, landing, storing and selling of oceanic whitetip sharks in fisheries managed by these three commissions. Other than that, there is the blanket ban on finning of harvested sharks in a number of countries and RFMOs and the requirement under the FAO Compliance Agreement[1] and the UN Fish Stocks Agreement[2] for States to require vessels entitled to fly their flags to have an authorization to fish in areas beyond national jurisdiction. To this end, a small number of States have made it a requirement in national legislation for vessels entitled to fly their flags to have an authorization to fish on the high seas or in areas beyond national jurisdiction. Other than the potential of some control in these few States, there would appear to be little impediment to jurisdictions certifying that specimens were legally obtained, should an Appendix II listing come into effect.

Export permits for products from Appendix II species must also be accompanied by NDFs showing that exports are not detrimental to survival of the species, that is, that they are consistent with sustainable harvesting. Development of an NDF requires appropriate scientific capacity, biological information on the species, and a framework for demonstrating that exports are based on sustainable harvests. The quality of NDFs is reviewed by the Scientific Committees of CITES (Animals and Plants Committees) and within individual parties, but perhaps with variable degrees of robustness and/or validity. Currently, the Western Central Pacific is the only region that has developed a stock assessment model for the species (Rice and Harley, 2012), which can be used to assess population status and sustainable harvests in the region. Other less-data-intensive methods would have to be applied in other parts of the species range. FAO (2004, paragraphs 28–29) provides some guidance on NDFs in a fisheries context. Resources and tools are available to inform other CITES Parties on the necessary information and steps to be taken in the making of NDFs (Rosser and Haywood, 2002; Anonymous, 2008).

2.3.3 Identification of products in trade

The proposal indicates that fins from oceanic whitetip are one of the most distinctive products in the Asian shark fin trade, possessing characteristic morphological and colour characters that facilitate identification. Traders in the China, Hong Kong SAR fin market classify oceanic whitetip fins to a single product category (“Liu Qui”) with a high degree of accuracy (100 percent on a sample of 23 fins) (Clarke *et al*. 2006b). In addition, a rapid and increasingly inexpensive identification method using a DNA technique has been developed by the European Union (Member Organization) recently.

Shark species codes

Accurate recording of international trade in sharks is seriously hampered by the absence of any species-specific reporting mechanism. To address this, FAO (2010) suggested that the Conference of the Parties encourage the WCO to establish specific headings within the standardized tariff classification of the Harmonized System to record trade in sharks and their products at the species level, with particular urgency for the major shark species in trade.

[1] The Agreement to Promote Compliance with International Conservation and Management Measures by Fishing Vessels on the High Seas.
[2] Agreement for the Implementation of the United Nations Convention on the Law of the Sea of 10 December 1982 relating to the Conservation and Management of Straddling Fish Stocks and Highly Migratory Fish Stocks.

2.3.4 "Look-alike" issues

CITES allows for Appendix II listing of species whose parts and derivatives resemble those of other Appendix I or II species to the extent that enforcement officers who encounter such products are unlikely to be able to distinguish between them (Article II paragraph 2 (b)).

From the information available, fins of oceanic whitetip are relatively distinctive, and could possibly be distinguished from those of other species by enforcement officers using identification manuals.

2.4 Likely effectiveness of a CITES listing for the conservation of the species

The Panel noted that the benefits of a CITES Appendix II listing for oceanic whitetip will depend on whether it is implemented effectively. If this is the case, there could be significant benefits for monitoring and assessing the status of oceanic whitetip populations. A CITES Appendix II listing would require that exports are accompanied by certificates attesting that the species was legally obtained and that harvest was non-detrimental to the species. If the institutional arrangements and capacity for undertaking NDFs are addressed, particularly for specimens introduced from the sea, the listing would provide an additional control that products entering international trade are derived from legal and sustainable fisheries.

As most harvest is expected to be from international waters, the catch documents required under the IFS provisions would provide species-level information on catches brought from international waters to national jurisdiction. In addition, the requirement for NDFs to accompany such transfer of specimens or products could contribute to developing better assessments of species status. A CITES Appendix II listing could also act as a complementary measure for regulations implemented by fisheries management authorities. For example, the CITES listing could help ensure compliance with the ban on retention recently implemented by three of the tuna RFMOs.

Catches of oceanic whitetip in target fisheries are likely to be minor compared with the bycatch in fisheries targeting other species, particularly tuna. Although this latter aspect limits the benefits of the Appendix II listing, because a proportion of the sharks taken as bycatch may already be dead, the experience with the application of a finning ban in the United States Hawaii longline fleet shows that bycatch mortality can be reduced when sharks taken incidentally are released.

Requirements for additional information will create a burden that may need to be addressed through capacity building, particularly in developing countries. However, this is not unique to a potential CITES listing for oceanic whitetip; it applies in general to all new management measures or regulations.

3. Conclusion

Both the current FAO Expert Panel and the previous one (FAO, 2010) concluded that, based on the available evidence, oceanic whitetip shark, *Carcharhinus longimanus*, meets the biological criteria for listing in CITES Appendix II. Importantly, new information from the first-ever full stock assessment conducted (in 2012) for oceanic whitetip for the Western and Central Pacific area corroborated and reinforced this conclusion. This assessment, which was peer reviewed by the WCPFC, estimated that oceanic whitetip declined by approximately 86 percent over the period 1995–2010, and that declines of unknown magnitude had already occurred prior to the assessment period. There are three time series for the Indian Ocean, all of which decline, with one meeting the Appendix II decline criterion.

The relevant conclusions from FAO (2010) and the new information from the Western and Central Pacific Ocean and the Indian Ocean are summarized here. There is a paucity of quantitative data with which to determine global trends in this widely-distributed tropical oceanic shark. Most of the available indices are based on fishery CPUE. Two regional studies provide long time series (45–50 years) that show historical extents of decline conforming to the Appendix II decline criterion, and a short (10 years) recent time series in one area also shows a historical extent of decline consistent with the Appendix II decline criterion. Information from other areas is very limited and difficult to interpret.

Indices are available for a number of regions although they are of variable reliability (see Table 2). In the Northwest Atlantic, the longest time series (from the 1950s to the 1990s) shows a substantial extent of decline consistent with the Appendix II decline criterion. Indices from the Northwest Atlantic covering more recent periods (1992–2005) showed continuing declines. In the Central Pacific, the longest time series (from the 1950s to 1999–2002) shows a substantial extent of decline consistent with the Appendix II decline criterion. A set of time series (from the 1960s to the early 1990s) shows declines in four subareas of the Central Pacific, but not to levels consistent with the Appendix II decline criterion, when information uncorrected for depths of sets is considered. When corrected data are considered, trends are conflicting. More recent series (1995–2005) show a continuing large decline. The most compelling evidence for decline in the Southern Hemisphere is the recent full stock assessment conducted for the Western and Central Pacific, which estimated a decline in spawning biomass of 86 percent between 1995 and 2010 (15 years). In the Eastern Pacific, the only available index shows a very large extent of decline, consistent with the Appendix II decline criterion, over a short time period (1994–2006), but the reliability of this index may be low.

Both panels agreed that oceanic whitetip is a species with low productivity. There were no other biological or ecological vulnerability or modifying factors that would alter the conclusions regarding biological listing criteria.

Fins for this species are in demand and of high value in the world market, and there is evidence that international trade is driving retention of bycatch. This species is generally not targeted, but rather is taken as bycatch in fisheries targeting other species. The Panel noted that a large proportion of individuals captured as bycatch could be released alive.

Retaining bycatch for international trade in high-seas tuna fisheries constitute important risk factors for oceanic whitetip, although the risk may have been mitigated to some extent by the introduction of regulations related to sharks. Nine RFMOs and some countries have introduced shark finning regulations, while some countries have banned the retention of shark catch. In principle, these regulations could reduce mortality or at least improve monitoring of shark catches but compliance with these management measures is likely to be variable. More recently, three of the tuna RFMOs have adopted bans on the retention of oceanic whitetips that would have a positive impact on the stock recovery if they are implemented effectively. In principle, these regulations could reduce mortality or at least improve monitoring of shark catches but compliance with these management measures is likely to be variable. In addition, the effectiveness of finning regulations has recently been debated because of the possibility that the focus on finning detracts from more explicit and directed efforts to ensure that catches are sustainable. The more direct measures of prohibiting retention adopted by ICCAT, IATTC and WCPFC in the Atlantic and Pacific oceans should have a positive impact on the stock recovery if they are implemented effectively.

The benefits of an Appendix II listing of oceanic whitetip shark would depend on its effective implementation. As most harvest is expected to be from international waters, the CITES requirements for IFS and for NDFs, if implemented effectively, could contribute to developing better assessments of the species status in the Indian Ocean, where mandatory reporting of oceanic whitetip is not required. It would also provide an additional control to ensure that products entering international trade are derived from legal and sustainable fisheries. Furthermore, a CITES Appendix II listing, if implemented effectively, could also act as a complementary measure for regulations implemented by fisheries management authorities; in particular, where RFMOs have adopted measures prohibiting retention of oceanic whitetip.

Requirements for additional information will create a burden that may need to be addressed through capacity building, particularly in developing countries. However, this is not unique to a potential CITES listing for oceanic whitetip; it applies in general to all new management measures or regulations.

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TABLES AND FIGURES

TABLE 1

Information for assessing the productivity of oceanic whitetip shark

Parameter	Information	Productivity	Source
Intrinsic rate of increase	General (r_{2M}): 0.067	Low	Smith *et al*. (1998)
	General: 0.067 (from λ = 1.069)	Low	Cortes (2008)
	Western/Central Pacific: 0.11 (from λ = 1.117)	Low	Cortes (2002)
Age at maturity	Southwest Atlantic:		
	Males: 6–7 years Females: 7–8 years	Medium	Lessa, Marcante and Paglerani (1999)
	West Pacific: 4–5 years (both sexes)	Medium	Seki *et al*. (1998)
	Southwest Indian Ocean 6–7 years (both sexes)	Medium	Bass *et al*. (1973)
Natural mortality	Western Central Pacific: 0.18 (0.12–0.32)	Low	Rice and Harley (2012)
von Bertalanffy K	Southwest Atlantic: 0.099 based on observed lengths	Low	Lessa, Marcante and Paglerani (1999)
	Northwest Atlantic: 0.04–0.09	Low	Branstettar (1990)
	West Pacific: 0.103	Low	Seki *et al*. (1998)
Generation time	General: 10 years	Low/Medium	Cortes *et al*. (2008) cited in proposal
	General: 11.1 years	Low	Cortes (2008)
	Atlantic: 10.4 years	Low	Cortes *et al*. (in press)
	Western/Central Pacific: 7 years	Medium	Cortes (2002)

Source: FAO (2010).

TABLE 2

Decline indices for oceanic whitetip

Criterion	Index	Trend	Basis	Coverage	Reliability	Source
Northwest Atlantic	CPUE longline	EOD 99%	Calculated by authors, 1950s to 1990s	Gulf of Mexico	Research surveys (1950s), observers (1990s) (2)	Baum and Myers (2004)
	CPUE longline	EOD 88%	Calculated by authors, 1950s to 1990s	Gulf of Mexico	Research surveys (1950s), observers (1990s) (4–5)	Baum and Myers (2004), corrected on the basis of Driggers *et al.* (2011).
	CPUE, longline	EOD 70%	Calculated by authors, 1992–2000	Northwest Atlantic	Commercial logbook data (3)	Baum *et al*. (2003)
	CPUE longline	EOD 57%	1986–2005 CPUE logbooks	Northwest Atlantic	Commercial logbook data (3)	Cortes *et al*. (2007)
	CPUE longline	EOD 9%	1992–2005 CPUE observed sets	Northwest Atlantic	Observer programme data (4)	Cortes *et al*. (2007)
Western Central Pacific	Spawning biomass estimated from stock assessment	EOD 86%	Stock synthesis model, multiple sources of data, 1995–2010	Western Central Pacific	Assessment based on multiple sources of data (5)	Rice and Harley (2012)
	CPUE longline	EOD 90%	Calculated by the authors, 1996–2009	Western Central Pacific	Western Central Pacific	Clarke *et al*. (2012)
	CPUE longline	EOD 91.5% in deep sets, 89.6% in shallow sets	Calculated by the authors, 1995–2010	Central Pacific	Standardized CPUE from observer data (4)	Walsh and Clarke (2011)
	CPUE longline	EOD 90%	Calculated by authors, 1950s–1990s	Central Pacific Ocean	Research surveys (1950s), observers (1990s) (4–5)	Ward and Myers (2005)
	CPUE longline	EOD 76% in deep sets, 53% in shallow sets	Calculated by authors, 1995–2000 vs 2004–2006	Central Pacific Ocean	Observer data from commercial fleet (4). Shallow sets data with higher weight.	Walsh *et al*. (2009)
	CPUE longline	EOD 53%, 5%, 27%, 52% in 4 subareas	Late 1960s to mid-1990s	Central Pacific, uncorrected for depth changes	Unstandardized CPUE (3)	Matsunaga and Nakano (1999); see Table 3 of present report.
	CPUE longline	EOD 32%, 31% in 2 subareas; increases of 38%, 4% in 2 subareas	Late 1960s to mid-1990s	Central Pacific, corrected for depth changes	Unstandardized CPUE (3)	Matsunaga and Nakano (1999); see Table 3 of present report.
Eastern Pacific	CPUE, observed purse seine sets on floating objects	EOD 95%	Inspection of figure, 1994–2006	Eastern Pacific Ocean	Standardized observer data (4)	IATTC (2007a, 2007b)
Indian Ocean	CPUE longline	EOD 90%	Comparing CPUE data from survey (1987–1988) and commercial fishery (2000– 2004)	Northern Maldives	Comparison of unstandardized CPUEs from different sources (3)	Anderson, Adam and Saleem (2011); Andreson and Waheed (1990)
	CPUE longline	RRD 40% (annual rate of decline of 8%)	Inspection of figure, 2003– 2009	Indian Ocean	Standardized logbook data (4)	Semba and Yokawa (2011)
	CPUE longline	Decline 25–30% 1998–2011	Spanish commercial longline fleet	Indian Ocean	Standardized CPUE (4)	Ramos-Cartelle *et al*. (2012)

Notes: EOD: historical extent of decline; RRD: recent rate of decline.

Sources: Revised from FAO (2010). Reliability values are based on FAO (2001).

TABLE 3

Catch rate observations and decline calculations in Central Pacific

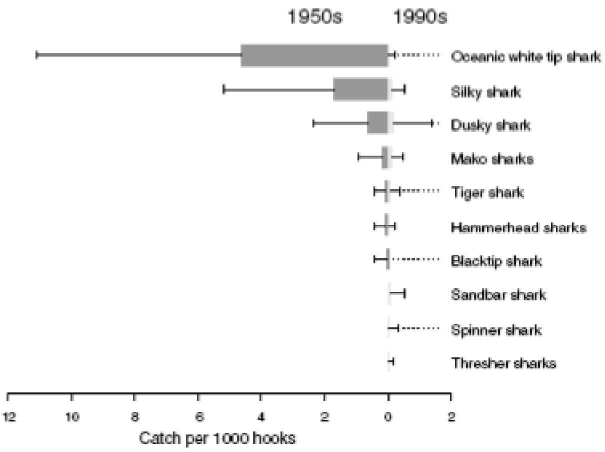
Years	Uncorrected				Corrected			
	0–10E	0–10W	10–20E	10–20W	0–10E	0–10W	10–20E	10–20W
1960s	1.6	1.73	0.51	0.77	1.6	1.73	0.51	0.77
1990s	0.76	1.65	0.37	0.37	1.09	2.38	0.53	0.53
Decline (%	53	5	27	52	32	–38	–4	31

Notes: 0–10E, 0–10W, etc. are different subareas of the Central Pacific. "Uncorrected" are 1990 observations uncorrected for depth changes between periods; "corrected" are 1990s observations corrected for depth differences. In "Decline" row, positive numbers are declines, negative numbers are increases.

Source: Matsunaga and Nakano (1999).

FIGURE 1

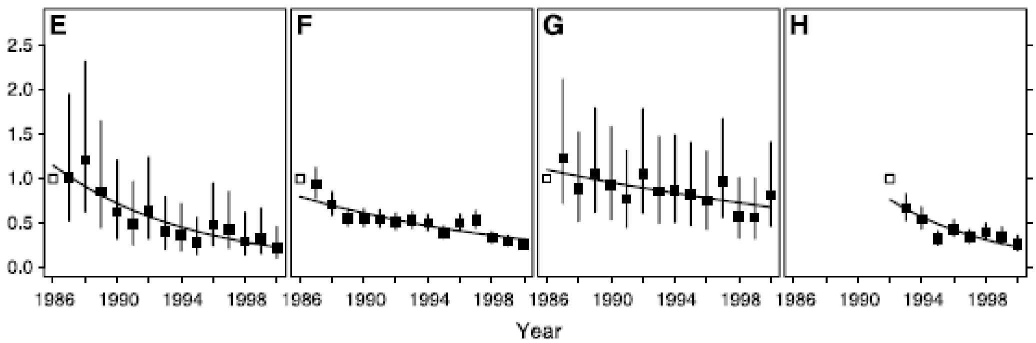
Mean catch rates (+/– SD) in 1950s (longline research survey) and 1990s (commercial observer from longline fleet) from Gulf of Mexico



Source: Baum and Myers (2004).

FIGURE 2

Relative abundance index (CPUE) of oceanic sharks in the NW Atlantic from logbook records in the pelagic longline fishery

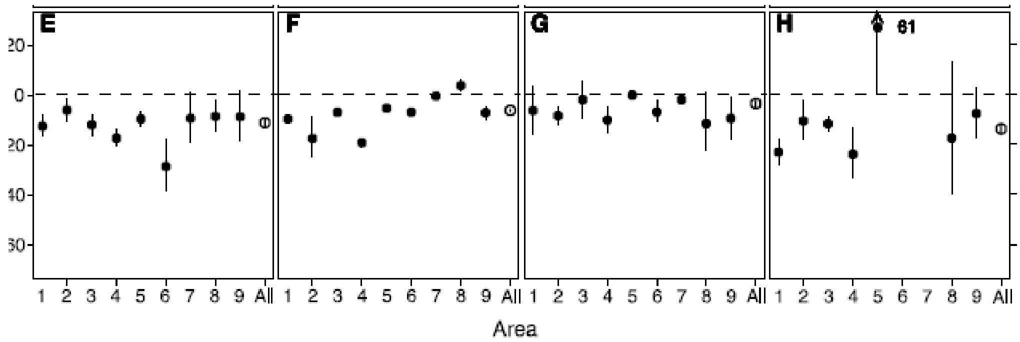


Note: H = oceanic whitetip.

Source: Baum *et al*. (2003).

FIGURE 3

Rate of change in abundance over time in subareas of the NW Atlantic

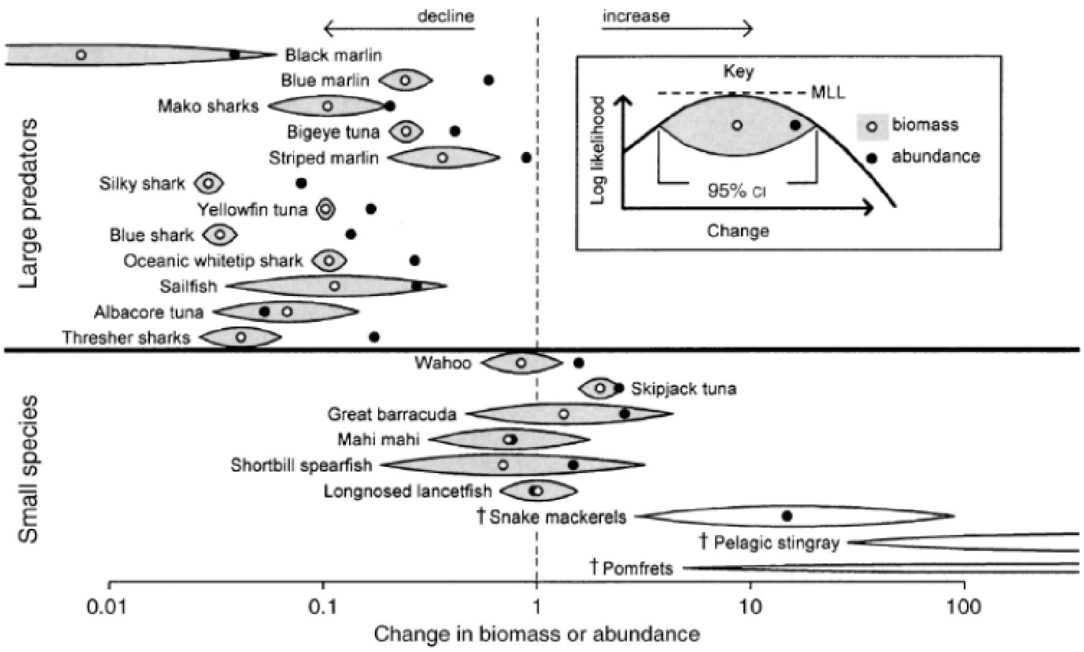


Note: H = Oceanic whitetip.

Source: Baum *et al.* (2003).

FIGURE 4

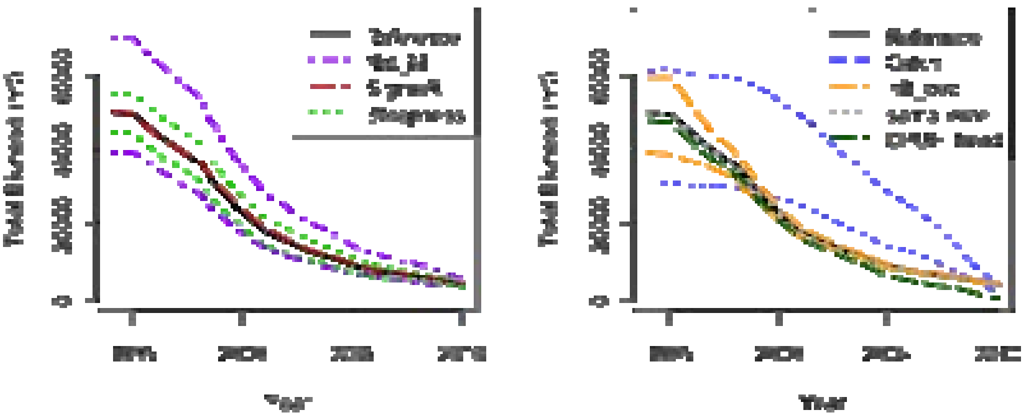
Change in biomass and abundance between 1950s and 1990s, Central Pacific Ocean



Source: Ward and Myers (2005).

FIGURE 5

Estimated biomass of oceanic whitetip sharks in the Western Central Pacific, 1995–2010

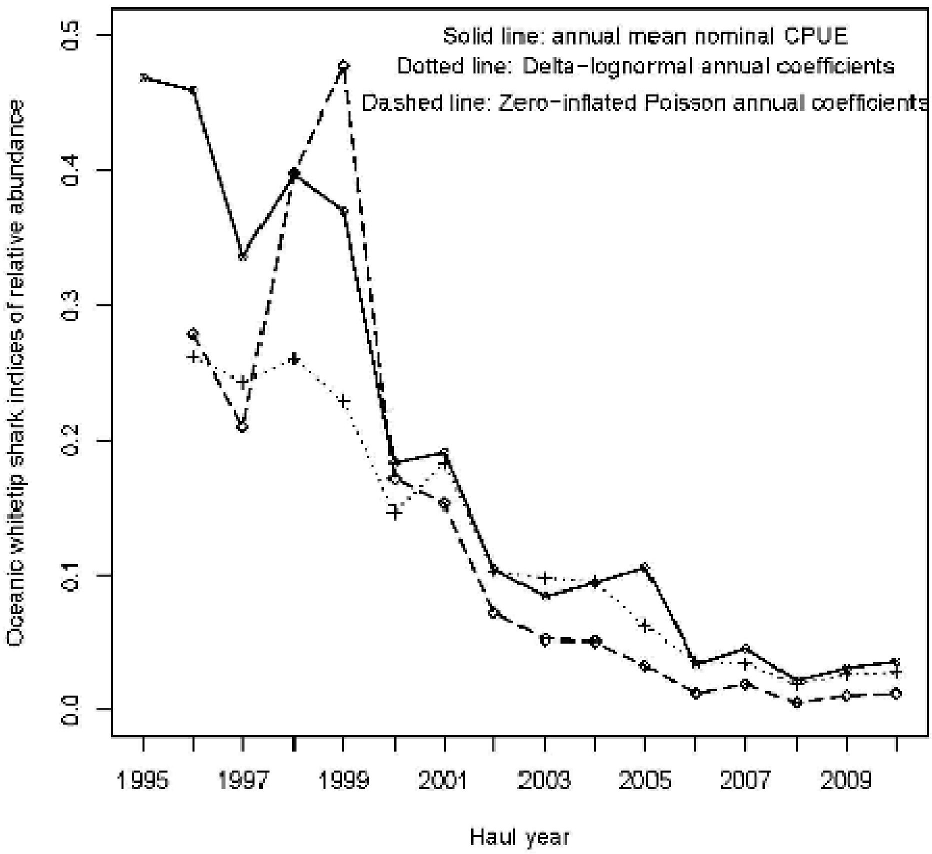


Notes: The graphs show the results of sensitivity analysis effects on total biomass of alternate variable levels on the reference case. The figure on the left shows the effects of the natural mortality, SigmaR (the s.d. on the recruitment devs.), and the steepness of stock-recruitment relationship. The figure on the right shows the effects of changing the catch inputs, initial depletion, sample size down weighting, and the CPUE inputs.

Source: Rice and Harley (2012).

FIGURE 6

Annual indices of relative abundance from the delta lognormal and zero-inflated Poisson analyses of oceanic whitetip shark CPUE in the Hawaii-based pelagic longline fishery in 1995–2010

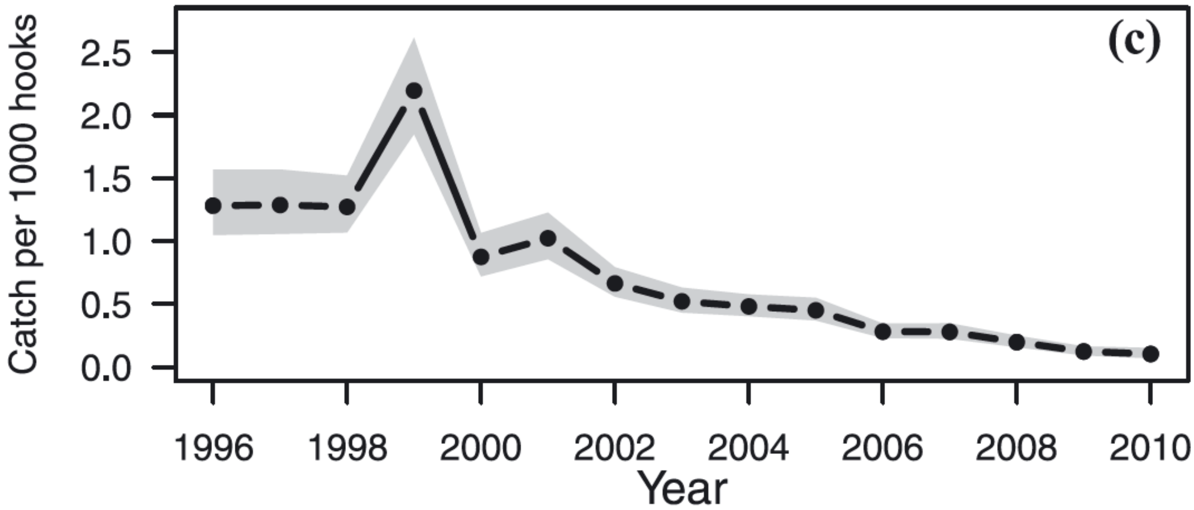


Note: The nominal CPUE trend is included for comparison.

Source: Walsh and Clarke (2011).

FIGURE 7

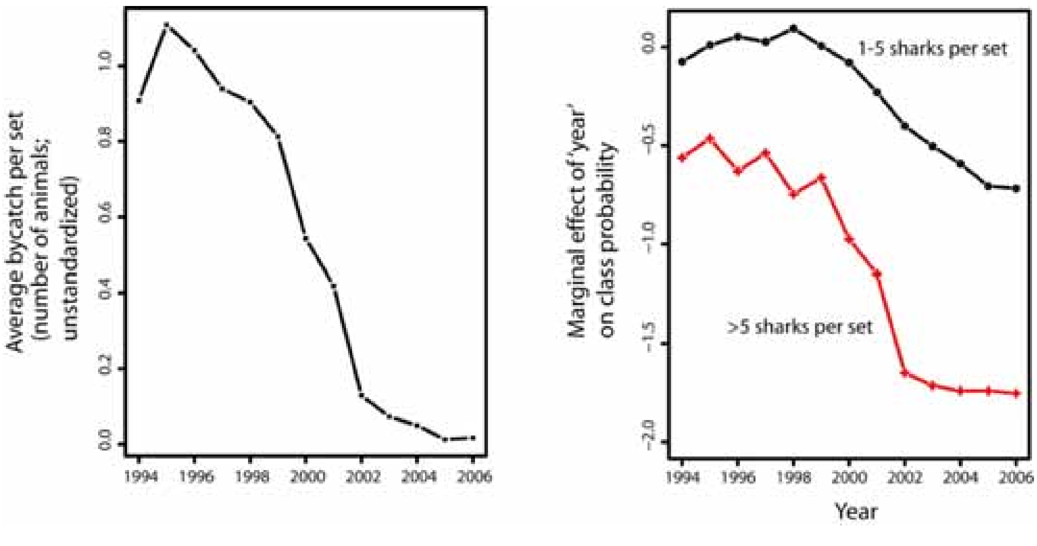
Standardized longline catch rates for oceanic whitetip in the Western and Central Pacific Ocean, 1996–2009



Source: Clarke *et al*. (2012).

FIGURE 8

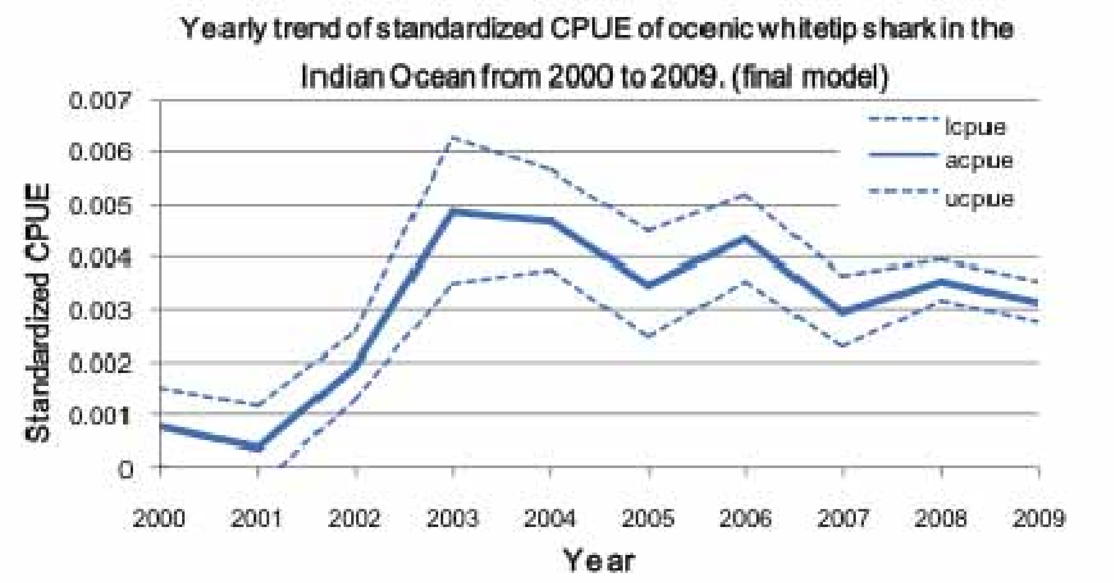
CPUE of oceanic whitetip sharks, purse seine research surveys, Eastern Pacific Ocean (left panel)



Source: IATTC (2008).

FIGURE 9

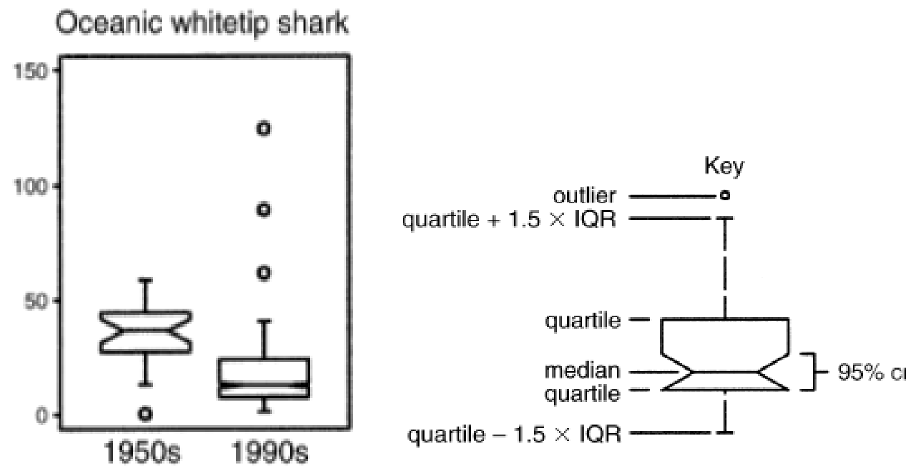
Trends of standardized CPUE of oceanic whitetip shark in Japanese tuna longline fisheries in the Indian Ocean, with 95 percent confidence interval, 2000–2009



Source: Semba and Yokawa (2011).

FIGURE 10

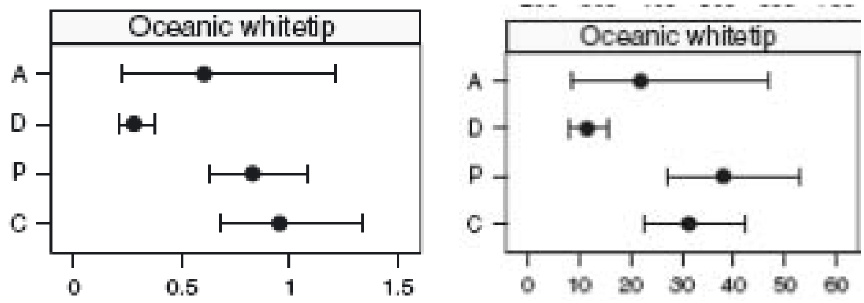
Change in mean body mass (kilograms), longline-caught individuals, Central Pacific Ocean



Source: Ward and Myers (2005).

FIGURE 11

Estimated annual catches of oceanic whitetip based on trade data from China, Hong Kong SAR fin market

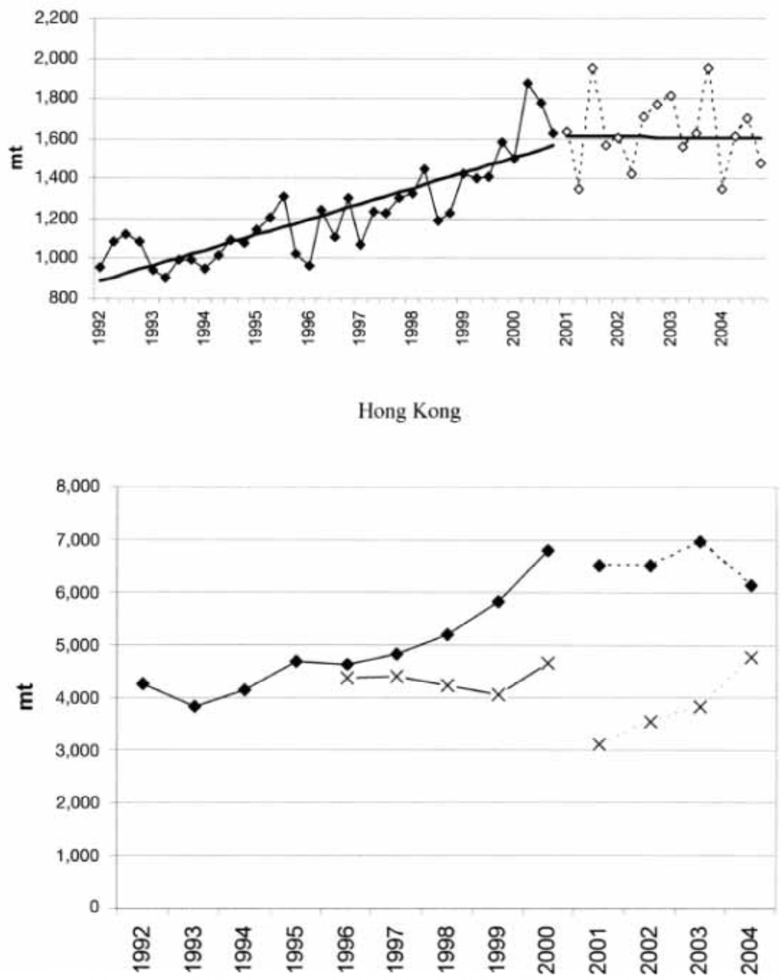


Notes: Left panel – thousands of individuals. Right panel – tonnes. Estimates based on dorsal fins (D), pectoral fins (P), caudal fins (C) and a mixture distribution (A).

Source: Clarke *et al*. (2006a).

FIGURE 12

Imports of shark fins to China, Hong Kong SAR and mainland China



Notes: Upper figure: quarterly imports to China, Hong Kong SAR (a change in statistical reporting means values before and after 2001 are not strictly comparable). Lower figure: annual imports to China, Hong Kong SAR (solid symbols) and mainland China (×).

Source: Clarke *et al.* (2007).

FAO Expert Advisory Panel assessment report: scalloped hammerhead shark, great hammerhead shark and smooth hammerhead shark
- CoP16 Proposal 43 -

Species:

Sphyrna lewini (scalloped hammerhead shark), *Sphyrna mokarran* (great hammerhead shark) and *Sphyrna zygaena* (smooth hammerhead shark).

Proposal:

Inclusion of *Sphyrna lewini* in Appendix II in accordance with Article II 2(a) and inclusion of *S. mokarran* and *S. zygaena* in Appendix II in accordance with Article II 2(b).

Basis for proposal:

Sphyrna lewini: The proposal indicates that this species qualifies for inclusion in Appendix II because it is overexploited for its fins, which are highly valued in trade, and has experienced historic declines of at least 15–20 percent from the baseline in multiple ocean basins. Based on the rate of exploitation, the species is likely to become threatened with extinction unless trade regulations provide the incentives to improve monitoring and management.

Sphyrna mokarran and *Sphyrna zygaena*: The specimens of the species in the form in which they are traded resemble specimens of a species included in Appendix II under the provisions of Article II, paragraph 2(a), or in Appendix I, such that enforcement officers who encounter specimens of CITES-listed species, are unlikely to be able to distinguish between them. The proposal indicates that hammerhead fins are morphologically similar, and that traders often lump fins from these species together.

ASSESSMENT SUMMARY

CITES biological listing criteria

The Panel concluded that based on the available evidence scalloped hammerhead (*Sphyrna lewini*) meets the biological criteria for listing on CITES Appendix II. The other two proposed species, great hammerhead shark (*S. mokarran*) and smooth hammerhead shark (*S. zygaena*) fulfil the criteria for inclusion under CITES Appendix II stipulated in Article II, paragraph 2b (“look-alike clause”).

When evaluated on a population-by-population basis, the historically large population in the Northwest Atlantic was considered to meet the Appendix I decline criterion; there is a declining trend in the Southwest Atlantic population considered by the Panel to meet Appendix II listing criteria. In the Eastern Central Atlantic, the historical trends did not show significant declines but the recent rate of decline would meet the Appendix I criterion. The Indian Ocean and Eastern Pacific populations have declined, and in the Western Pacific the trends are inconsistent.

Comments on technical aspects of the proposal

Biology and ecology: Scalloped hammerhead is a circumglobal coastal species of warm temperate and tropical seas. It can be characterized as a species of low productivity.

Trade: Scalloped hammerhead fins are traded internationally and command a high price, while the meat is mainly consumed locally but a small portion of the meat is also traded internationally.

Fisheries management: Hammerhead sharks are a target and/or bycatch species in diverse industrial and artisanal fisheries around the globe. General shark management measures for sharks (such as finning regulations and closed areas) exist but species-specific fisheries management is rare and illegal, unreported and unregulated (IUU) fishing has been identified as a problem.

Likely effectiveness of a CITES listing for the conservation of the species: Except for the Northwest Atlantic, species-specific assessments that could provide a basis for NDFs are lacking. The Panel felt that a CITES listing, if implemented effectively, would improve the catch data for stocks going into international trade. In principle, a CITES Appendix II listing will be more effective for fisheries targeting sharks for their fins that enter international trade. However, a CITES Appendix II listing will have limited effect if the shark catches are consumed and traded locally.

DETAILED PANEL ASSESSMENT

1. Scientific assessment in accordance with CITES biological listing criteria

1.1 Biological aspects

The following summary review of the biological aspects of *S. lewini* is mostly based on the previous FAO Panel report on the species (FAO, 2010). The report was updated with any new information presented in the proposal and information made available by Panel participants.

1.1.1 Population assessed

Scalloped hammerhead is a circumglobal shark species found in coastal warm temperate and tropical seas (Compagno, 1984; Fowler *et al*., 2005). Like other hammerhead sharks (the family Sphyrnidae comprises nine species), this species is primarily found on continental shelves and in deep water adjacent to them, to depths of at least 275 m, but is rarely found in open ocean areas. The species seems to be endemic to certain coastal regions, especially the females, which tend to display site fidelity to nursery areas and rarely disperse to oceanic areas (Daly-Engel *et al*., 2012). Males are found to disperse long distances, thus helping to facilitate gene flows across oceanic basins (Daly-Engel *et al*., 2012).

1.1.2 Productivity level

Most values of life history parameters are consistent with a low productivity level (Table 1). The biology of scalloped hammerhead shark is characterized by vivipary, with fecundity varying from 12 to 41 pups per female per year (proposal). The gestation period varies from 8 to 12 months and is followed by a one-year resting period. Information is available from the Atlantic (Piercy *et al*., 2007; Cortes *et al*., 2009), Western Indian Ocean (Dudley and Simpfendorfer, 2006), Western Pacific (Chen *et al*., 1990) and Eastern Pacific (Tolentino and Mendoza, 2001) (Table 1). Values from the Western Pacific (Chen *et al*., 1990) indicate a faster growth rate than in other parts of the world but these results have not been validated.[3]

Based on the information summarized in Table 1, the Panel concludes that the species should be characterized as having low productivity. Still, the scalloped hammerhead shark ranked seventh and ninth (South and North Atlantic, respectively) of 20 Atlantic sharks in productivity (Cortes *et al*., 2012).

[3] Chen *et al*. 1990 assumed two vertebral bands are formed annually (see footnote **Error! Bookmark not defined.**).

1.1.3 Anthropogenic sources of mortality

Hammerheads are taken in target fisheries and as bycatch in fisheries operating within EEZs and in the high seas, and these are the main sources of anthropogenic mortality. However, the species are relatively less vulnerable to high seas pelagic longline fisheries than other pelagic sharks (Cortes *et al*., 2009) and their coastal distribution implies a higher vulnerability to fisheries on the continental shelf. The fins are traded internationally while the meat is mainly consumed locally[4] but a small portion of the meat is also traded internationally. A variety of fisheries are known to capture hammerheads, including small and large fisheries using gillnets, pelagic and bottom longlines (proposal). The capture of juvenile hammerheads in inshore fisheries has been documented in many parts of its range, representing an additional threat to the species (Dudley and Simpfendorfer, 2006; Hayes, Jiao and Cortes, 2009; proposal).

Reported catches of hammerhead sharks to FAO (including scalloped hammerhead, smooth hammerhead and other unidentified hammerhead sharks) show an increasing trend since the early 1990s, reaching the highest volume (6 187 tonnes) in 2010 (Figure 1). The largest share of the catches comes from the Western Central Pacific, Eastern Indian Ocean and the Eastern Central Pacific. Separating scalloped hammerhead from this total is difficult because most of the catch is reported at the family level (about 93 percent of the total). The reported catches of scalloped hammerhead reached a peak of 798 tonnes in 2002 and have been declining since then (FAO FishStat; proposal).

The trend and absolute catch volumes must be viewed with caution because of reporting problems. Under-reporting can be significant, particularly because fins are the main target product and finning is a common practice. Catches of hammerheads estimated from shark fin trade data are, for example, in the order of 50 000–90 000 tonnes per year (Clarke *et al*., 2006). However, improvements in the reporting of shark catches at lower taxonomic levels during the last decade (Fischer *et al*., 2012) may have affected the positive trend in reported catches.

Habitat degradation and pollution negatively affect the coastal ecosystems used as nursery areas by hammerheads. The effects of these changes on the populations are unknown but are likely to be minor relative to fisheries removals.

In Indonesia, scalloped hammerhead is one species of shark caught in longline fisheries for sharks, and as bycatch of tuna gillnet, and trawl fisheries in several offshore areas (White *et al*., 2006). Inshore fisheries in Southeast Asia are reported to exploit immature sharks heavily (SEAFDEC, 2006). In addition, sharks are targeted by foreign vessels in eastern Indonesian waters (Clarke and Rose, 2005).

1.1.4 Population status and trends

The sections below have been updated from the previous FAO Ad Hoc Panel report (FAO, 2010) with any new information presented in the proposal and made available by Panel participants.

Population size

The only population estimate available is that of Hayes, Jiao and Cortes (2009) for the Northwest Atlantic. The authors estimated a decrease in population size from between 142 000 and 169 000 individuals in 1981 to 24 500 individuals in 2005. The proposal also cites an estimates stock biomass of 2 466 tonnes in the Pacific coast of Mexico, but the reference could not be checked.

No worldwide population estimate is available.

Area of distribution

No estimate of distribution area is available, but given that this species is circumglobal in tropical and warm temperate waters, it can be concluded that it does not have a restricted distribution. Two studies

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[4] Only juveniles are consumed uncured; adults will be cured because of their high urea content.

of global genetic structure based on mitochondrial DNA showed strong geographic population subdivisions, corresponding to ocean barriers against migration. Duncan *et al*. (2006) found genetic differences among the Eastern Central Atlantic and Indo-Pacific populations. Chapman, Pinhal and Shivji (2009) concluded that the Northwest Atlantic, Caribbean Sea and Southwest Atlantic populations are genetically distinct from one another.

Population trend

The Panel evaluated all available sources of information on population trends. With few exceptions in the Pacific Ocean, the proposal does not include any new data on population trends compared with the last submission evaluated by the FAO Ad Hoc Panel (FAO, 2010).

Because this species occurs in several widely separated areas, and in distinct populations, no single abundance index can be applied to the species as a whole. Assessment of decline in abundance of the species can only be done using abundance indices from as many parts of the species' distribution as possible.

A number of abundance indices are available from different parts of the range (proposal; Table 2), but these are of varying reliability as indices for this species. In some cases, indices are for scalloped hammerhead as a species, in others for a complex of hammerhead sharks (*Sphyrna* spp.), in yet others for a broader shark complex.

Northwest Atlantic Ocean

Hayes, Jiao and Cortes (2009), based on a population assessment of scalloped hammerhead shark using two forms of surplus production model and incorporating multiple abundance indices (including those listed below), found an extent of decline of 83 percent from 1980 to 2005 (Figure 2). Their study indicates that the population has been increasing since 1995 and that there is a high probability of population recovery under most plausible scenarios, although the time to recovery varies with fishery removals (Table 3). However, they note that surplus production models are often overly optimistic in estimating rebuilding times.

Jiao, Hayes and Cortes (2009) conducted an assessment of the hammerhead shark complex (scalloped, smooth, great), concluding that the recent depletion level (extent of decline) would be 91–93 percent for 1980–2005, based on the ratio of current number to N_{MSY} and the fact that N_{MSY} is half of unexploited abundance.

Myers *et al*. (2007) summarized abundance trends for scalloped hammerhead and other shark species from a number of survey and commercial CPUE databases. A 31-year survey in North Carolina coastal waters (University of North Carolina) showed an instantaneous rate of decline of 0.127 for scalloped hammerhead, equivalent to a 98 percent extent of decline over the series (Figure 3). A SEAMAP survey in coastal waters of the southeast United States of America showed an instantaneous rate of increase for scalloped hammerhead of 0.094 over 17 years; the authors note that this was one of only 2 out of 31 shark abundance trends that showed an increase, and hypothesized that, as the individuals taken were mostly juveniles, the increase could reflect release of competition and/or predation owing to decline in abundance of large sharks. Commercial logbook and observer time series for all hammerheads pooled (noting that scalloped hammerhead was the most abundant of the three species in the group) showed extents of decline of 91 percent and 79 percent, respectively, over 14–15-year series, based on instantaneous rate of decline estimates. Myers *et al*. (2007) indicate an instantaneous rate of decline from a meta-analysis of trends from several surveys of approximately 0.05 (Figure 4).

Baum *et al*. (2003), apparently based on the same logbook data set as Myers *et al*. (2007), indicated a decline from 1986 to 2000 of 89 percent in commercial CPUE of pooled hammerhead species (Figure 5), and noted that this species group had declined in all fishing areas examined (Figure 6). Burgess *et al*. (2005) provided arguments that the declines in abundance indices observed by Baum *et al*. (2003) were probably greater than population declines, while Baum, Kehler and Myers (2005) in responding to this critique provided arguments that their estimates of population decline were robust.

Two survey indices from Ingram *et al*. (2005) are included in Table 2 as they were included in the proposal; however, these are considered of low reliability for scalloped hammerheads as they are based on all coastal sharks, of which scalloped hammerhead made up only 6–7 percent. Inspection of survey CPUEs for this complex showed no trend for the Atlantic coast of the United States of America for 1995–2005 and for the Gulf of Mexico coast 1995–2003, contrary to the interpretation in the proposal.

Central Eastern Atlantic

Fishery statistics of elasmobranch fisheries in Mauritania show fluctuations. Catch rates in 1997 were 27.3 kg/day at sea, peaked at 55.0 kg/day at sea in 2006 but in 2009 were 26.2 kg/day at sea. Scientific research cruises show that scalloped hammerhead abundance was variable from 1982 to 2008 but that there has been a statistically significant decrease of 95 percent since 1999 (Dia *et al*., 2012)**.** The same study also shows some indication of a decrease in the average size since 2006. This situation is representative of hammerhead fisheries in West Africa, which are a target artisanal fishery (Diop and Dossa, 2011) and bycatch in pelagic fisheries (Zeerberg, Corten and de Graaf, 2006). Catches of hammerheads reached 250 tonnes in 2007 and decreased to 150 tonnes in 2010.

Southwest Atlantic

Information from southern Brazil fisheries targeting hammerhead sharks (Kotas, personal communication), shows strong declines from 2000 to 2008 in two of three available series. Surface longline CPUE and bottom gillnet CPUE declined by 80 percent or more (Figure 7). Surface gillnet CPUE varied without trend (Figure 7). Catch and CPUE information from the same fishery (Kotas *et al*., 2008) indicates that these fluctuated by about a factor of five between 1995 and 2005, with a decline in the last years of the series (Figure 8). Catch would not be a strong abundance index. The targeted hammerhead fishery was abandoned after 2008 because the species had become rare (Kotas, personal communication).

Vooren, Klippel and Galina (2005) provide information from this area for an earlier period, 1993 to 2001. Annual landings of hammerheads (*S. lewini* and *S. zygaena* combined) in the main fishing ports in southern Brazil (Rio Grande and Itajai) increased from 30 tonnes in 1992 to 700 tonnes in 1994 and oscillated from 100 to 300 tonnes between 1995 and 2002 (Figures 9 and 10). From 2003 to 2005, landings were in the order of 300 tonnes, showing a declining trend since then, with the lowest level (55 tonnes) reported in 2008 (GEP/UNIVALI; CEPERG/ICMBIO). Vooren, Klippel and Galina (2005) noted that landings may not represent the actual catches of hammerheads in the region because of shark finning practices. The CPUE of the oceanic gillnet fisheries varied between 100 and 300 kg per trip without a clear trend from 1992 to 2002 (Figure 9). The CPUE of longline fisheries increased from 1993 to 2000 and then declined to 2002 (Figure 10). Effort data used to calculate CPUE were not corrected for changes in the size of gillnets or in number of hooks in the longline fisheries (C. Vooren, personal communication). The CPUE of recreational fisheries targeted to neonate hammerheads in shallow coastal waters also do not show a clear trend from 1999 and 2004, but possibly indicate a decline after 2001 (Figure 11). Based on the above results, the authors concluded that hammerheads were not threatened in southern Brazil but that effective conservation measures were needed to maintain the population at its current level of abundance.

Mediterranean Sea

The proposal (Annex 2) indicates that Ferretti *et al.* (2008) show a 99 percent decline in hammerhead sharks. However, Ferretti *et al*. (2008) indicate that *Sphyrna zygaena* is the only species of hammerhead covered by their indices, and that other species occurred only sporadically. Accordingly, this was not considered an appropriate index for scalloped hammerhead.

Western Indian Ocean

In an analysis of CPUE in large-mesh gillnets used to protect beaches from sharks in South Africa, Dudley and Simpfendorfer (2006) indicated a steady decline in abundance between 1978 and 2003; the level at the end is 35 percent of that at the beginning of the series, i.e. an extent of decline of 65 percent (Figure 12).

Eastern Indian Ocean

Information from the Tanjung Luar artisanal shark longline fishery off East Lombok shows a decrease from 15 percent to 2 percent in scalloped hammerhead portion of the catch (percentage by number) from 2001 to 2011 (ACIAR, 2011).

Western Pacific Ocean

De Jong and Simpfendorfer (2009) reported a decline of more than 85 percent in the hammerhead genus standardized CPUE over 44 years in a beach protection net programme in eastern Australia (northern Queensland). The 2009 Panel was advised that a range of 65–85 percent was consistent with the most recent analyses of this information (Simpfendorfer, personal communication; FAO, 2010).

A more recent study using data from the Queensland Shark Control Program (QSCP) showed an increasing trend in CPUE of *S. lewini* in mesh net in 1 of 10 locations monitored from 1996 to 2006 (Noriega *et al*., 2011). Among the possible reasons discussed by the authors for the local increasing trend in abundance are: greater abundance of prey; decreased abundance of other shark predators; changes in animal behaviour; improved fisheries management; and decreased fishing effort of the species (Noriega *et al*., 2011).

Gribble *et al*. (2005) presented catch and CPUE for all species combined in the Queensland shark fishery, in which *S. lewini* is one of the most important species (second in abundance and 18 percent of the total shark catch on 4 observed trips). Both catch and CPUE (all fisheries combined, kilograms per day) increased steadily from the late 1980s to the early 2000s (Figure 13). This index cannot be considered to be of high reliability for *S. lewini* as there are no data on species composition over time, and this could well have changed (FAO, 2010).

Eastern Pacific Ocean

Myers *et al.* (2005) found a 71 percent decline in a diver visual sightings index for scalloped hammerhead in a protected area in Cocos (Keeling) Islands, from 1992 to 2002. In the Gulf of Tehuantepec, Pacific coast of Mexico, the CPUE of *S. lewini* in the targeted artisanal longline fishery showed a marked decline from 1996 to 2001 (Figure 14; INP, 2006), following an apparent declining trend observed since 1987 (proposal).

The proposal documented declines in catches in other areas off Mexico, Costa Rica, Colombia, El Salvador, Panama, and Ecuador. However, the Panel could not obtain copies of these documents to verify the decline.

Other indices

Myers *et al*. (2007) presented information on change in length of scalloped hammerhead in the Northwest Atlantic, which indicates that there has been a slight decline over the period sampled (Figure 15). Dudley and Simpfendorfer (2006) found no trend in length of females, and a significant increasing trend for males, for the Southwest Indian Ocean over the period observed (1978–2003) (Figure 16).

1.2 Assessment relative to quantitative criteria

1.2.1 Small population

No global population estimate is available for this species, although an estimate for the Northwest Atlantic is available.

The CITES guideline is considered generally inappropriate for populations of commercially exploited marine species, except for a few species such as some sessile or semi-sessile species, some species with extremely low productivity, and some small endemics (FAO, 2001).

1.2.2 Restricted distribution

No guidelines for restricted area of distribution are provided in the CITES criteria, which indicate that thresholds should be taxon-specific (Conf. Res. 9.24 Rev. CoP15). FAO (2001) recommended that historical extent of decline in area of distribution would be a better measure of extinction risk than absolute value of distributional area, but that if no other suitable information is available and absolute area of distribution has to be used for an exploited fish population, analyses should be on a case-by-case basis as no numeric guideline is universally applicable.

No estimate of global distribution area is available, but given the circumglobal distribution of the species, it would not appear to be characterized by a restricted distribution.

1.2.3 Decline

Under the CITES criteria for commercially exploited aquatic species (Conf. Res. 9.24 Rev. CoP15), a decline to 15–20 percent of the historical baseline for a low productivity species might justify consideration for Appendix I. For listing on Appendix II, being “near” this level might justify consideration, “near” for a low productivity species being 20–30 percent of the historical abundance level (15–20 percent + 5–10 percent).

No overall population decline index is available for comparison with the guidelines. Indices in the individual areas are considered below. Most relevant indices available show declines consistent with the criterion threshold for listing a low productivity species on Appendix II.

In the Northwest Atlantic, the most robust index of abundance available (Hayes, Jiao and Cortes 2009) indicates a historical extent of decline of some 83 percent from 1980 to 2005. This assessment indicates that numbers were increasing in the period 1995–2005, and that the increase would be expected to continue under most plausible catch scenarios. The results of this assessment are consistent with an assessment of three hammerhead species pooled (Jiao, Hayes and Cortes 2009), which indicated a historical extent of decline of 91–93 percent in the period 1980–2005. These assessments incorporate other abundance index series available for the Northwest Atlantic (Table 2), some of which show conflicting trends. The 83 percent or 91–93 percent extents of decline would be consistent with the decline criterion for an Appendix I listing.

In the Central East Atlantic, fishery statistics of elasmobranch fisheries in Mauritania show fluctuations (Dia *et al*., 2012). Catch rates in 1997 were 27.3 kg/day at sea, peaked at 55.0 kg/day at sea in 2006 but in 2009 were 26.2 kg/day at sea. Scientific research cruises show that scalloped hammerhead abundance was variable from 1982 to 2008 but that there has been a statistically significant decrease of 95 percent since 1999 (Dia *et al*., 2012)**.** While the historical trends do not show a significant decline, the recent rate of decline would meet the Appendix I biological decline criteria**.**

For the Southwest Atlantic, two of three CPUE time series available for fisheries in southern Brazil historical extents of decline of the order of 80 percent or more for the period 2000–08. These are the most recent data available in this area, following earlier time series that show inconsistent trends. This fishery closed subsequent to 2008 because low abundance of the hammerhead sharks targeted no longer justified fishing.

For the Western Indian Ocean, the 64 percent historical extent of decline 1978–2003 of Dudley and Simpfendorfer (2006) would not be consistent with Appendix II decline guidelines, but does indicate a substantial, sustained decline.

In the Pacific Ocean, the historical extent of decline of 71 percent for 1992–2002 (Cocos [Keeling] Islands, Eastern Pacific) is consistent with an Appendix II listing, while the extent of decline of 65–85 percent over 44 years (northern Queensland, Western Pacific; De Jong and Simpfendorfer [2009]) is consistent with or at least very close to the decline criterion for Appendix II listing but for the latter area the species complex on which the analysis was based is mixed. There is evidence of an increasing trend in abundance in a location off Queensland, Australia, but the information is more localized and covers a shorter period than the study by De Jong and Simpfendorfer (2009).

Were trends due to natural fluctuations?

There is no indication in the materials consulted that natural fluctuations caused any of the observed abundance trends.

2. Comments on technical aspects in relation to trade, management and implementation issues

2.1 Trade aspects

Trade in scalloped hammerhead parts and derivatives

Scalloped hammerhead is exploited in many parts of its range, both in directed shark fisheries or as bycatch in fisheries for pelagic and demersal species. Recreational fisheries are or have been important in some parts of the range, for example the United States of America (Hayes, Jiao and Cortes, 2009), Australia (Gribble *et al*., 2005) and Brazil (Vooren, Klippel and Galina, 2005), but would not contribute significantly to trade.

Although meat, oil and hides are used locally, the meat and hides are traded internationally but at a smaller scale (proposal). The hammerhead meat is not as palatable as that of some other species (for example, porbeagle) but it is consumed and may be processed (salted and/or dried) for export. Limited trade in meat is documented in east Africa, West Africa and South America (sources cited in proposal, Section 6.3.1).

Fins are widely traded and demand is high. Trade statistics are not available, as this species (as most other shark species) does not have its own customs code under systems currently in international use (Harmonized Tariff Schedule). Recent work on quantities of fins of different shark species transiting the China, Hong Kong SAR fin market has helped clarify amounts of scalloped hammerhead fins in trade.

The China, Hong Kong SAR fin market has represented a substantial proportion of the global trade in shark fins: 65–80 percent in 1980–90, 50–65 percent from 1991–1995, 44–59 percent from 1996–2000, 30–50 percent since 2000 (Clarke, 2008). The decline in China, Hong Kong SAR’s share of world trade is attributed to increasing trade through mainland China, where statistics are difficult to obtain (Clarke, Milner-Gulland and Cemare, 2007). Despite the estimated decline over time in share of the world trade transiting China, Hong Kong SAR, total imports to China, Hong Kong SAR increased in the 1990s (Figure 17), suggesting that total world trade in shark fins was increasing during this period.

Hammerhead fins are highly valued in the international fin trade because of the fin size and high needle (ceratotrichia) count (Rose, 1996). The high prices for the various species (USD88–135/kg, proposal) provide evidence of high demand (Clarke, Milner-Gulland and Cemare, 2007).

Fins of scalloped hammerhead and smooth hammerhead (*S. zygaena*) together made up 4.4 percent of fins traded in the China, Hong Kong SAR market (Clarke *et al*., 2006; Table 5) between November 2002 and February 2004. With respect to the origin of the fins in trade, a recent study by Chapman, Pinhal and Shivji (2009) indicated that 21 percent of a sample of scalloped hammerhead fins in the China, Hong Kong SAR market was derived from the Western Atlantic populations.

Overall, it seems clear that scalloped hammerhead fins are traded internationally, although a relatively minor component of the overall fin trade. Hammerhead sharks are a target species in some areas, while in others they are taken as bycatch in fisheries targeting tuna-like or other shark species. Ease of processing and storage of dried fins facilitates trade, and the products command relatively high prices in trade, e.g. EUR27.50 per kilogram of frozen hammerhead fin in Europe versus EUR8.13 per kilogram of frozen thresher shark and EUR7.71 per kilogram for blue shark (Fowler and Seret, 2010); and EUR75 per kilogram of dried fin in West Africa (IMROP, 2006).

Basis for Article II paragraph (2b) ("look-alike") Appendix II listing of great hammerhead shark and smooth hammerhead shark

As indicated in the CITES listing criteria (Resolution Conf. 9.24 Rev. CoP15), listing of the two shark species named above could be justified if the parts and derivatives of these species in trade resemble those of the listed Appendix II species (scalloped hammerhead in this case) to the extent that enforcement officers would be unable to distinguish them.

The fins of hammerhead sharks have a similar morphology (thin, falcate, dorsal fin height higher than base) that facilitates their identification by traders. China, Hong Kong SAR traders are generally able to identify fins in trade to species or to small species groups, as indicated by a comparison of categories of shark fins used by traders in the China, Hong Kong SAR market with species identifications based on DNA testing (Clarke *et al*., 2006). Scalloped and smooth hammerhead were not separated by traders but pooled in a single category, with a high rate of correspondence between the market category and the identification to this species pair (95 percent). Fins of the great hammerhead ("gu pian") are separated from the rest with a rate of correspondence of 86 percent.

This study indicates that it is possible to identify shark fins in trade to species, with the important exception of scalloped and smooth hammerhead, which are not currently separated. However, expert knowledge and experience are doubtless required to attain the level of identification demonstrated in the China, Hong Kong SAR market. Accordingly, this study supports the argument that enforcement officers with general knowledge (possibly even with some additional identification materials) would have difficulty in identifying fins in trade to species. Available DNA technology could provide a backup to identification (Holmes, Steinke and Ward, 2009), but current technology is generally considered not to provide useful techniques for routine separation of species at customs posts.

Scalloped and smooth hammerhead fins cannot be distinguished, or are not distinguished, even with expert knowledge. Fins of all three hammerhead species are quite similar, to the extent that separating them would be difficult for non-experts. It is not clear why the other species in the family Sphyrnidae were not proposed to be listed as "look-alikes". For fresh meat (MarViva, 2012) and fins (PEW guide), there exist guides to identify hammerhead sharks by species; however, the visual species identification based on processed shark products (in particular meat, cartilage and oil, lower lobe of caudal fin) is difficult and this could present a problem for customs officers. The Panel had different opinions regarding the feasibility of visual identification of dried fins by non-experts.

2.2 Fisheries management aspects

Scalloped hammerhead is distributed across several EEZs and in the high seas, being therefore under different management regimes throughout its range. Information on management measures of relevance to the species conservation adopted by some of the main fishing nations (identified based on the reported catches to FAO) and RFMOs is summarized below. The measures vary from those aimed at controlling the direct take in targeted fisheries (quotas, effort control and minimum sizes), to the protection of reproduction and nursery areas (fishing exclusion zones), to measures aimed reducing bycatch mortality. In this regard, the most commonly adopted measure is the banning of finning, i.e. removal of fins and discard of the body carcass.

The effectiveness of the finning ban in reducing mortality rests on the disincentive to keep the specimens incidentally caught, or even to target areas where the species is known to occur, owing to limited storage space on board. It also depends on the post-capture survival of the individuals incidentally caught. A recent study concluded that fisheries that market shark meat are not affected by a finning ban whereas fisheries targeting shark fins released more sharks with a finning ban in place (Gilman *et al*., 2008). In 2004–06, in the Hawaiian longline fishery, following a finning ban, almost all sharks were released, although some were dead on release. Minimum mortality estimates declined substantially with the finning ban from 81.9 percent to 25.6 percent in deep sets and from 61.3 percent to 9.1 percent in shallow sets (Walsh, Bigelow and Sender, 2009). There are few specific studies on the post-capture mortality of hammerhead sharks in commercial fisheries. Using data from a catch and release field study in Florida, the United States of America, Hueter *et al*. (2006) estimated that 60 percent of bonnetheads (*Sphyrna tiburo*) survived the stress of gillnet capture, tagging and release.[5] Heuter and Manire (1994) estimated a post-capture mortality of 55.6 percent of small scalloped hammerhead caught in gillnets in Florida. In the southeastern Australian commercial gillnet fishery for sharks, Braccini, Van Rijn and Frick (2012) estimated that about 90 percent of the smooth hammerhead sharks arrive dead on deck, and of the live individuals released about 60 percent survive. The overall post-capture mortality of sharks in pelagic longline fisheries tends to be lower, usually less than 30 percent (Cosandey-Godin and Morgan, 2011). For blue sharks, Campana, Joyce and Manning (2009) estimated that, on average, 19 percent of the individuals incidentally caught and released in longline fisheries end up dying. Available estimates for other species and gear types vary widely (Cosandey-Godin and Morgan, 2011). Solely on the basis of post-capture mortality, it can therefore be concluded that finning bans will have variable effects on reducing bycatch mortality on hammerheads. However, they can be effective in controlling the impact of direct fisheries that have been motivated by the fin trade.

The level of compliance plays a key role in the effectiveness of the management measures. Considering that hammerheads are among the most frequently cited species taken in illegal fishing activities (Lack and Sant, 2008), it can be presumed that compliance is a problem in many parts of the species range.

Approximately half of the fishing nations reporting catches of hammerhead sharks to FAO have not yet adopted a national plan of action for sharks (Fischer *et al*., 2012).

Finning bans have been adopted by several of the top shark-fishing nations, including countries of the European Union (Member Organization), the United States of America, Brazil, Mexico, Taiwan Province of China, Argentina, Sri Lanka, New Zealand, Nigeria, Canada and Australia. The measure has also been adopted by several RFMOs: ICCAT, GFCM, IOTC, IATTC, NAFO, SEAFO, WCPFC, CCAMLR and NEAFC (Fischer *et al*., 2012).

ICCAT has also recently issued a recommendation (08/2010) for parties to prohibit retaining onboard, transshipping, landing, storing, selling, or offering for sale any part or whole carcass of hammerhead sharks of the family Sphyrnidae (except for the *Sphyrna tiburo*), taken in the convention area in association with ICCAT fisheries. Parties are further recommended to promptly release unharmed, to the extent practicable, hammerhead sharks when brought alongside the vessel. Developing coastal States that catch hammerheads for local consumption are exempted from these measures.

In the United States of America, the scalloped hammerhead, great hammerhead and smooth hammerhead sharks are managed as part of the Atlantic Large Coastal Shark Complex, which includes eight additional species of sharks. Harvest of any species in the group must be in accordance with a total quota for the complex, limited entry, time-area closures, recreational bag limits, and the requirement to land sharks with fins attached to the body (NMFS, 2006). Given the overfished status of scalloped hammerhead, a specific quota for the species is now under consideration (NMFS, 2012). In Brazil, hammerhead sharks are indirectly affected by laws prohibiting finning, restricting the length of pelagic gillnets and banning trawl and gillnet fishing at a variable distances from shore (Fischer *et al*., 2012; proposal), coinciding with the species nursery areas. Shark fishing has been banned in the entire EEZs or in specific protected areas of many countries described in the proposal, including several Pacific Small Island Developing States. Access control through licensing is adopted in the directed artisanal fishery in the Gulf of Tehuantepec, Pacific coast of Mexico (INP, 2006). Another measure cited in the proposal is the banning of the trade of sharks and shark products, which has been adopted for example by the Bahamas, Maldives and the Marshall Islands.

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[5] This study was done under very different conditions than commercial fishing and, therefore, might not be representative for normal fishing mortalities.

The scalloped hammerhead was recently (2012) included in Appendix III of CITES by Costa Rica, meaning that trade in the species can only occur with CITES documentation.

2.3 Implementation issues

2.3.1 Introduction from the sea

Based on current knowledge of distribution, hammerheads are primarily species of continental shelf and coastal waters, and are uncommon in oceanic waters (Compagno, 1984; Fowler *et al*., 2005). Most of the fisheries that exploit these species operate within continental-shelf waters rather than in the open ocean. As such, most harvests would be from waters within state EEZs, for which the IFS provisions of CITES would not apply.

2.3.2 Basis for findings: legally obtained, non-detrimental

Non-detriment findings are the responsibility of the exporting country and must show that exports are not detrimental to survival of the species, that is, that they are consistent with sustainable harvesting. Development of an NDF requires appropriate scientific capacity, biological information on the species, and an approach to demonstrating that exports are based on sustainable harvests. The quality of NDFs is assured by review in the Scientific Committees of CITES (Animals and Plants Committees) and in individual parties. FAO (2004, paragraphs 28–29) provides some guidance on NDFs in a fisheries context.

For the Northwest Atlantic, NDFs could be based on the recent assessments of this species (Hayes, Jiao and Cortes, 2009; Jiao, Hayes and Cortes, 2009). The United States Fisheries Management Plan (NMFS, 2006) treats scalloped hammerhead as 1 of 11 species in a large coastal shark complex, and as such does not include a quota for this species alone, but harvest levels consistent with stock rebuilding have been determined (Hayes, Jiao and Cortes, 2009) and NDFs could be issued for harvests consistent with such levels.

For other parts of the distribution, no species-specific assessments are available that could provide a basis for NDFs. In general, NDFs are required to be made at the species-specific level. However, if decided by Parties, a combined NDF of several species may be issued, for example stony coral.

2.3.3 Identification of products in trade

Fins are the principal product in trade. Two species of hammerheads (scalloped and smooth) are not differentiated even by expert traders in the market; however, great hammerhead can often be differentiated by fin traders (Clarke *et al*., 2006). As mentioned above, meat and other shark parts are also traded internationally in a lower portion and are difficult to identify.

Accurate recording of international trade in sharks is seriously hampered by the absence of any species-specific reporting mechanism. To address this, the Conference of the Parties should encourage the WCO to establish specific headings within the standardized tariff classification of the Harmonized System to record trade in sharks and their products at the species level. The Panel recognized the progress under the WCO in developing specific codes for sharks, and encourages this work to continue.

2.3.4 "Look-alike" issues

Non-experts would probably have difficulty separating shark fins and other products in trade, but regional and global identification guides for shark fins exist or are currently in preparation (with some reservations expressed regarding the feasibility of reliable visual species identification of dried fins). Moreover, CITES does not have clear standards for making decisions on whether to list species under

Article II paragraph 2(b). For example, other shark species will meet look-alike criteria because of the difficulty of identifying processed products.

2.4 Likely effectiveness of a CITES listing for the conservation of the species

Some measures have already been taken by most RFMOs and many countries in terms of finning regulations. In addition, ICCAT has adopted measures of no retention for hammerhead sharks while other RFMOs and States have adopted more general shark conservation measures, e.g. gear regulations, size limitations and area/seasonal closures. The Panel believes that a wide adoption of such measures also by coastal States would play an important role for hammerhead conservation and shark conservation in general. In principle, a CITES Appendix II listing will be more effective for fisheries targeting sharks for their fins that enter international trade. However, a CITES Appendix II listing will have limited effect if the shark catches are consumed and traded locally (see also FAO, 2012).

An Appendix II listing for hammerhead shark might improve monitoring of catches of species that enter international trade (through documentation of trade flows) and assessment of sustainabililty of harvests (through provision of NDFs). A CITES listing would support measures already taken by RFMOs and nations, for example helping to close loopholes in the regional reporting of hammerheads. Few national markets for hammerhead shark products exist, so most of the products in trade would move internationally and would thus come under the Appendix II regulatory provisions. However, it is also possible that enhanced regulation of trade would encourage more sustainable use of this species and thus reduce pressure on stocks.

A CITES listing may also improve enforcement of existing national shark fishing bans by improving documentation of nation of origin.

For the two species proposed for listing under Article II paragraph 2(b) the same comments are relevant.

3. Conclusion

The Panel concluded that, based on the available evidence presented, the biological criteria for a listing of scalloped hammerhead (*Sphyrna lewini*) in CITES Appendix II are met. The other two species proposed for an Appendix II listing in accordance with Article II paragraph 2b (look-alike clause), great hammerhead shark (*S. mokarran*) and smooth hammerhead shark (*S. zygaena*), meet the criteria.

The fins are traded internationally while the meat is mainly consumed locally but a small portion of the meat is also traded internationally. This may have implications for considering the inclusion of other hammerhead species under the “look-alike” clause.

Fins from scalloped hammerhead are in high demand and are easily preserved and transported, and the species coexists with other high-value pelagic species and is readily taken as bycatch.

Risk in the Northwest Atlantic may be mitigated by the existence of a United States NMFS Fishery Management Plan for Highly Migratory Species, including scalloped hammerhead shark, which is managed as 1 of 11 species in a “large coastal shark” complex. Risks may be also mitigated by the existence of shark-finning bans in several countries and RFMOs, although provisions of these bans and thresholds (for example, ratio of fins to carcass weights in landings) are variable and compliance is likely to be variable. The effectiveness of finning bans in reducing mortality is likely to be low in some fisheries, such as in gillnet fisheries, because of the high post-capture mortality.

An Appendix II listing for hammerhead shark might improve monitoring of catches at the species level (through documentation of trade flows) and assessment of sustainability of harvests (through provision of NDFs). With the recent listing of the species in Appendix III, improvements in trade monitoring are expected to occur independently of the Appendix II listing. Few national markets for hammerhead shark products exist, so most of the products in trade would move internationally and would thus come under the Appendix II regulatory provisions. However, it is also possible that enhanced regulation of trade would encourage more sustainable use of this species and thus reduce pressure on stocks. The same comments are relevant for the two species proposed for listing under Article II paragraph 2(b).

The Panel also noted that the difficulty of identifying products in trade and making an NDF might limit the effectiveness of this CITES listing. Requirements for additional information will create a burden that may need to be addressed through capacity building, particularly in developing countries. However, this is not unique for potential CITES listings; it applies in general to all new management measures or regulations.

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TABLES AND FIGURES

Table 1

Information for assessing productivity of scalloped hammerhead shark

Parameter	Information	Productivity	Source
Intrinsic rate of increase	a. NW Atlantic: 0.082 (λ = 1.086)	a. Low	a. Cortes, 2002
	b. Atlantic: 0.105	b. Low	b. Cortes *et al*., 2009
	c. W Pacific: 0.472 (λ = 1.600)	c. undetermined[1]	c. Cortes, 2002
	d. W. Indian Ocean: 0.103	d. Low	d. Dudley and Simpfendorfer, 2006
	e. R_{2M} = 0.028	e. Low	e. Smith, Au and Show, 1998
	f. S Atlantic: R = 0.121 N Atlantic: R = 0.096	f. Low	f. Cortes *et al*., 2012
Natural mortality	M = 0.129	Low	Smith, Au and Show, 1998
Age at maturity	a. W Indian Ocean: 11 years	a. Low	a. Dudley and Simpfendorfer, 2006
	b. Females: 15 years	b. Low	b. Smith, Au and Show, 1998
Maximum age	a. NW Atlantic: 30.5 years	a. Low	a. Piercy *et al*., 2007
	b. W Indian Ocean: 30 years	b. Low	b. Dudley and Simpfendorfer, 2006
	c. E. Pacific: 18.6 years	c. (Medium)[3]	c. Tolentino and Mendoza, 2001
	d. 35 years (Eastern Pacific)	d. Low	d. Smith, Au and Show, 1998
	e. 14 years (Western Pacific)[2]	e. Low	e. Chen *et al*., 1990
von Bertalanffy K	a. NW Atlantic: Male (0.13) Female (0.09)	a. Low	a. Piercy *et al*., 2007
	b. W Indian Ocean: 0.057	b. Low	b. de Bruyn, 2000, cited in Dudley and Simpfendorfer, 2006
	c. W Pacific: Male (0.22) Female (0.25)[4]	c. Undetermined	c. Chen *et al*., 1990, cited in proposal
	d. E Pacific: Male (0.13) Female (0.16)	d. Low	d. Tolentino and Mendoza, 2001
Generation time	a. NW Atlantic: 16.7 years	a. Low	a. Cortes, 2002
	b. W Indian Ocean: 18.3 years	b. Low	b. Dudley and Simpfendorfer, 2006
	c. W Pacific: (5.7 years)[5]	c. Undetermined	c. Cortes, 2002

[1] The Cortes 2002 paper used the Chen *et al*. 1990 age and growth information and therefore came to the conclusion of a high productivity.

[2] Chen *et al*. 1990 assumed two vertebral bands are formed annually. Passeroti *et al*. (2010) validated annual band deposition in hammerhead sharks, which has been accepted by most authors. Assuming one annual band, the estimate by Chen *et al*. would be about 28 years maximum age.

[3] Highest age observed but probably not maximum for species

[4] This estimate is based on the Chen *et al*. 1990 assumption and not accepted by the Panel.

[5] Ibid.

TABLE 2

Decline indices for scalloped hammerhead Criterion	**Index**	**Trend**	**Basis**	**Coverage**	**Reliability**	**Source**
Northwest Atlantic	Abundance estimate from population assessment	EOD 83%	Surplus production model, multiple indices, 1980–2005 Scalloped hammerhead	Atlantic coast United States of America	Assessment based on multiple surveys (5)	Hayes, Jiao and Cortes (2009)
	Abundance estimate from population assessment	EOD 91–93%	Surplus production model for mixed hammerhead species, probabilistic, multiple indices, 1980–2005	Atlantic coast United States of America	Assessment based on multiple surveys, for mixed species (5)	Jiao, Hayes and Cortes (2009)
	CPUE, UNC research survey	EOD 98%	Instantaneous decline of 0.127 over 31 years (1973–2003) Scalloped hammerhead	North Carolina coastal	Designed survey (5)	Myers *et al*. (2007) Table S5
	CPUE, SEAMAP survey	Increase	Instantaneous increase of 0.094 over 17 years (1989–2005) Scalloped hammerhead	Southeast coast United States of America	Designed survey (5)	Myers *et al*. (2007) Table S5
	CPUE, commercial logbook (all hammerheads)	EOD 91%	Instantaneous decline of 0.158 over 15 years (1986–2000) mixed	Northwest Atlantic	Commercial data (3)	Myers *et al*. (2007) Table S5
	CPUE, commercial observers (all hammerheads)	EOD 79%	Instantaneous decline of 0.110 over 14 years (1992–2005) mixed	Northwest Atlantic	Commercial observer data (4)	Myers *et al*. (2007) Table S5
	CPUE, commercial logbooks (all hammerheads, mainly *S. lewini*)	EOD 89%	Calculated by authors, 1986–2000 mixed	Northwest Atlantic	Commercial logbooks (3)	Baum *et al*. (2003)
	CPUE, longline survey	No trend	Inspection of figure, 1995–2005 Mixed all sharks	Atlantic coast United States of America	Pooled coastal sharks, *S. lewini* is 6% of total (0)	Ingram *et al*. (2005), Figure 39
	CPUE, longline survey	No trend	Inspection of figure, 1995–2003 Mixed all sharks	Gulf of Mexico, United States of America	Pooled coastal sharks, *S. lewini* is 7% of total (0)	Ingram *et al*. (2005), Figure 42
Southwest Atlantic	CPUE, surface gillnet	EOD 80% or more	Inspection of figure, 2000–08 Scalloped hammerhead	Southern Brazil	Unstandardized CPUE, (3)	J.E. Kotas, personal communication (FAO, 2009)
	CPUE, bottom gillnet	EOD 80% or more	Inspection of figure, 2000–08 Scalloped hammerhead	Southern Brazil	Unstandardized CPUE, (3)	J.E. Kotas, personal communication (FAO, 2009)

Decline indices for scalloped hammerhead Criterion	**Index**	**Trend**	**Basis**	**Coverage**	**Reliability**	**Source**
	CPUE, surface longline	No trend	Inspection of figure, 2000–08 Scalloped hammerhead	Southern Brazil	Unstandardized CPUE (3)	J.E. Kotas, personal communication (FAO, 2009)
	CPUE (*S. lewini* and *S. zygaena*) gillnet fisheries	No trend	Inspection of figure, 1992–2002 Mixed	Southern Brazil	Uncorrected effort data (1–2)	Vooren, Klippel and Galina (2005)
	CPUE (*S. lewini* and *S. zygaena*) longline fisheries	Increase from 1993 to 2000, decline from 2000–02	Inspection of figure, 1992–2002 Mixed	Southern Brazil	Uncorrected effort data (1–2)	Vooren, Klippel and Galina (2005)
	CPUE (*S. lewini* and *S. zygaena*) recreational fisheries	No trend, possible decline from 2001	Inspection of figure, 1999–2004 Mixed	Southern Brazil	Commercial data (2)	Vooren, Klippel and Galina (2005)
Central Eastern Atlantic	Scientific survey CPUE	Variable 1982–1999. 95% decline 1999–2008	Regression of data provided by Dia *et al*. (2012) Scalloped hammerhead	Mauritania	Scientific survey (4)	Dia *et al*. (2012)
Western Indian Ocean	CPUE, shark protection nets	EOD 65%	Inspection of figure, 1978–2003 Scalloped hammerhead	South Africa	Good species identification, designed for sharks (5)	Dudley and Simpfendorfer (2006), Figure 2
Eastern Indian Ocean	Fraction of scalloped hammerhead in commercial catch	87% decrease from 2001 to 2011	Calculated from fraction in catch Scalloped hammerhead	East Lombok, Indonesia	Based on fraction of species in catch (1)	ACIAR (2011) White, Barton and Potter (2008)
Western Pacific Ocean	CPUE, all fisheries, all sharks	Increasing trend	Inspection of figure, 1978–2003 All shark species combined	Queensland, Australia	All fisheries combined (0)	Gribble *et al*. (2004), Figure 2
	CPUE, shark protection nets	EOD 65–85%	Provided by authors mixed	Queensland, Australia	Standardized CPUE (4)	De Jong and Simpfendorfer (2009)
	CPUE, mesh net, *S. lewini*	Increasing trend	Provided by authors Scalloped hammerhead	Queensland, Australia	Standardized CPUE (5)	Noriega *et al*. (2011)
Eastern Pacific Ocean	Diver sightings index	EOD 71%	Provided by authors Scalloped hammerhead	Cocos Islands, Costa Rica	Visual sightings (2)	Myers *et al*. (2005)
	CPUE, targeted shark fishery	Decreasing trend	Inspection, figure 1996–2000 Scalloped hammerhead	Gulf of Tehuantepec, Mexico	Unstandardized CPUE (3)	INP (2006), Figure 12.

Note: EOD: extent of decline.

Source: FAO (2012). Reliability values based on FAO (2001).

TABLE 3

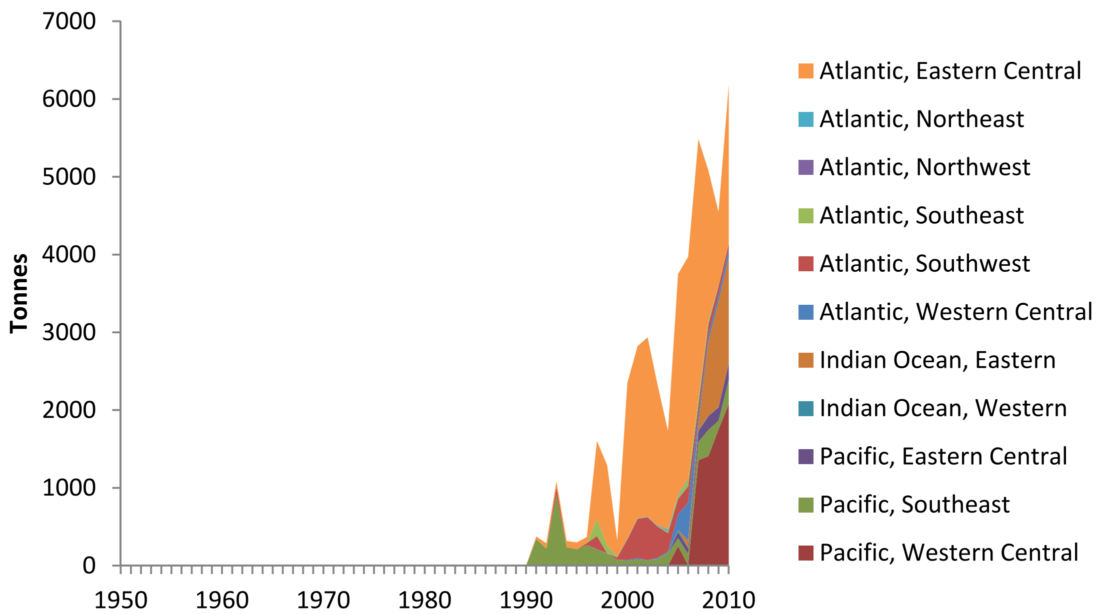
Probability that the stock of scalloped hammerheads will rebuild (i.e. attain a final population size greater than N_{MSY}) in 10, 20 and 30 years under several constant-catch scenarios (relative to the catch in 2005) using the BASE scenario with the Fox surplus-production model

		Percent of 2005 catch (number)			
		50	69	100	150
Time frame	No catch	(2 068)	(2 853)	(4 135)	(6 203)
10 years	95	85	70	58	20
20 years	99	96	92	86	50
30 years	99	98	96	91	63

Source: Hayes, Jiao and Cortes (2009).

FIGURE 1

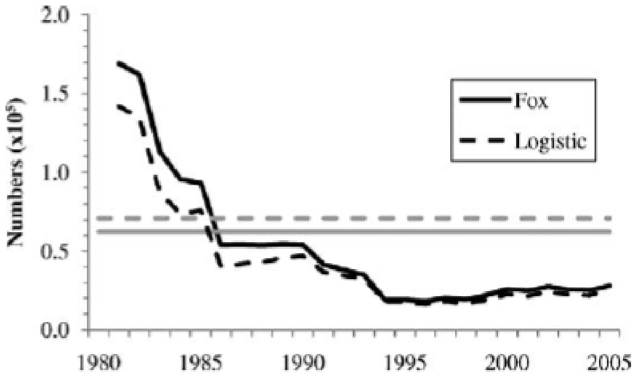
Reported catches of hammerheads, 1950–2010



Source: FAO FishStat.

FIGURE 2

Scalloped hammerhead population estimates from two models, 1981–2005

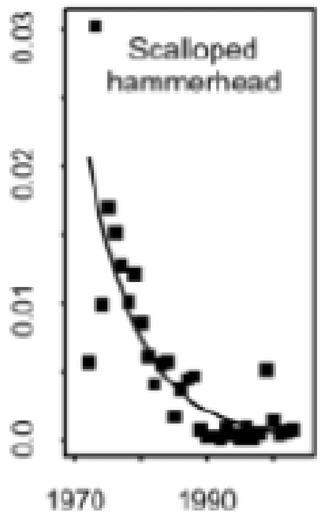


Note: Grey lines are MSY levels for the two models.

Source: Hayes, Jiao and Cortes (2009).

FIGURE 3

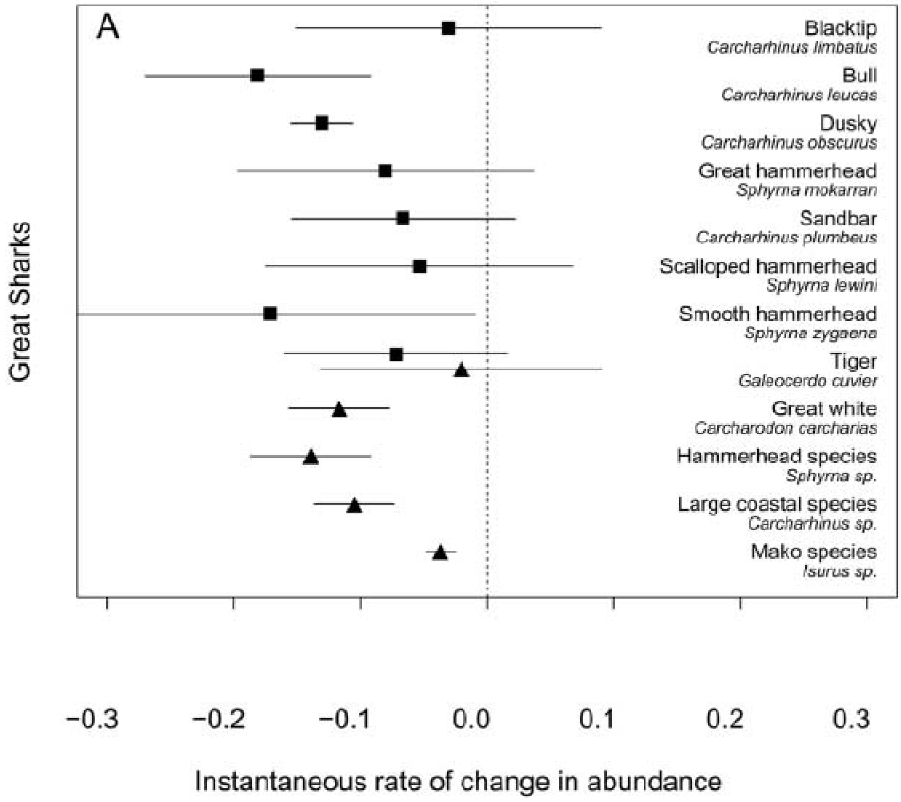
Abundance trend, scalloped hammerhead, UNC survey



Source: Myers *et al*. (2007), Figure 1.

FIGURE 4

Instantaneous rate of change in abundance, meta-analysis of multiple research surveys

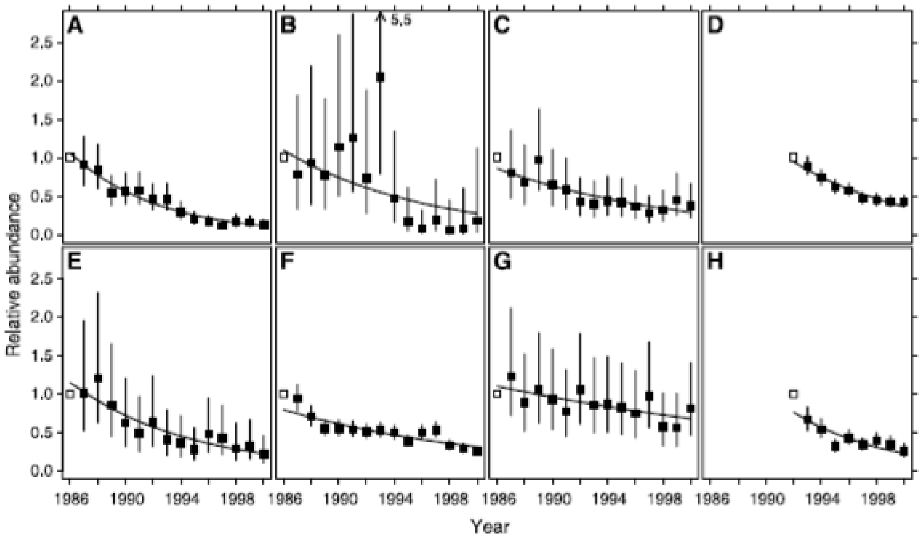


Note: Mean time span of surveys 28 years.

Source: Myers *et al*. (2007), Figure 2.

FIGURE 5

Changes in abundance indices

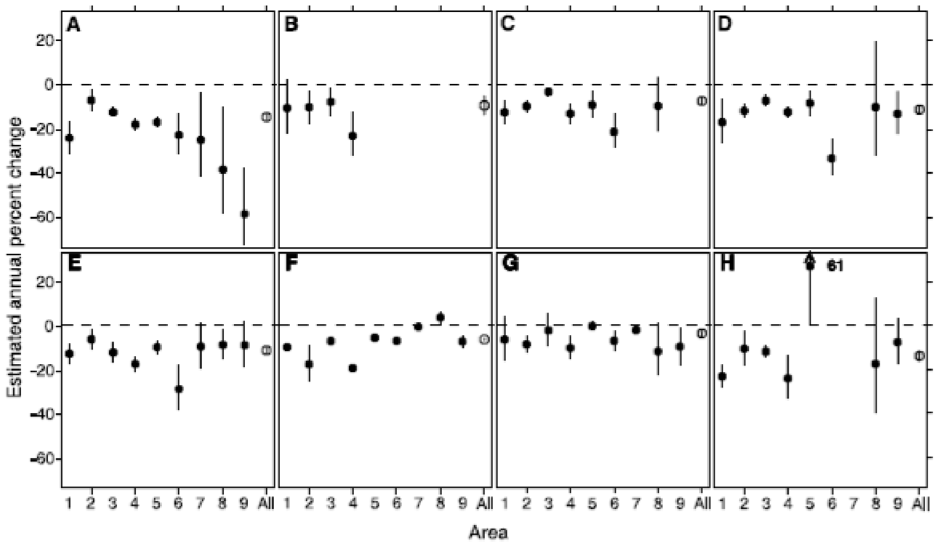


Note: A = hammerhead sharks pooled.

Source: Baum *et al*. (2003).

FIGURE 6

Annual rate of change in abundance, 1986–2000, in 10 subareas of the northwest Atlantic

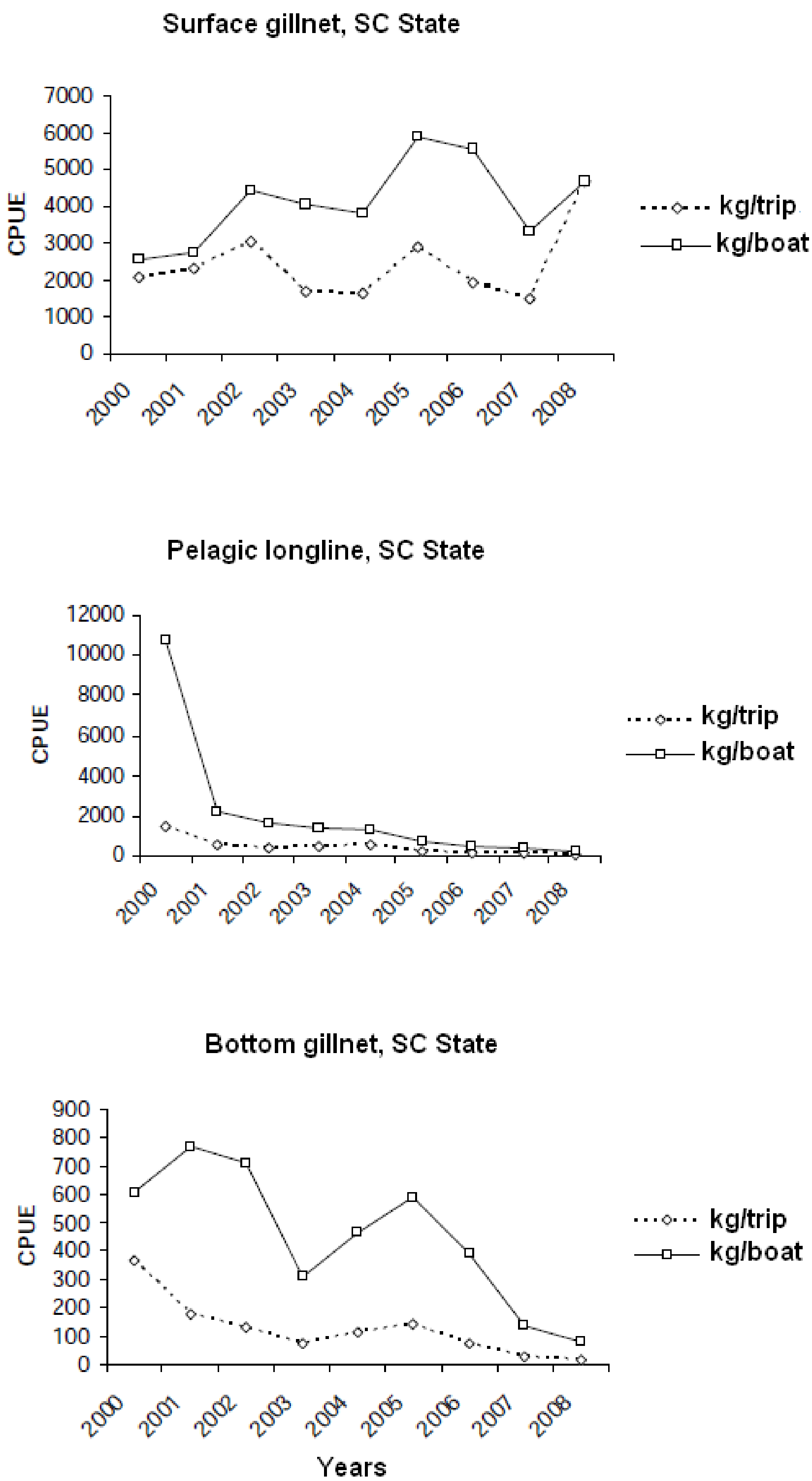


Note: A = hammerhead sharks pooled.

Source: Baum *et al*. (2003).

FIGURE 7

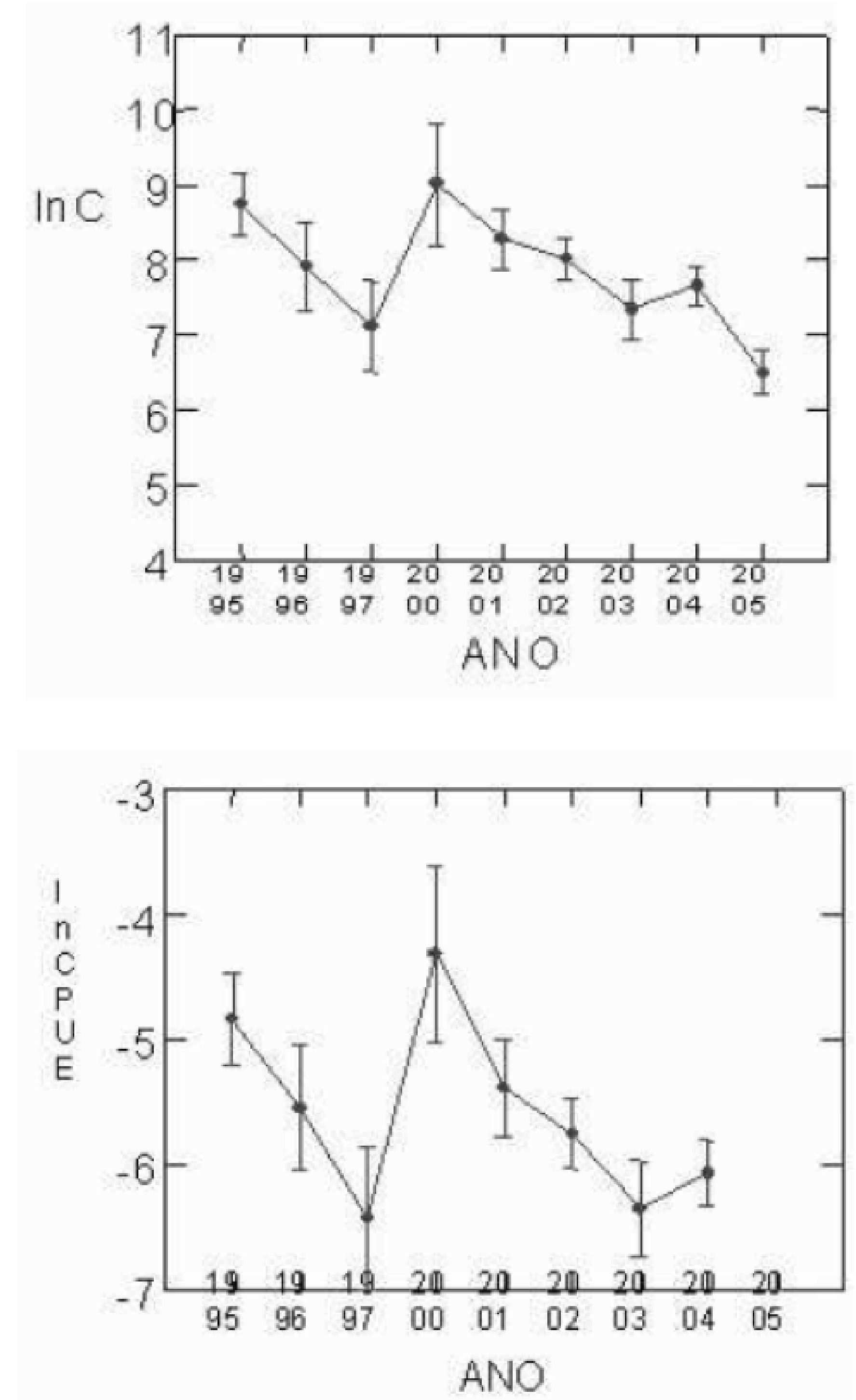
CPUE, scalloped hammerhead, southern Brazil



Source: J.E. Kotas, personal communication (FAO, 2009).

FIGURE 8

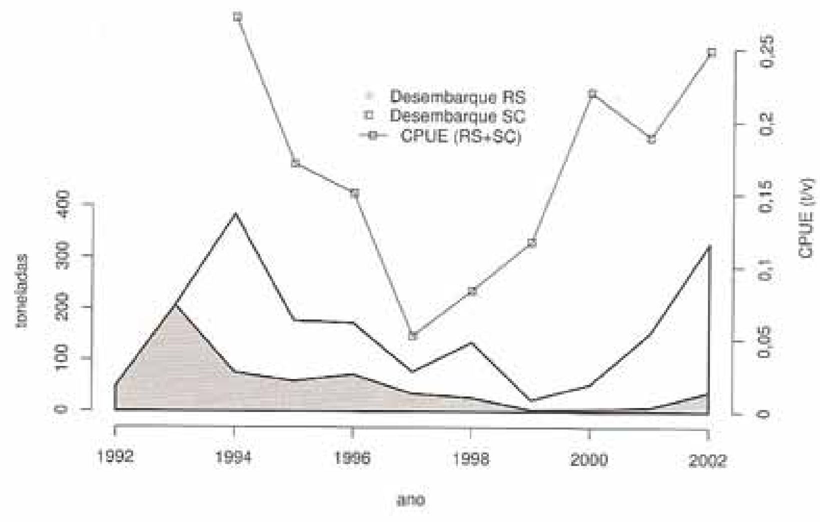
Catch (kilograms) (top) and CPUE (kilograms per square metre of net) (bottom) of pooled scalloped and smooth hammerheads, surface gillnets, southern Brazil



Source: Kotas *et al*. (2008).

FIGURE 9

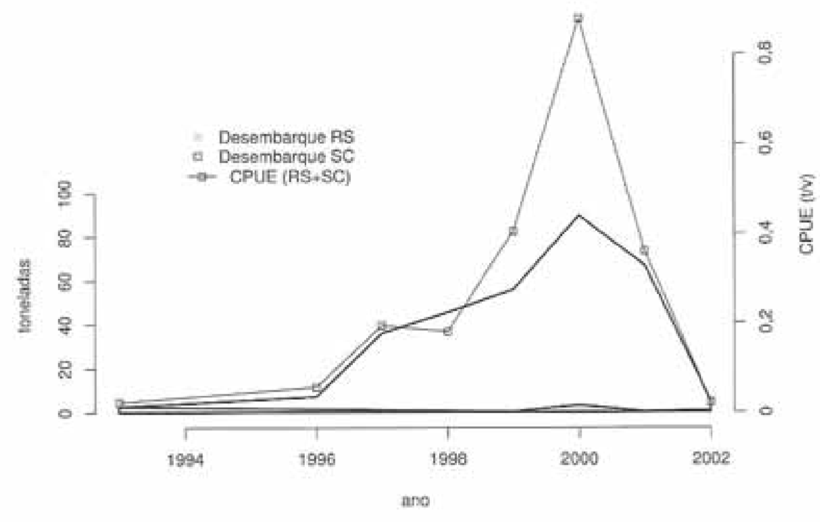
Landings and CPUE of oceanic gillnet fisheries in southern Brazil



Source: Vooren, Klippel and Galina (2005).

FIGURE 10

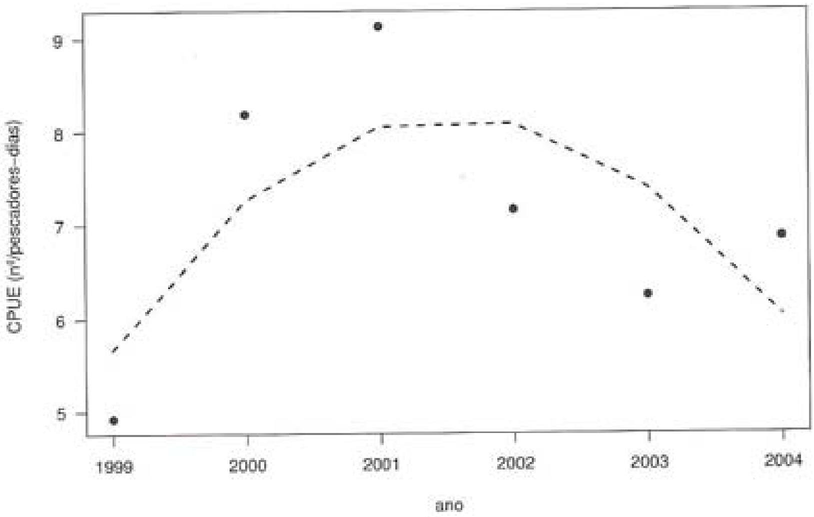
Landings and CPUE of longline fisheries in southern Brazil



Source: Vooren, Klippel and Galina (2005).

FIGURE 11

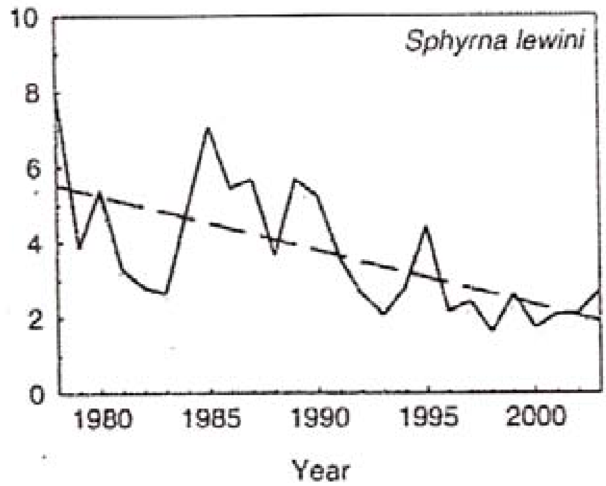
Daily CPUE (numbers/fisher) of the recreational fishery targeted to neonate hammerheads in southern Brazil



Source: Vooren and Klippel (2005).

FIGURE 12

Annual CPUE of scalloped hammerhead in the KwaZulu-Natal Beach Protection Programme, 1978–2003

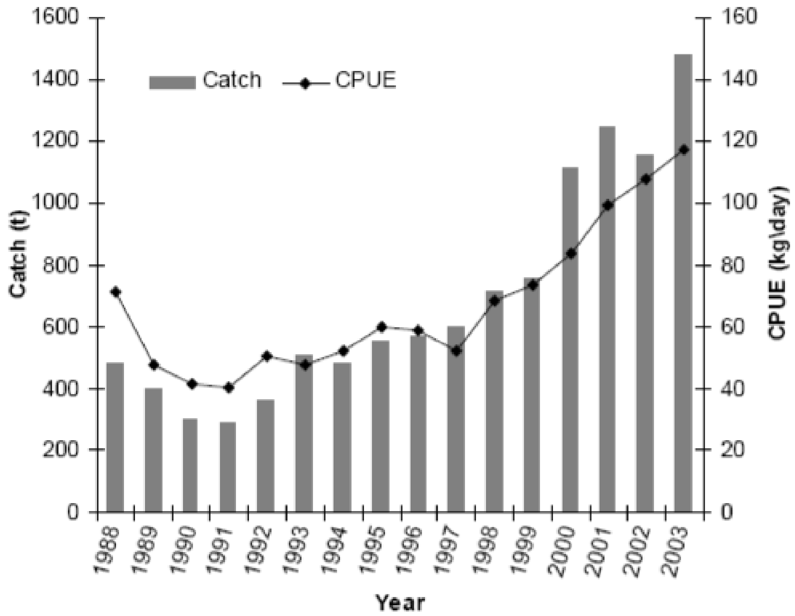


Note: Units are number per kilometre of net per year.

Source: Dudley and Simpfendorfer (2006).

FIGURE 13

Annual catch and CPUE, all fisheries combined, all shark species combined, Australian east coast

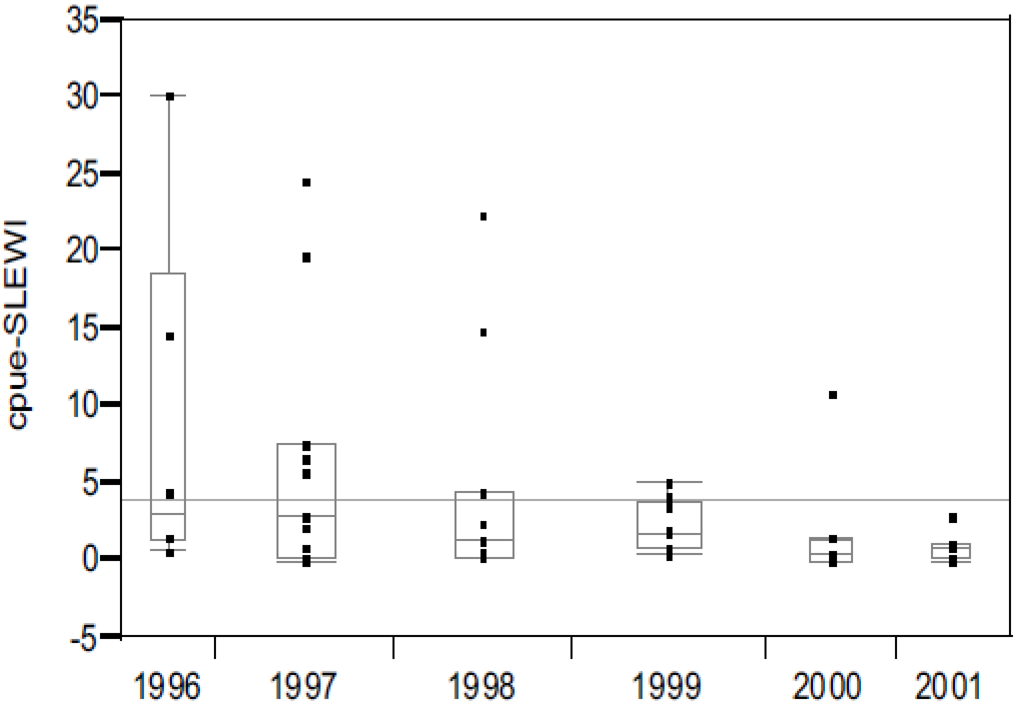


Note: S. lewini made up 18 percent of the total catch on 4 observed trips.

Source: Gribble *et al*. (2005).

FIGURE 14

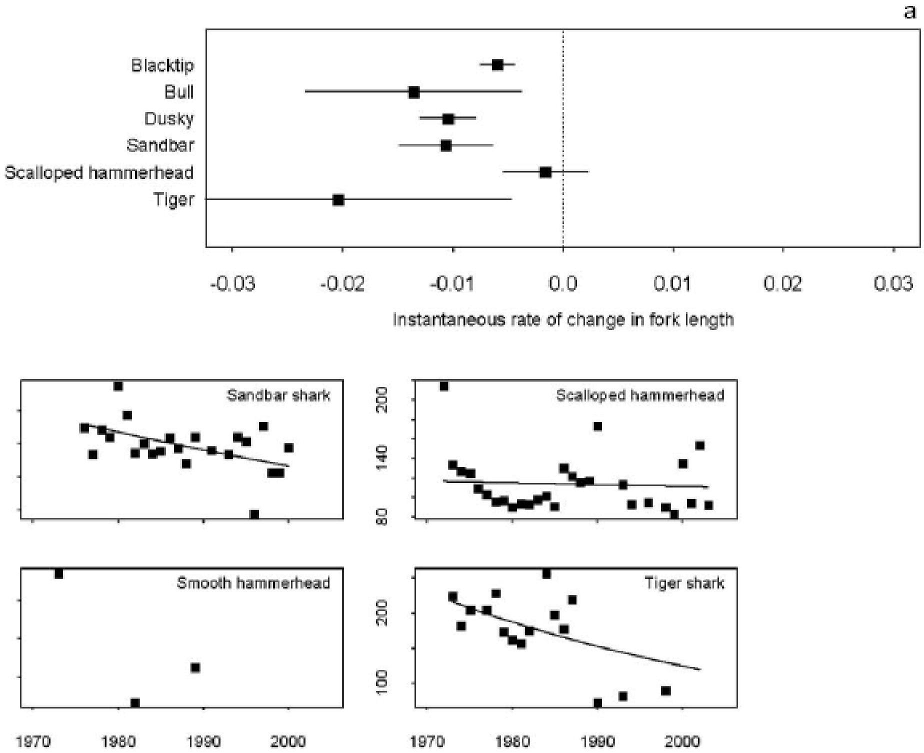
CPUE of *S. lewini* in the artisanal longline fishery in the Gulf of Tehuantepec, Mexico



Source: INP (2006), Figure 12b.

FIGURE 15

Changes in fork length, scalloped hammerhead, North Carolina shark survey

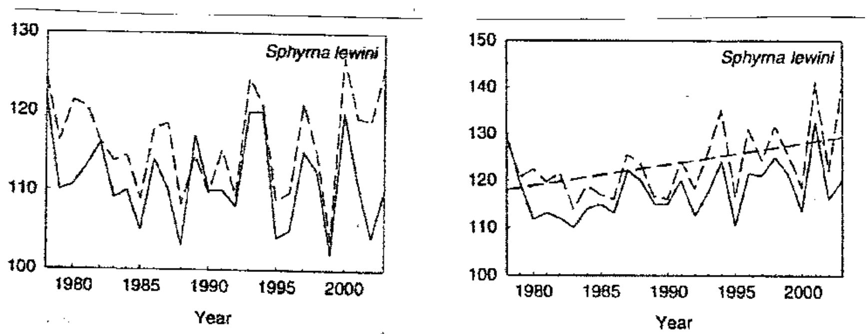


Note: In lower figure, y-axis is fork length.

Source: Myers *et al*. (2007), supplementary material, Figure S3.

FIGURE 16

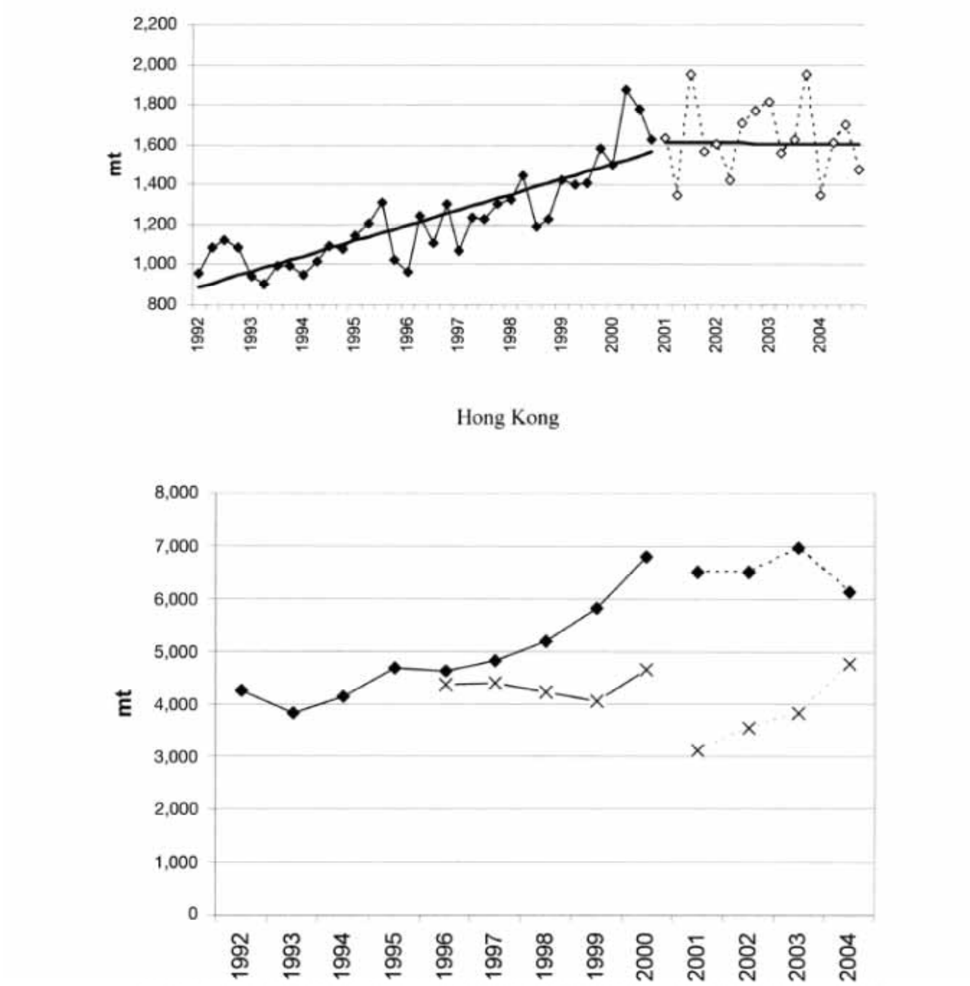
Annual median (solid line) and mean (dashed line) sizes (precaudal length) of scalloped hammerhead caught in the KwaZulu-Natal Beach Protection Programme, 1978–2003



Notes: Left panel: females; right panel: males. Straight line fit to male data indicates a significant regression.

FIGURE 17

Imports of shark fins to China, Hong Kong SAR and mainland China



Notes: Upper figure: quarterly imports to China, Hong Kong SAR (a change in statistical reporting means values before and after 2001 are not strictly comparable). Lower figure: annual imports to China, Hong Kong SAR (solid symbols) and mainland China (×).

Source: Clarke *et al*. (2007).

FAO Expert Advisory Panel assessment report: porbeagle shark - CoP16 Proposal 44 -

Species:

Lamna nasus – porbeagle shark.

Proposal:

Inclusion of *Lamna nasus* in Appendix II in accordance with Article II 2(a).

Basis for proposal:

The proposal states that the regulation of international trade in the species is necessary to avoid it becoming eligible for inclusion in Appendix I in the near future (consistent with Annex 2a A), and that regulation of international trade in the species is required to ensure that the harvest of specimens from the wild is not reducing the wild population to a level at which its survival might be threatened by continued harvesting or other influences (consistent with Annex 2a B). According to the proposal, the North and Southwest Atlantic and Mediterranean stocks meet the decline criteria for a low-productivity species while other Southern Hemisphere stocks are likely to experience similar decreases unless international trade regulations are put in place. The proposal further states that unsustainable target and bycatch fisheries are driven largely or partly by international trade demand for the high value of porbeagle meat and, to a lesser extent, by the international demand for fins.

ASSESSMENT SUMMARY

CITES biological listing criteria

The majority of Panel members considered that the species as a whole meets the decline criteria for Appendix II.

When evaluated on a population-by-population basis, the historically large porbeagle populations in the North Atlantic (Northeast and Northwest) and the Mediterranean Sea were considered to meet the Appendix II decline criterion.

Assessments for the Southwest Atlantic region indicated substantial declines, but the results were too uncertain to determine whether porbeagle in this region meets the decline criterion for Appendix II. The status elsewhere in the Southern Hemisphere was considered to be above the Appendix II decline thresholds.

The new information on distribution in the Southern Hemisphere was considered by some Panel members to indicate that the porbeagle shark has a wider distribution in the Southern Hemisphere than previously thought and that this also indicated a higher abundance. In the view of these Panel members, this brings into question the conclusion of the 2009 Panel that the species globally meets the decline criteria for Appendix II. Other members of the Panel were of the opinion that the new study did not provide information on population size in the Southern Hemisphere or the relative abundance of the Northern and Southern Hemisphere populations and that, therefore, the information did not change the conclusion of the 2009 Panel.

Comments on technical aspects of the proposal

Biology and ecology: The Panel agreed that the porbeagle shark has low productivity. Life-history characteristics such as low fecundity, slow growth and late maturation make the species particularly vulnerable to overexploitation. Such vulnerability factors are addressed in the decline criterion threshold for a low-productivity species.

Trade: Although porbeagle products are traded internationally, the actual proportion of the catches in international trade remains unknown owing to potentially substantial under-reporting and the lack of widely adopted specific customs codes for the species. These observations, in conjunction with the high value of products from the species (particularly its meat) in domestic and international markets, constitute a risk to the conservation of the species.

Fisheries management: High levels of unreported catch represent a significant potential risk factor as this will constrain accurate assessments of stock status, and subsequent management actions. The existence of rebuilding plans in Canada and the United States of America represent an important mitigating factor for the Northwest Atlantic population. Catches in the high seas areas of the North Atlantic may undermine these efforts if they are not strictly regulated. The recently adopted European Commission (EC) Regulations prohibiting fishing for porbeagle shark in waters of the European Union (Member Organization) and also prohibiting fishing vessels flagged to the European Union (Member Organization) operating in all waters to fish for, retain on board, transship or land porbeagle sharks is expected to mitigate to some extent the risk to the Northeast Atlantic population, and also to other populations affected by the fleet of the European Union (Member Organization). The Appendix III listing recently implemented by some countries of the European Union (Member Organization), which came into effect on 25 September 2012, is also likely to have a positive impact on improving information on the catches that enter international trade.

Several RFMOs have adopted regulations related to shark finning. However, finning regulations are unlikely to have much impact for porbeagle, given that the meat appears to be the most highly valued porbeagle product.

Likely effectiveness of a CITES listing for the conservation of the species: The 2012 Expert Panel and FAO (2010) noted that, if properly implemented, a CITES Appendix II listing would be expected to result in better monitoring and reporting of catches entering international trade from all porbeagle populations and subpopulations. Improved catch monitoring should enable new or enhanced assessments of stock status and the subsequent adoption of management measures that ensure the sustainability of harvests. Harvests from international waters would fall under the IFS provisions of the Convention. These would require catch documentation to the species level for specimens entering the jurisdiction of a State from international waters, along with an NDF indicating that the harvest was sustainable.

Considering the measures in place in the European Union (Member Organization) and North America to control harvest and to rebuild stocks, the listing would mainly affect the meat trade from countries in the Southern Hemisphere to the European Union (Member Organization), and the shark fin trade to China and other Asian countries. Listing in CITES Appendix II would probably strengthen current efforts to ensure that harvesting for trade is commensurate with the Canadian and United States rebuilding plans for the Northwest Atlantic stock.

The Panel also noted that the difficulty of identifying porbeagle products in trade and formulating NDFs might limit the effectiveness of a CITES listing. Species-specific assessments that could provide a basis for NDFs are lacking in the Southern Hemisphere, and requirements for additional information will create a burden that may need to be addressed through capacity building, particularly in developing countries. However, this is not unique to a potential CITES listing for porbeagle; it applies in general to all new management measures and regulations to utilize both marine and terrestrial species sustainably.

DETAILED PANEL ASSESSMENT

1. Scientific assessment in accordance with CITES biological listing criteria

1.1 Biological aspects

The information on biological aspects of the porbeagle shark summarized below in sections 1.1.1 and 1.1.2 are reproduced from the previous FAO Expert Panel reports on the species (FAO, 2007, 2010).

1.1.1 Population assessed

Porbeagle, *Lamna nasus* (Bonnaterre 1788), is distributed throughout the North Atlantic Ocean and in a broad circumglobal band in the Southern Hemisphere. Porbeagles generally occur in the Northwest and Northeast Atlantic Ocean. Tagging studies indicate that populations in the Northwest and Northeast Atlantic are distinct (COSEWIC, 2004), although occasional movements between the two areas have been observed (ICES, 2006b; proposal). The Northwest Atlantic population migrates seasonally between southern Newfoundland/the southern Gulf of St. Lawrence, and Massachusetts (COSEWIC, 2004; Campana *et al*., 2010). A single stock is considered to exist in the Northeast Atlantic (ICES, 2006a, 2011). Stock boundaries in the Southern Hemisphere are unclear. The SCRS (2009) suggests that a stock in the Southwest Atlantic could also include the Southeast Pacific and that a stock in the Southeast Atlantic could also include the Southwest Indian Ocean, but insufficient data are available to confirm these hypotheses. Semba, Yokawa and Matsunaga (2012) suggest that there may be a single widely distributed Southern Hemisphere stock.

1.1.2 Productivity level

Biological information indicates that the species falls into the category of "low" productivity (Campana *et al*., 2001, 2008; Natanson, Mello and Campana, 2002; FAO, 2007; Table 1). Age determination has been validated up to at least 26 years but ages may be underestimated in older fish (Campana, Natanson and Myklevoll, 2002; Francis, Campana and Jones, 2007). Fecundity in porbeagle is very low at an average of 4 pups per female, with females giving birth annually (Campana, Natanson and Myklevoll, 2002; Jensen *et al*., 2002; Francis and Stevens, 2000). There is no relationship between fecundity and age (Jensen *et al*., 2002). The intrinsic rate of increase of the population was estimated to be between 0.026 and 0.07 for the Southwest Pacific and Northwest Atlantic, respectively. Porbeagle shark off New Zealand may be less productive than stocks in the North Atlantic Ocean. A recent study estimated age at maturity at 8–11 years for males and 15–18 years for females, while longevity may be about 65 years (Francis, Campana and Jones, 2007). In comparison with the Northwest Atlantic, the estimated age at maturity is 7–8 years for males and 13–14 years for females, while longevity may be about 25 years (Campana *et al*., 2002; Gibson and Campana, 2005; Cassoff, Campana and Myklevoll, 2007).

1.1.3 Anthropogenic sources of mortality

The main sources of anthropogenic mortality of porbeagle are the exploitation in direct fisheries and the bycatch, mainly in longline fisheries. As porbeagle is a high-value species for meat and fins, it is unlikely that large numbers of individuals caught as bycatch are discarded. The global reported catches of porbeagle shark are shown in Figure 1.

In the Northeast Atlantic, the species has been fished by many European countries, mainly by Denmark, France, Norway, Faroe Islands and Spain (Figure 2). According to the International Council for the Exploration of the Sea (ICES, 2010), the target fishery for porbeagles before the Second World War was mainly a Norwegian longline fishery in the North Sea, starting in 1926 and landing about 500 tonnes annually in the first few years. After a peak in 1933 (about 3 800 tonnes) the fishery declined. After the war, the target fishery resumed with Norwegian, Faroese and Danish vessels involved. In the 1950s, the catches began to decline to less than 2 000 tonnes annually, and in 1961 a fleet of Norwegian longliners extended their fishing for porbeagle to Northwest Atlantic waters. In the 1970s, other countries started to report landings of the species, including Faroe Islands, France, the United Kingdom of Great Britain and Northern Ireland, Iceland, Germany and Sweden. French landings were mainly provided by a longline targeted fishery that landed relatively large quantities from the early 1970s, with a decline in landings and number of boats in the mid-1980s. After this, catches fluctuated between about 200 and 500 tonnes, with a peak of 640–840 tonnes between 1993 and 1995 (Jung *et al*., 2009).

Until recently, porbeagle sharks were landed by several European countries, principally France and, to a lesser extent, the United Kingdom of Great Britain and Northern Ireland, Faroe Islands, Norway and Spain. Fisheries have generally been seasonal, and many operations landed porbeagle opportunistically and sporadically rather than through directed fisheries (ICES, 2010). Landings from Spain are thought to be taken mainly in longline fisheries targeting swordfish and tuna. The only regular, directed target fishery that existed until recently was the French longline fishery. Since 2010, fishing or landing porbeagle by countries of the European Union (Member Organization) has been forbidden, and the main legal fisheries are from Norway and Faroe Islands.

In the Northwest Atlantic, the targeted fishery for porbeagle stated in 1961, when shark longline fleets from Norway and Faroe Islands moved from the Northeast Atlantic to the coast of New England and Newfoundland (proposal). Catches increased rapidly, reaching more than 9 000 tonnes in 1964 (Figure 3). The fishery collapsed after six years, landing less than 1 000 tonnes in 1970. Foreign fleets were excluded from the fishery in the early 1990s and, since 1995 all landings have been from Canadian and United States fisheries, including a Canadian target fishery. Landings and catch rates continued to decline in response to stock depletion, leading to the adoption of a recovery plan with reduced annual quotas after 2002. Since 2006, reported annual landings have been below 200 tonnes (Figure 3).

In addition, an unknown quantity of porbeagle is taken as bycatch on the high seas at high latitudes in the North Atlantic by tuna longline fisheries targeting bluefin tuna (Nakano and Homma, 1997), especially from Taiwan Province of China, the Republic of Korea and Japan. According to estimates made by the ICCAT SCRS (2009), the potential porbeagle catch for non-reporting longline fleets operating in the Northwest Atlantic high seas areas was about 40 tonnes per year from 2000 to 2007, which is minor compared with catches from target fisheries in this region. The level of high seas catch by tuna longliners may further decrease owing to reductions in the total allowable catch (TAC) for Atlantic bluefin tuna (ICCAT, 2012).

In the Southern Hemisphere, porbeagle is mainly caught as bycatch in longline fisheries for tuna and swordfish, including by Uruguay, New Zealand, Spain and Japan. The species is also incidentally caught in mid-water and bottom trawling in New Zealand, and in demersal longline and trawling fisheries for Patagonian toothfish (including by Argentina) (proposal). For New Zealand fleets, reported catches declined from 152–301 tonnes in the period 1997–2003 to a low of 54 tonnes in 2005–06. A TAC of 249 tonnes was adopted in 2004, but reported catches have subsequently been well below this level (New Zealand Ministry for Primary Industries, 2012). Reported catches in the Southern Hemisphere reached a peak of about 250 tonnes in 1999 (mostly from New Zealand fleets) and then declined to a low of 64 tonnes in 2010 (Figure 4). However, the volume of porbeagle catches for non-reporting fleets may be substantial, being estimated at about 200 tonnes per year from 2000 to 2007 (SCRS, 2009).

1.1.4 Population status and trends

The sections below have been updated from the previous FAO Ad Hoc Panel reports (FAO, 2007, 2010) with any new information presented in the proposal.

Population size

Available estimates for the Northwest Atlantic population are 11 000–14 000 mature females, 33 000–38 000 mature individuals, and 196 000–207 000 total individuals (DFO, 2005; FAO, 2007; SCRS, 2009; Campana *et al*., 2010). For the Northeast Atlantic, the population size was estimated between 127 000 and 204 000 individuals (SCRS, 2009). No information on population size is available from other areas where the species occurs.

Area of distribution

The extent of occurrence in Canada is estimated to be 1.2 million km^2, while the area of occupancy in Canada from recent catch locations is estimated to be 830 000 km^2; the range is not known to have changed since the fishery began in 1961 (COSEWIC, 2004). The area of occupancy and the extent of occurrence for the entire Northwest Atlantic would be greater than these values. There is no evidence that local depletion exists in this area for porbeagle because tagging data suggest this species is highly migratory. The area of occurrence in Norwegian waters is estimated at 395 000 km^2 (FAO, 2010). The area of occurrence in the Northeast Atlantic would be considerably larger than this. The area of distribution is potentially even larger in the Southern Hemisphere (Semba, Yokawa and Matsunaga, 2012; Figure 5).

Population trend

Because this species occurs in several widely separated areas, and in distinct populations, no single abundance index can be applied to the species as a whole. Assessments of the decline in abundance of the species can only be conducted using abundance indices from as many parts of the species' distribution as possible.

With the exception of updated catch data, the proposal does not present any new information concerning the stock status and trends since it was last submitted to CITES CoP15. Therefore, the information reviewed below is mostly based on the previous evaluations of the porbeagle listing proposals (FAO, 2007, 2010), with updates from new information for the Southern Hemisphere. Trend information is summarized in Table 2.

Northeast Atlantic

The first assessment of the porbeagle stock in the Northeast Atlantic was conducted by ICES/ICCAT in 2009 (SCRS, 2009). This assessment has not been updated since then. According to ICES (2011), no new information that could alter the conclusions about the status of the stock has been presented in more recent years. The results of the 2009 assessment, as summarized by FAO (2010), are reproduced below.

Two assessment models were used by the SCRS (2009) to assess the status of the Northeast Atlantic stock: a surplus production model and an age-structured production model. Both models used catch data from 1926 and CPUE data from Spanish (1981–2007) and French (1972–2008) longline fleets. Results from satisfactory model runs of the surplus production model (runs based on the longest time series and based on realistic values for the unexploited population size) estimated that the current population size is between 15 and 39 percent of the unexploited population size (Figure 6). Results from the age-structured production model estimated that the current stock biomass is 6 percent in biomass and 7 percent in numbers of the unexploited population size (Figure 7). Current fishing mortality is estimated at between 2.3 and 3.5 of the fishing mortality that would maximize yield in the long run. The SCRS (2009) concluded that all the models that used biologically plausible assumptions about unfished biomass inferred that the population is currently depleted. However, the results of both assessment models are considered highly uncertain, given that the majority of the fishery removals occurred before data were available to estimate abundance trends (SCRS, 2009).

Forward projections of the stock based on the surplus production model indicated that the TAC at that time of 436 tonnes was likely to cause the population to remain fairly stable at a low biomass level. Rebuilding of the stock could take several decades under lower fishing mortality rates. In the absence of better information to assess the status of the stock, the management recommendation of ICES was to prohibit the target fishing for porbeagle, to limit the bycatch and to prohibit landings (SCRS, 2009), which was adopted by the EC in 2010 (ICES, 2011).

Catch per unit of effort data from the French longliners decreased by one-third between the early 1970s and early 1980s and since then has oscillated without a trend. The Spanish CPUE has also oscillated without a trend since the mid-1980s (Figure 8; SCRS, 2009, ICES, 2011). As noted above, both CPUE time series were used in the stock assessment models for the Northeast Atlantic stock.

Catch data were also used in the proposal to demonstrate a decline in the Northeast Atlantic stock, as done in the previous proposal submissions (FAO, 2007, 2010). In the Northeast Atlantic, the species has been fished by many European countries, mainly by Norway, Denmark, France, Faroe Islands and Spain. Total landings in the Northeast Atlantic declined from an average of 2 953 tonnes in 1933–37 to 388 tonnes in 2004–08 (Figure 2). The use of more recent catch data as an indicator of stock trends is problematic because, since 2010, EC Regulations have prohibited fishing for porbeagle in waters of the European Union (Member Organization) and vessels of the European Union (Member Organization) from fishing the species in international waters (see management section). As a result, only 20 tonnes were reported in 2010, representing mainly porbeagle caught incidentally by gillnet in Norway and by pelagic trawlers in France (ICES, 2011). Landings of the Norwegian and Danish fleets in 2004–08 were about 1 percent of their historical peaks in the 1930s and 1950s, respectively (Table 2). No landings were reported by Denmark in 2010, while Norway reported 12 tonnes. French longliners started targeting porbeagle in the 1970s. Catches peaked in 1979 at 1 092 tonnes and were about 300 tonnes in 2009. The species is also caught opportunistically as bycatch in Spanish longliners targeting swordfish and sharks in the Atlantic (proposal). Reported catches have oscillated without a trend since the early 1970s, being always less than 70 tonnes/year. Since 2009, no catches of porbeagle have been reported by Spain (ICES, 2011). As stated by FAO (2007), landings data do not provide an accurate index of abundance because changes in landings may be influenced by market conditions and management measures rather than abundance of the species.

Mediterranean

Information about population trends in the Mediterranean is the same as presented in the proposal submitted to CoP15 and evaluated by the previous FAO Ad Hoc Panel (FAO, 2010). Only catch date have been updated based on FAO FishStat. The analysis conducted by FAO (2010) is reproduced below.

The proposal compiled different sources of information suggesting the disappearance of porbeagle in the Mediterranean. It is not known whether the porbeagles in the Mediterranean are part of a separate stock from the Northeast Atlantic. Declines of more than 99 percent in catches of lamnid sharks (including porbeagle) in tuna traps in the Ligurian Sea were estimated between 1950 and 2006 (Figure 9; Ferretti *et al.*, 2008). Ferretti *et al.* (2008) also estimated declines of more than 98 percent in the CPUE of longline fisheries in the Ionian Sea between 1978 and 1999. However, the authors noted that the CPUE at the beginning of the time series was already very low (of the order 0.2 sharks/1 000 hooks).

Reported landings in FAO FishStat have been below 5 tonnes per year since 1970, when catches of the species were first reported to FAO.

Northwest Atlantic

The most recent assessment of the porbeagle stock in the Northwest Atlantic was conducted by the SCRS (2009). This assessment has not been updated since then. A summary of the results of the stock assessment, as described in FAO (2010), is reported below.

Two assessment models were used by the SCRS (2009) to estimate the status of porbeagle shark in the Northwest Atlantic: a surplus production model and an age-structured model. Results from the surplus production model applied to data through 2009 estimated that the current stock biomass is about 32 percent of the stock biomass in 1961 (Figure 10). According to the age-structured model, the current population size is about 22–27 percent of its size in 1961. The number of mature females in the population is estimated at 12–16 percent of the estimated number in 1961 (Campana *et al.*, 2010). The models indicate that the reduced quotas since 2002 have stopped the decline in the populations, which has stabilized in recent years. Current population size is about 95–103 percent of its size in

2001. The recovery to B_{MSY} levels is likely to occur in about 20 years with no fishing and not until 2041 with the current catch levels (direct and bycatch) (Campana *et al*., 2010).

Catch data are also used in the proposal to infer the extent of decline in the Northwest Atlantic stock. Landings in the Northwest Atlantic fishery were high in the early 1960s, declined to low levels in the 1970s and 1980s, increased in the early 1990s and declined to low values in the early 2000s (Figure 3; Gibson and Campana, 2005). Average catches from 2004–08 were about 180 tonnes per year, representing 4 percent of the historical maximum levels (Table 2). According to FAO data, catches have been below 100 tonnes per year since then. The low reported catches in recent years have been due mostly to strict quota regulations in place in Canada and the United States of America. The amount of non-reported bycatch in longline fisheries operating in the Northwest Atlantic is not known precisely, but estimated to be less than 40 tonnes per year from 2004 to 2007 (SCRS, 2009). These estimates were used in the stock assessment conducted by the SCRS (2009), reported above.

Southern Hemisphere

The only new information presented in the proposal, as compared with the previous submission (FAO, 2010), is the updated catch data. However, a new study by Semba, Yokawa and Matsunaga (2012) and Forselledo Caldera (2012), and new information from the New Zealand Ministry for Primary Industries (2012) is also reported below.

Catch per unit of effort data of porbeagle caught as bycatch in the Uruguayan pelagic longline fleet shows a declining trend from 1982 to 2008 (Figure 11). According to these data, the relative abundance of porbeagle shark after 1995 was approximately 20 percent of the captures from previous years. However, as noted by FAO (2010), the changes in the Uruguayan CPUE time series occurred too quickly to be explained solely on the basis of abundance changes. Other factors related to changes in environmental conditions and fishing strategies could have also played a role (SCRS, 2009). The Uruguayan CPUE time series was used by the SCRS (2009) to assess the status of the porbeagle stock in the Southwest Atlantic using a surplus production models. Because of suspected high levels of unreported catches from all tuna longline fleets operating in the area, the model included estimates of the potential total catches based on pelagic longline fishing effort and the ratios of porbeagle to other species in the pelagic longline catch. Results indicated that the current stock biomass is about 18–39 percent of the unexploited stock size, depending on the assumption made about unreported catches (Figure 12). The Uruguayan CPUE data was also used by the SCRS (2009) to assess the stock using a catch-free age-structured production model. The model estimated that the current spawning stock biomass is 18 percent of the unexploited level and 54 percent of the biomass in 1982 (Figure 13). The SCRS (2009) concluded that, despite the convergence of the methods in showing potential declines in porbeagle abundance in the Southwest Atlantic, data are too limited to provide a robust indication on the status of the stock.

Other CPUE data from the Southern Hemisphere are from bycatch fisheries, including in Japanese and New Zealand longline fisheries for tuna. Porbeagle is one of the main pelagic shark species caught as bycatch in the Japanese longline fishery targeting southern bluefin tuna (Matsunaga, 2010). In the South Atlantic, porbeagle is caught as bycatch in the Japanese longline fishery targeting southern bluefin tuna and bigeye tuna (Semba and Yokawa, 2012). Standardized CPUE data from these fleets showed no trend from 1992 to 2007 and from 1994 to 2010, respectively. A more recent study (Semba, Yokawa and Matsunaga, 2012) analysed historical Japanese data from commercial fisheries and an exploratory survey using tuna longlines as well as past high seas drift-net exploratory surveys in the 1980s. The results indicated that porbeagle sharks are widely distributed in the high seas areas of the Southern Hemisphere at densities comparable with those in EEZs (Figure 5). This information is confirmed by Forselledo (2012) for the Southwest Atlantic. Semba, Yokawa and Matsunaga (2012) conclude that their results indicate there may be a single, widely distributed Southern Hemisphere stock. The standardized CPUEs of the high seas drift-net exploratory research and commercial longline (both logbook and observer) exhibited little trend over the period between 1982–1990 and 1992–2007. The standardized CPUE based on observer data increased in 2007–2010, and that based on logbook data increased in 2007–2011 (Figure 5). These abundance indices have the greatest temporal and spatial extent in the Southern Hemisphere.

Unstandardized CPUE indices from the New Zealand tuna longline fleet between 1993 and 2010 suggested an overall declining trend in this period (particularly for the most reliable of these indices), although with considerable variability (Figure 14). However, it is noted that trends for some of the indices may not reflect the stock abundance because of low observer coverage and changes in fishing operations (Griggs and Baird in press, cited in New Zealand Ministry for Primary Industries, 2012). Porbeagle CPUE was higher in the southern areas than it was in northern regions, but porbeagle CPUE has been very low for the past nine years in the south, while there has been a recent increase in the north (New Zealand Ministry for Primary Industries, 2012). The New Zealand Ministry for Primary Industries (2012) believes that the commercial CPUE analyses undertaken to date have not generated reliable estimates of porbeagle abundance, and that overfishing of the stock is occurring.

Porbeagle bycatch in the Argentinean trawl fisheries targeting fish on the southern Patagonian shelf for surimi products has been estimated at 20–70 tonnes in the period 2003–06 (Waessle, 2007). Waessle and Cortés (2011) provided updated information on the conventional trawl fleet, where catches were lower than those observed in Waessle (2007).

According to FAO FishStat data, reported landings in New Zealand reached a peak of 246 tonnes in 1999 and declined by 83 percent to a low of 42 tonnes in 2008, with catches slightly above this figure in recent years (Figure 4). Catches from Spain (prior to the adoption of the zero quota of the European Union [Member Organization] for porbeagle) and Uruguay, the two other countries reporting significant catches of porbeagle, do not show a clear trend (Figure 4).

Other indices

The average length of individuals taken in Northwest Atlantic fisheries declined from more than 200 cm in 1960–1980, to 140–150 cm in 1999–2000 (Campana *et al*., 2001; Figure 15).

1.2 Assessment relative to quantitative criteria

1.2.1 Small population

The estimate of the current total population size for the Northwest Atlantic is 11 000–14 000 mature females, and 196 000–207 000 total individuals. For the Northeast Atlantic, the total population size is 127 000–204 000 individuals. The composite population size in the North Atlantic would be therefore at least 323 000 individuals. The total population size worldwide is likely to be well above this. Therefore, it is unlikely that the species can be characterized as having a small population size.

1.2.2 Restricted distribution

No guidelines for restricted area of distribution are provided in the CITES Criteria, which indicate that thresholds should be taxon-specific (Conf. Res. 9.24 Rev. CoP15). FAO (2001) recommended that historical extent of decline in the area of distribution would be a better measure of extinction risk than the absolute value of distributional area, but that if no other suitable information is available and the absolute area of distribution has to be used for an exploited fish population, analyses should be on a case-by-case basis as no numeric guideline is universally applicable.

The total area of distribution for the species would be substantially greater than estimates for Canada, where the extent of occurrence is 1.2 million km^2, and the area of occupancy is 830 000 km^2. For the Northeast Atlantic, the area of occurrence is at least 395 000 km^2. The area of occurrence may be even greater in the Southern Hemisphere. Therefore, the species is not characterized by a “restricted” distribution.

1.2.3 Decline

Under the CITES criteria for commercially exploited aquatic species (Res. Conf. 9.24 Rev. CoP15), a decline to 15–20 percent of the historical baseline for a low-productivity species might justify consideration for an Appendix I listing. For listing on Appendix II, being “near” this level might justify consideration for a listing, which for a low-productivity species would be 20–30 percent of the historical level (15–20 percent + 5–10 percent).

No global population decline index is available for comparison with the guidelines. Because the proposal does not include new information on population trends since the last submission to CITES, the analysis and conclusions by FAO (2010) are generally still considered to be valid, and are reported below.

For the Northwest Atlantic population, the current mature female population estimated with an age-structured model is 12–16 percent of the historical baseline prior to major fisheries (1961), while the total population is 22–27 percent of that historical baseline. Results from a surplus production model applied to the same time series of data estimated that current stock biomass is about 32 percent of the stock biomass in 1961, which is only slightly above the decline threshold of 30 percent for an Appendix II listing. These results indicate that the population in the Northwest Atlantic meets the criterion for Appendix II, as concluded in the previous Panel report (FAO, 2007). The population is under a conservative harvesting regime in Canada and the United States of America, which is expected to allow the recovery of the stock. However, recovery to target levels will take decades owing to the low productivity of the species. As noted by the SCRS (2009), there is probably unreported catch in the high seas within the stock area, and increased effort in these areas could compromise stock recovery efforts.

For the Northeast Atlantic, assessment against the decline criterion is more difficult owing to the lack of long-term indices of abundance. The only CPUE data available are from longline fisheries from 1972 to 2008, well after the historical peak in landings in the 1930s. Stock assessment results based on the available catch and CPUE data indicate that the current population size is about 15–39 percent of the unexploited population size, according to one modelling approach, and 6 percent in biomass and 7 percent in numbers of the unexploited population size according to another modelling approach. Despite the uncertainties of the results, these levels of decline put the Northeast Atlantic stock generally within the decline threshold for an Appendix II listing.

In the Mediterranean, a decline of more than 99 percent in catches in tuna traps was estimated between 1950 and 2006. Although catches are not generally an appropriate measure of abundance trends, catch data from the fixed tuna traps were considered a relatively reliable source of information about abundance trends. Considering in addition the estimated decline of more than 98 percent in longline CPUE between 1978 and 1999 and other anecdotal information about the disappearance of the species, the Panel concluded that the decline in porbeagle abundance in the Mediterranean meets the criterion for an Appendix II listing.

For the Southern Hemisphere, less information is available. A stock assessment based on CPUE data from the Uruguayan fleet and on reconstructed catches in the Southwest Atlantic estimated that the current stock biomass is at about 18 percent and 39 percent of the unexploited stock size. This level of decline would be generally within the decline criterion for an Appendix II listing. However, the results were considered highly uncertain because of data limitations (FAO, 2010). The current Panel and FAO (2010) concluded that other stocks in the Southern Hemisphere are probably not lightly fished but may be above the Appendix II decline criteria threshold.

In summary, FAO (2010) concluded that the available evidence indicates that the stocks of porbeagle in the North Atlantic (Northwest and Northeast stocks) and Mediterranean Sea meet the decline criteria for inclusion in CITES Appendix II. The status of stocks in the Southern Hemisphere is more uncertain but overall it was considered that these stocks are likely to be above the decline threshold for an Appendix II listing.

The 2009 Panel took notice of the wording of Resolution Conf. 9.24 (Rev. CoP15) indicating that Parties had resolved to adopt measures that are proportionate to the anticipated risks to the species when considering proposals to amend the Appendices (FAO, 2010). Using the same rationale, the majority of the 2012 Panel considered that the species as a whole met the decline criteria for Appendix II.

Were trends due to natural fluctuations?

There is no evidence that the observed trends were due to natural fluctuations.

2. Comments on technical aspects in relation to trade, management and implementation issues

2.1 Trade aspects

Porbeagle shark products, particularly the meat and, to a lesser extent, the fins, are highly valued in markets and accordingly are in high demand (proposal; Rose, 1996; Fowler, Raymakers and Grimm, 2004; FAO, 2007, 2010). Prior to 2010, all global trade in porbeagle products was reported under the general customs commodity codes for sharks and could not be differentiated from other species of sharks. Therefore, it was impossible to assess the volume of catches supplying domestic and international trade. In 2010, the European Union (Member Organization) introduced new species-specific customs codes for fresh and frozen porbeagle products (excluding shark fins). At the same time, a zero quota was established for waters and fleets of the European Union (Member Organization), meaning that, since 2010, the market demand in markets of the European Union (Member Organization), where porbeagle meat has a high value (proposal; Vannucinni, 1999; FAO, 2007), should theoretically be met by imports.

Trade data of the European Union (Member Organization) for 2010 and 2011, presented in the proposal, indicate that 50.5 tonnes of porbeagle meat was imported to the European Union (Member Organization) and 141.3 tonnes were exported from it in the two-year period. The main suppliers were South Africa, Japan, Morocco, Norway and Faroe Islands. The main destination of exports from the European Union (Member Organization) was Morocco, which imported 137 tonnes in the two-year period. The proposal notes some inconsistencies in the trade data, including, for example, that exports occurred when a zero quota was in place for fisheries of the European Union (Member Organization). In fact, more meat was exported than imported over the period. As discussed by the proponents, the exports may be explained by stockpiles of catches landed and frozen before 2010, or by re-exports. On the other hand, South Africa, the main exporter in 2011, does not have a directed fishery for porbeagle and has never reported landings of porbeagle to FAO. As discussed in the proposal, the high quantities exported to the European Union (Member Organization) are likely to be derived from foreign vessels fishing in the South Atlantic and landing in South African ports. Until species-specific customs codes are widely applied, it will be very difficult to determine the exact origin of the species products in trade.

Nonetheless, the adopted customs code in the European Union (Member Organization) sheds some light on the importance of international trade of the species. For example, the comparison between imports into the European Union (Member Organization) and reported global catches of porbeagle gives a rough idea of the relative importance of international trade in the species. On average, 25 tonnes of porbeagle meat was imported into the European Union (Member Organization) per year between 2010 and 2011, representing approximately 50 tonnes of live weight per year (after applying a conversion factor of 2.0 for sharks gutted, with the head off; FAO, 1997). This figure represents approximately 20 percent of the reported global catches of 253 tonnes in 2010, the last year of available catch data in the FAO FishStat database.

The recent trade data of the European Union (Member Organization) also confirmed the existence of exports from Japan to the European Union (Member Organization), documented in earlier studies (Vannuccini, 1999). However, there were no records of the European Union (Member Organization) importing porbeagle from Canada, or of the European Union (Member Organization) exporting porbeagle to the United States of America, also reported in earlier studies (Vannuccini, 1999; FAO, 2010).

Trade within the European Union (Member Organization) is also well documented (proposal; FAO, 2007, 2010), but this does not qualify as international trade. According to data reported in the proposal, Italy and Spain were the main destinations for trade of fresh and frozen porbeagle within the European Union (Member Organization), and Portugal and Spain were the main suppliers of the traded products.

Besides the meat, fins of porbeagle are also highly valued. Porbeagle is among the preferred species for fins in Indonesia (Vannuccini, 1999). The species is found in the global fin market (Shivji *et al*., 2002; cited in the proposal). In this regard, FAO (2007) noted that "porbeagle fins are found in markets in China, Hong Kong Special Administrative Region (SAR), and internationally (proposal; Shivji *et al*. 2002), but are apparently not one of the common species in the Hong Kong dried fin market, possibly because fins in that market primarily come from areas other than those where porbeagle is most abundant (northwest and northeast Atlantic) (Table 2 in Clarke *et al*. 2006)". Other products probably in trade cited in the proposal are hides, liver oil and cartilage, but the actual traded volumes are unknown.

2.2 Fisheries management aspects

Management measures are in place in several countries having direct fisheries for porbeagle or landing porbeagle incidentally caught in other fisheries. Since 2010, EC Regulations have prohibited fishing for porbeagle shark in waters of the European Union (Member Organization) and also prohibited fishing vessels flagged to the European Union (Member Organization), operating in all waters, from fishing for, retaining on board, transshipping or landing porbeagle sharks (Council Regulations (EU) Nos. 23/2010, 57/2011 and 43/2012). Porbeagle is listed as "vulnerable" in the Norwegian Red List for Species (Kålås *et al*., 2010). In 2007, Norway banned all direct fisheries for porbeagle, and from 2007 to 2011 individuals taken as bycatch had to be landed and sold. Since 2011, live specimens must be released, whereas dead specimens can be landed and sold (proposal). The species is under a rebuilding plan in Canada and the United States of America, where catch quotas have been reduced to levels that will support the population recovery (SCRS, 2009). The current TAC in Canada is 185 tonnes, with 135 tonnes allocated to commercial fisheries and 50 tonnes reserved to account for bycatch of porbeagle in other fisheries (DFO, 2012). In 2008, the United States of America adopted a TAC of 11.3 tonnes (dressed weight) for all fisheries, of which 1.7 tonnes was allocated as a commercial quota. The commercial quota was further reduced to 0.7 tonnes in 2012 (Atlantic Shark Season Final Rule 77 FR 3393, issued on 24 January 2012). Once the quota is exceeded, the fishery is closed. New Zealand has included porbeagle under a quota management system since 2004 (FAO, 2010). The current TAC is 249 tonnes, with 215 tonnes being allocated to the commercial sector (Ministry of Fisheries of New Zealand). There are no specific regulations for porbeagle in Argentina or Uruguay. In Argentina, regulations related to finning and requiring the release of all live sharks longer than 1.6 m provides some level of control of the bycatch in longline and trawl fisheries (FAO, 2010).

Measures by the European Union (Member Organization) have severely restricted the options for fishing porbeagle in European waters and the Northeast Atlantic since 2010 and have resulted in a regulated prohibition on the fishing or landing of porbeagle by vessels of the European Union (Member Organization) anywhere in the world in 2012. Vessels from outside the European Union (Member Organization) are now not permitted to fish porbeagle in waters of the European Union (Member Organization); however, they are still permitted to fish porbeagle in other international waters, including those of the Mediterranean Sea.

More recently, porbeagle has been listed on CITES Appendix III by some countries of the European Union (Member Organization) with effect from 25 September 2012. European Union (Member Organization) Wildlife Trade Regulations implementing CITES within the European Union (Member Organization) and the inclusion of the species in Annex C of Council Regulation (EC) No. 338/97 took effect on the same day. These regulations include some measures that are stricter than those imposed by CITES in regard to export, re-export and import. Specifically, any export of porbeagle from any member State of the European Union (Member Organization) needs to be accompanied by an export permit attesting both the legality of the catch and the fact that collection and/or trade of the specimens will not have a harmful effect on the conservation status of the species (essentially an NDF). Under CITES, an export permit for Appendix III-listed species only requires the authority to attest to the legality of the specimens. As fishing and landing of porbeagle by vessels of the European Union (Member Organization) is currently prohibited by regulation, any porbeagle landed in member States of the European Union (Member Organization) is assumed to have been fished by a vessel not from the European Union (Member Organization), outside of the waters of the European Union (Member Organization). With reference to import into the European Union (Member Organization), the European Union (Member Organization) requires an import notification to be completed by the importer; this is not required for imports into any other CITES Party. Re-export from a member State of the European Union (Member Organization) would therefore need to include both the certificate of origin and the import notification in the European Union (Member Organization).

Many RFMOs have adopted generic management and conservation measures for sharks (Fisher *et al*., 2012). The CCAMLR has prohibited directed fishing on shark species and requires any specimen taken accidentally to be released, as far as possible, alive. Measures related to shark finning have been adopted by ICCAT, GFCM, IATTC, IOTC, NAFO, NEAFC, SEAFO and WCPFC. In November 2011, the NEAFC agreed to ban all directed fisheries for porbeagle in its regulatory area from 3 February 2012 to 31 December 2014. The WCPFC also encourages the live release of incidentally caught sharks in fisheries for tunas and tuna-like species. There is limited information on the effectiveness of these measures and levels of compliance with them.

In the context of internationally agreed instruments, the voluntary FAO International Plan of Action for the Conservation and Management of Sharks urges all States with shark fisheries to implement conservation and management plans. According to a recent review prepared by FAO (Fischer *et al*., 2012), of the 26 top shark-fishing nations (responsible for 84 percent of total reported global catches from 2000 to 2009), 17 have adopted National Plans of Action for Conservation and Management of Sharks (NPOA-Sharks). Among them are important porbeagle range and/or fishing countries or entities, such as Australia, Argentina, Canada, member States of the European Union (Member Organization), Japan, New Zealand, the Republic of Korea, Taiwan Province of China, the United States of America and Uruguay. The level of effective implementation of the adopted plans varies among countries (Techera and Klein, 2011).

Porbeagle is also listed in various international agreements aimed at fostering international cooperation for the protection of threatened species. The species is listed in Annex II of the Barcelona Convention Protocol, in Appendix III of the Bern Convention on the Conservation of European Wildlife and Habitats, in Annex V of the OSPAR Convention for the Protection of the Marine Environment of the Northeast Atlantic, in Appendix II of the Convention on the Conservation of Migratory Sharks (CMS) and in the Annex to the CMS's Migratory Sharks Memorandum of Understanding (proposal).

2.3 Implementation issues

2.3.1 Introduction from the sea

As stated in the proposal, and also in FAO (2007, 2010), in the Northern Hemisphere, most porbeagles are harvested within EEZs. By contrast, in the Southern Hemisphere, porbeagle is known to be taken as a bycatch in the longline fisheries of Japan, the Republic of Korea and Taiwan Province of China operating on the high seas, and this is believed to represent the major portion of the catch in the Southern Hemisphere. As such, the IFS provisions of CITES would be an issue for this species for high seas longline fleets, requiring landings of these specimens to be accompanied by IFS certificates. Exactly how this certification process would be carried out is still a matter of debate within CITES.

According to the proposed amendments to Resolution Conf. 14.6 (Rev. CoP15) on IFS (CITES CoP16 Doc. 32), the IFS certificate would be required whenever any specimen of a species included in Appendix I or II is taken in the marine environment not under the jurisdiction of any State by a vessel registered in one State and is transported into that same State (in some circumstances the same would apply to a State chartering a vessel from another State and transporting the specimen to the chartering State). Whenever a specimen taken by a vessel registered in one State is transported into a different State, that transportation would be treated as any other export, requiring export and, where applicable, import permits.

- Irrespective of whether an IFS certificate or an export permit applies, a species taken in the high seas can only be authorized to be landed if the following requirements are met:

- The Scientific Authority of the State where the vessel is registered makes an NDF, ideally in consultation with other national scientific authorities or, possibly, international scientific authorities that have been involved in the assessment and management of the stock. Some level of involvement of RFMOs would be expected to occur at this stage, in areas where such organizations have been established with mandate over shark fisheries.
- The Management Authority of the same State makes a legal acquisition finding (i.e. a finding that the specimen was not obtained in contravention of the laws of that State for the protection of fauna and flora).

General issues concerning the implementation of these two requirements are discussed below. Other more complex implementation issues may arise when the fishing operation involves practices such as transshipment, onboard processing and treatment of catch taken partly from waters under national jurisdiction and partly outside on the same fishing trip (FAO, 2004). In the case of transshipments, the proposed amendments to Resolution Conf. 14.6 (Rev. CoP15) recommend that the transshipment be interpreted as a means of transportation, and the same considerations for IFS or export should apply. In both cases, the master of the vessel receiving the transshipped specimens should obtain satisfactory proof that the IFS certificate or export permit exist or will be issued before the transshipment occurs. The practical difficulties in implementing such provisions will be significant.

In matters related to listed shark species, it is imperative that RFMOs and the CITES Secretariat work closely together. The same applies to the CITES Secretariat and national scientific and management authorities.

2.3.2 Non-detriment findings

Non-detriment findings are the responsibility of the exporting country and must show that exports are non-detrimental to survival of the species; that is, that they are consistent with sustainable harvesting. Development of an NDF requires appropriate scientific capacity, biological information on the species, and an approach to demonstrating that exports are based on sustainable harvests.

For the Northwest Atlantic population, the basis for NDFs should follow the current rebuilding plans and TACs established by Canada and the United States of America based on results from stock assessments. For the Northeast Atlantic, scientific advice is available to inform NDFs. In addition, the recently adopted European Community Action Plan for the Conservation and Management of Sharks could eventually provide the management reference points needed to evaluate NDFs. For porbeagle introduced from the sea, existing RFMOs could be used to provide the basis for NDFs (FAO, 2007, 2010). Guidelines and tools are available to inform other CITES Parties on the necessary information and steps to be taken in formulating NDFs (Rosser and Haywood, 2002; Anonymous, 2008). Although this is the case, it will frequently be necessary to build capacity, particularly in developing countries, in order to put in place effective management and monitoring systems for the harvesting of porbeagle, which will be required for making meaningful NDFs and for ensuring compliance with an Appendix II listing.

2.3.3 Findings that specimens were legally obtained

Porbeagle harvests from the Northwest Atlantic population are regulated under the Canadian and the United States management plans. Exports of products based on legal harvesting under these management plans would qualify as legally obtained for CITES. In the Northeast Atlantic, recently established EC regulations prohibit porbeagle catches by fleets of the European Union (Member Organization), meaning that the demand has to be met by imports. The regulations in place provide the basis to judge whether takes were legally obtained. New Zealand, Norway and Faroe Islands have also established TACs for the species, and a maximum landing size is in place in Argentina. Regulations related to shark finning are also in place in many countries and RFMOs. Exports from these countries and areas that are in agreement with the established regulations would qualify as legally binding under CITES.

2.3.4 Identification of products in trade

FAO (2007) noted that "it would probably be difficult for a non-expert to distinguish the meat of porbeagle from that of other similar lamnoid sharks in trade such as shortfin mako." Based on experience in the China, Hong Kong SAR market, expert knowledge and experience are probably required to identify porbeagle fins in trade (Clarke *et al*., 2006), although porbeagle dorsal fins with skin on have a characteristic white rear edge (proposal). DNA techniques are not considered practical as initial screening tools although they may be useful for secondary inspections for enforcement (CITES, 2006). Such techniques for porbeagle are already available (Holmes et al., 2009) and could be even potentially be used for distinguishing between Southern and Northern hemisphere stocks (proposal); however, they may be too expensive for routine separation of species at customs posts. Also, as noted in the proposal, it will be important to develop species-specific commodity codes for meat and fins of porbeagle in order to monitor the origin of products in trade. Currently, only porbeagle meat has specific commodity codes in the European Union (Member Organization). Recognizing this problem, the proposal requests an 18-month delay in the entry into effect of the listing, to resolve this and other technical and administrative issues.

2.3.5 "Look-alike" issues

In relation to "look-alike" issues, FAO (2007) noted that "Listing for 'look-alike' reasons (i.e., listing on Appendix II under Article II (b) of the Convention) is justified when enforcement officers who encounter specimens of CITES-listed species are unable to distinguish between them and unlisted species. Trade in porbeagle product is predominantly meat and fins. If the trade in products was undermining the conservation effectiveness of a porbeagle listing, and tools such as identification guides and DNA tests were not feasible, there would be potential justification for proposals to list other species of sharks on the basis that their products resemble those of porbeagle in trade, were porbeagle shark to be listed on Appendix II".

Considering the problems discussed in the section on identification of products in trade, it is expected that the listing of porbeagle in Appendix II of CITES will potentially create justification for listing other species of sharks with similar products in international trade.

2.4 Likely effectiveness of a CITES listing for the conservation of the species

The current Panel and FAO (2010) noted that, if properly implemented, a CITES Appendix II listing would be expected to result in better monitoring and reporting of catches entering international trade from all porbeagle populations and subpopulations. Improved catch monitoring should enable new or enhanced assessments of stock status and the subsequent adoption of management measures that ensure the sustainability of harvests. Harvests from international waters would fall under the IFS provisions of the Convention. These would require catch documentation to the species level for specimens entering the jurisdiction of a State from international waters, along with an NDF indicating that the harvest was sustainable.

Considering the measures in place in the European Union (Member Organization) and North America to control harvest and to rebuild stocks, the listing would mainly affect the meat trade from countries in the Southern Hemisphere to the European Union (Member Organization), and the shark fin trade to China and other Asian countries. A listing in CITES Appendix II would probably strengthen current efforts to ensure that harvesting for trade is commensurate with the Canadian rebuilding plan for the Northwest Atlantic stock.

The Panel also noted that the difficulty of identifying porbeagle products in trade and formulating NDFs might limit the effectiveness of a CITES listing. Requirements for additional information will create a burden that may need to be addressed through capacity building, particularly in developing countries. However, this is not unique to a potential CITES listing for porbeagle; it applies in general to all new management measures and regulations to utilize both marine and terrestrial species sustainably.

3. Conclusion

When evaluated on a population-by-population basis, the historically large porbeagle populations in the North Atlantic (Northeast and Northwest) and the Mediterranean Sea were considered to meet the Appendix II decline criterion.

Porbeagles in the Northeast Atlantic Ocean were considered to meet the Appendix II decline criterion, with no evidence that the decline has ceased. Past management appears to have been inadequate. The decline in the population abundance of the Northwest Atlantic stock meets the Appendix II decline criterion, although the population is currently recovering. Although no stock assessment has been performed, the tuna trap catch data for porbeagle in the Mediterranean Sea indicate that this population also meets the Appendix II decline criterion.

Assessments for the Southwest Atlantic region indicated substantial declines, but the results were too uncertain to determine whether porbeagle in this region meets the decline criterion for Appendix II. The status elsewhere in the Southern Hemisphere was considered to be above the Appendix II decline thresholds.

The 2009 Panel had taken note of the wording of CITES Resolution Conf. 9.24 (Rev. CoP14) indicating that Parties had resolved to adopt measures proportionate to the anticipated risks to the species when considering proposals to amend the Appendices. The 2009 Panel, taking into account that the apparently smaller, less-exploited Southern Hemisphere populations were considered to be above the Appendix II decline thresholds, considered that the species as a whole met the decline criteria for Appendix II.

The new information on distribution in the Southern Hemisphere was considered by some Panel members to indicate that porbeagle shark has a wider distribution in the Southern Hemisphere than previously thought and that this also indicated a higher abundance. In the view of these Panel members, this brings into question the conclusion of the 2009 Panel that the species globally meets the decline criteria for Appendix II. Other members of the Panel were of the opinion that the new study did not provide information on population size in the Southern Hemisphere or the relative abundance of the Northern and Southern Hemisphere populations and that therefore the information did not change the conclusion of the 2009 Panel. The majority of Panel members considered that the species as a whole meets the decline criteria for Appendix II.

The Panel agreed that porbeagle shark has low productivity. Life-history characteristics such as low fecundity, slow growth and late maturation, make the species particularly vulnerable to overexploitation. Such vulnerability factors are addressed in the decline criterion threshold for a low-productivity species.

Although porbeagle products are traded internationally, the actual proportion of the catches that is in international trade remains unknown owing to potentially substantial under-reporting and the lack of widely adopted specific customs codes for the species. However, available specific trade data from the European Union (Member Organization), one of the main markets for porbeagle meat, indicate that imports into the European Union (Member Organization) represent a significant part of the global reported catch. Porbeagle is also found in the global fin market.

The high value of products from the species, particularly its meat, in domestic and international markets constitutes a risk to the conservation of the species. In addition, the species is taken with longline fishing gear both in directed fisheries and as bycatch in fisheries for other high-value species such as tuna and swordfish.

High levels of unreported catch represent a significant potential risk factor as this will constrain accurate assessments of stock status, and subsequent management actions. Even in the area where stock status information is considered best (the Northwest Atlantic), unreported catch is apparently being taken (Campana and Gibson, 2008), and it is likely that actual global catches are substantially above reported catches (SCRS, 2009). The high seas catch taken by tuna longline fisheries in the North Atlantic used to be monitored only by observers, but some commercial catches and discards have been reported to ICCAT in recent years (ICCAT, 2012).

The existence of rebuilding plans in Canada and the United States of America represents an important mitigating factor for the Northwest Atlantic population. Catches in the high seas areas of the North Atlantic may undermine these efforts if they are not strictly regulated. The recently adopted EC Regulations prohibiting fishing for porbeagle shark in waters of the European Union (Member Organization) and also prohibiting fishing vessels flagged to the European Union (Member Organization) operating in all waters from fishing for, retaining on board, transshipping or landing porbeagle sharks is expected to mitigate to some extent the risk to the Northeast Atlantic population, and also to other populations affected by the fleet of the European Union (Member Organization). The Appendix III listing recently implemented by some countries of the European Union (Member Organization), which came into effect on 25 September 2012, is also likely to have a positive impact on improving information on the catches that enter international trade. In the Southern Hemisphere, management measures adopted by Argentina and New Zealand should also contribute to lowering the risk to porbeagle sharks in this area.

Several RFMOs have adopted regulations related to shark finning. However, finning regulations are unlikely to have much impact for porbeagle, given that the meat appears to be the most highly valued porbeagle product. Measures forbidding direct fisheries on sharks or requiring the live release of incidentally caught sharks, adopted by some tuna RFMOs, would be more beneficial for the conservation of the species.

The FAO International Plan of Action for the Conservation and Management of Sharks urges shark fishing nations to implement conservation and management plans that will lead to sustainable utilization of sharks. Some of the main porbeagle fishing countries have adopted an NPOA-Sharks, although the level of implementation of the plans is unknown. Strengthening the implementation of the International Plans of Action for Conservation and Management of Sharks (IPOA-Sharks) by countries and RFMOs could be expected to benefit the conservation of porbeagle throughout its range.

The current Panel and FAO (2010) noted that, if properly implemented, a CITES Appendix II listing would be expected to result in better monitoring and reporting of catches entering international trade from all porbeagle populations and subpopulations. Improved catch monitoring should enable new or enhanced assessments of stock status and the subsequent adoption of management measures that ensure the sustainability of harvests. Harvests from international waters would fall under the IFS provisions of the Convention. These would require catch documentation to the species level for specimens entering the jurisdiction of a State from international waters, along with an NDF indicating that the harvest was sustainable.

Considering the measures in place in the European Union (Member Organization) and North America to control harvest and to rebuild stocks, the listing would mainly affect the meat trade from countries in the Southern Hemisphere to the European Union (Member Organization), and the shark fin trade to China and other Asian countries. A listing in CITES Appendix II would probably strengthen current efforts to ensure that harvesting for trade is commensurate with the Canadian and the United States rebuilding plans for the Northwest Atlantic stock.

The Panel also noted that the difficulty of identifying porbeagle products in trade and formulating NDFs might limit the effectiveness of a CITES listing. Species-specific assessments that could provide a basis for NDFs are lacking in the Southern Hemisphere, and requirements for additional information will create a burden that may need to be addressed through capacity building, particularly in developing countries. However, this is not unique to a potential CITES listing for porbeagle; it applies in general to all new management measures and regulations to utilize both marine and terrestrial species sustainably.

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TABLES AND FIGURES

TABLE 1

Information for assessing the productivity level of porbeagle

Parameter	Information	Productivity	Source
Intrinsic rate of increase	0.05–0.07 (North Atlantic) 0.026 (Southwest Pacific)	Low	Proposal; Campana *et al*., 2001
Natural mortality	0.10 (immature), 0.15 (mature males), 0.20 (mature females, Northwest Atlantic), 0.05–0.1 (Southwest Pacific)	Low	Proposal; Campana *et al*., 2008
Age at maturity	NWA – F: 13 years; M: 8 years	Low	Natanson *et al*., 2002; Campana *et al*., 2002; Gibson and Campana, 2005
	NWA – F: 14 years; M: 7 years		Cassoff *et al*., 2007
	SWP – F: 15–18 years; M: 8–11 years		Francis and Duffy 2005; Francis *et al*., 2007
Maximum age	NWA – F: 24 years; M: 25 years; both: 19–30	Low	Natanson *et al*., 2002; Campana *et al*., 2002; Gibson and Campana, 2005
	NWA – both 24 years		Cassoff *et al*., 2007
	SWP – both 65 years		Francis *et al*., 2007
K	NWA – F: 0.061; M: 0.080; both: 0.066	Low	Campana *et al*., 2002; Gibson and Campana, 2005
	SWP – F: 0.060; M: 0.112		Francis and Duffy, 2005; Francis *et al*., 2007

Notes: Unless otherwise indicated, information is from the proposal. “Productivity” is relative to the guidelines in FAO (2001).

Source: FAO (2010) with revisions.

TABLE 2

Decline indices for porbeagle

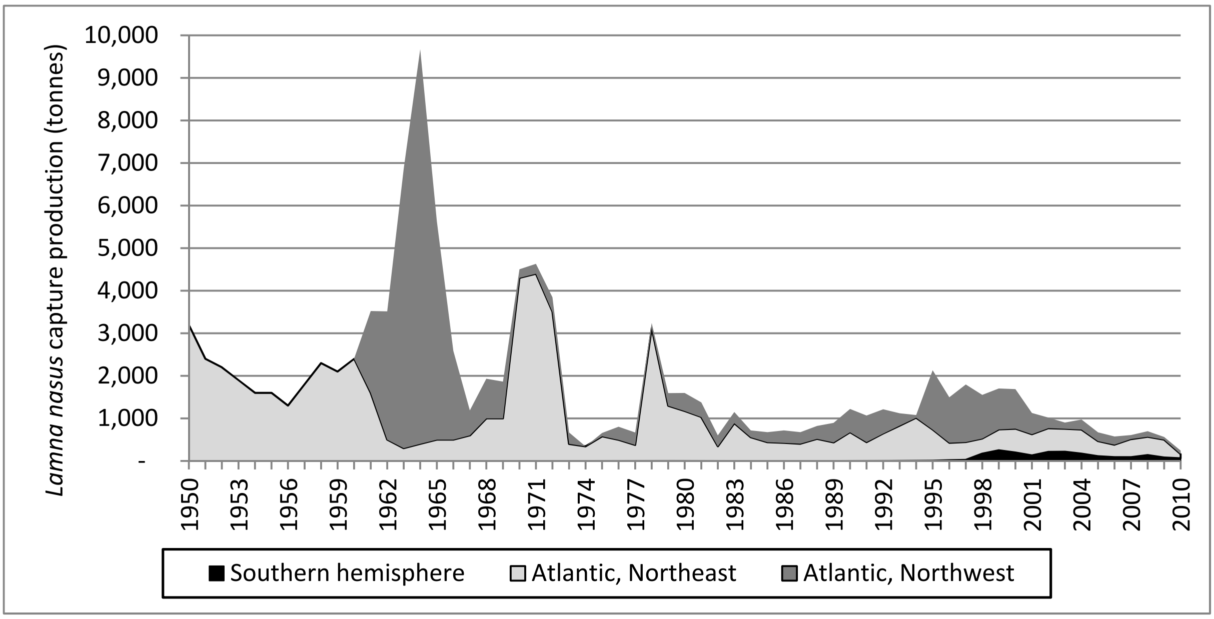
Area	Index	Trend	Basis	Coverage	Reliability	Source
NE Atlantic	Landings	Landings declined to 13% of historical peak of 2 953 tonnes in 1933–37. 20 tonnes landed in 2010.	Average landings 1933–37 vs 2004–08	Northeast Atlantic	Catch data (2)	Proposal; SCRS (2009); ICES (2011)
	Landings	Danish landings declined from average of 1 380 tonnes in 1950–54 to 6 tonnes in 2004–08 (< 1%). No landings reported in 2010.	Average landings 1950–54 vs 2004–08	Danish fleet	Catch data (2)	Proposal; SCRS (2009); ICES (2011)
	Landings	Norwegian landings declined from 2 953 tonnes/year in mid-1930s to less than 20 tonnes/year in 2004–08 (< 1% of peak). 12 tonnes reported in 2010.	Average landings 1933–37 vs 2004–08	Norwegian fleet	Catch data (2)	Proposal; FAO (2009); ICES (2011)
	CPUE	Decline by one-third from early 1970s and 2004–08	Inspection	French longline fleet	CPUE (standardized) (4)	SCRS (2009); FAO (2009); ICES (2011)
	Stock biomass	Decline to 15–39% of unexploited biomass	Surplus production model	Northeast Atlantic, 1926–2008	Catch data and CPUE (standardized) (4)	Proposal, SCRS (2009); FAO (2009)
	Stock biomass and numbers	Decline to 6% in biomass and 7% in numbers of unexploited biomass	Age-structured production model	Northeast Atlantic, 1926–2008	Catch data and CPUE (standardized) (4)	Proposal, SCRS (2009); FAO (2009)
Mediterranean	Compiled observations, landings	“Virtually disappeared”	Landings recorded in FAO FishStat, observations in research surveys.	Mediterranean	Catch data (2), observations (1)	Proposal; FAO FishStat; FAO (2009)
	Catches tuna traps	Decline of 99% between 1950 and 2006	GLM of catches over time	Ligurian Sea	Catch data (2)	Proposal; Ferretti *et al.* (2008)
	CPUE pelagic longlines	Decline of 98% between 1978 and 1999	GLM of CPUE over time	Ionian Sea	CPUE standardized (4)	Proposal; Ferretti *et al.* (2008)
Northwest Atlantic	Landings	Recent catches are 4% of historical highs Catches in 2010 less than 100 tonnes.	Average catch 2004–08 vs average catch 1961–65	Northwest Atlantic fishery	Catch data (2)	Proposal; Campana *et al.* (2010); FAO FishStat
	Stock biomass	Current stock is 32% of the size in 1961	Surplus production model	Northwest Atlantic	CPUE standardized (4)	Proposal; SCRS (2009); FAO (2009)

Area	Index	Trend	Basis	Coverage	Reliability	Source
	Total numbers	Current population size is 22–27% of its size in 1961	Age-structured model	Northwest Atlantic	CPUE standardized (4)	Proposal; SCRS (2009); FAO (2009); Campana *et al*. (2010)
	Numbers of mature females	Current numbers are 12–16% of numbers in 1961	Age-structured model	Northwest Atlantic	CPUE standardized (4)	Proposal; SCRS (2009); FAO (2009); Campana *et al*. (2010)
Southern Hemisphere	Stock biomass	Current stock biomass about 18–39% of the unexploited stock size	Surplus production model	Southwest Atlantic	CPUE of Uruguayan fleet (4)	SCRS (2009); FAO (2009)
	Spawning stock biomass (SSB)	Current SSB is 18% of unexploited SSB	Catch-free, age-structured production model	Southwest Atlantic	CPUE of Uruguayan fleet (4)	SCRS (2009); FAO (2009)
	Landings	Decline of 82% between 1999 and 2008	Inspection	New Zealand	Landings (2)	Proposal; FAO FishStat
	Longline CPUE	Decline to about 30% between 1992 and 2002	Inspection	New Zealand	unstandardized CPUE (3)	Proposal; Ministry of Fisheries New Zealand
	Deep longline CPUE	No decline, recent increase	Scientific survey and commercial fisheries	Southern Hemisphere	Standardized CPUE (4)	Semba, Yokawa and Matsunaga (2012)

Source: Reliability indices are based on FAO (2001).

FIGURE 1

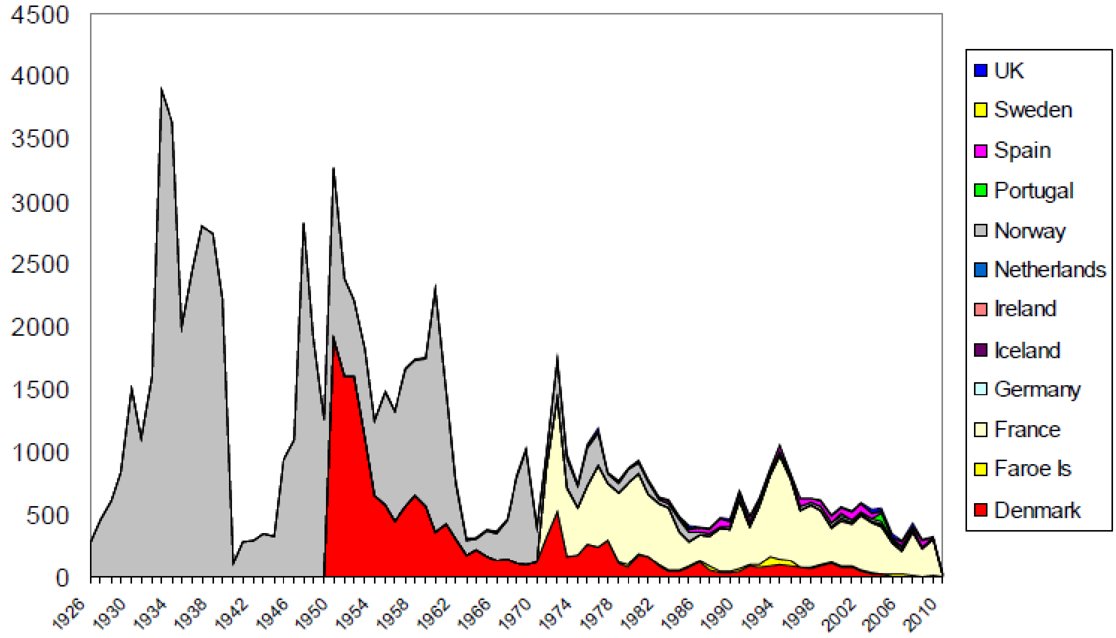
Global reported catches of *Lamna nasus*



Source: FAO FishStat (proposal).

FIGURE 2

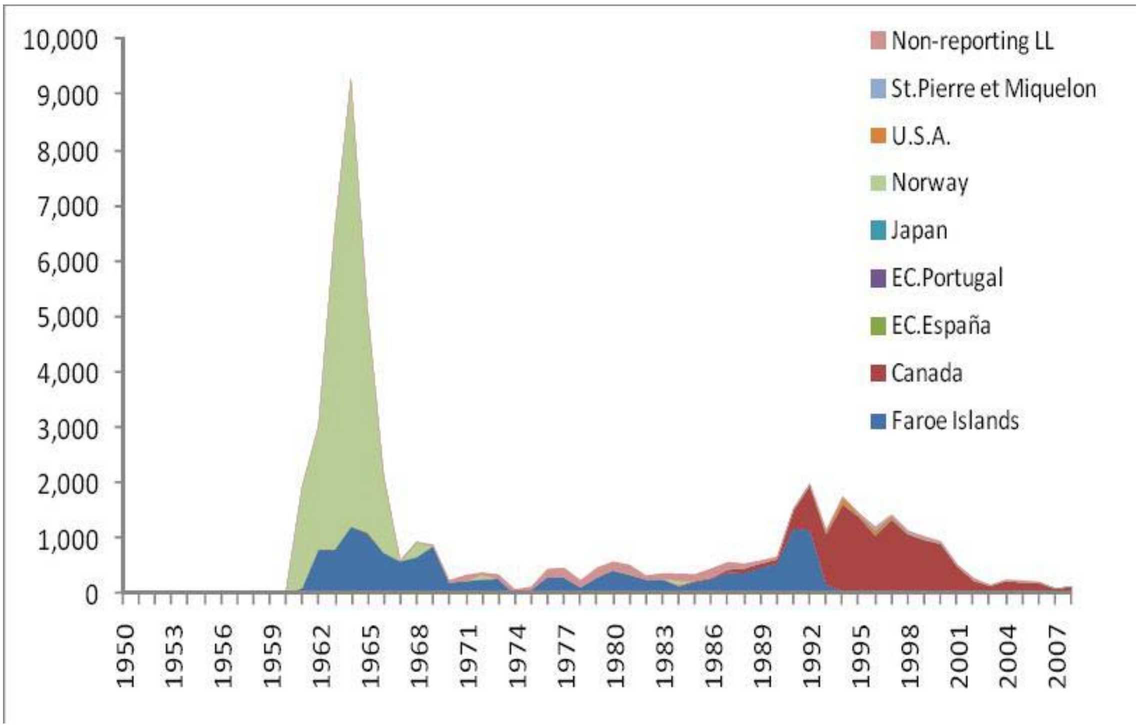
Reported catches of porbeagle sharks from the Northeast Atlantic by country



Source: ICES (2011).

FIGURE 3

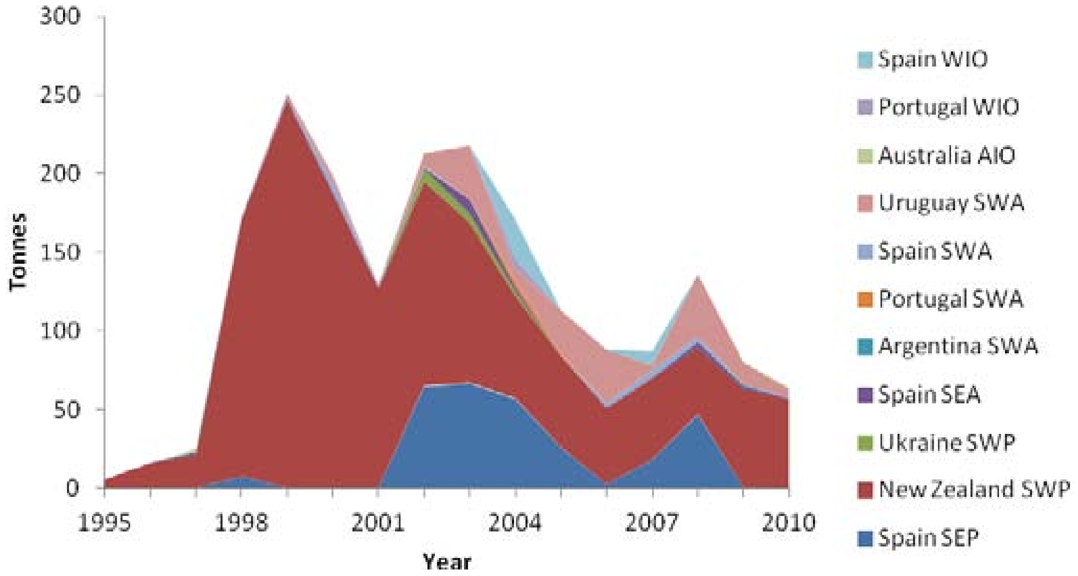
Catch of porbeagle sharks from the Northwest Atlantic by country used in the assessment undertaken by the SCRS



Source: SCRS (2009).

FIGURE 4

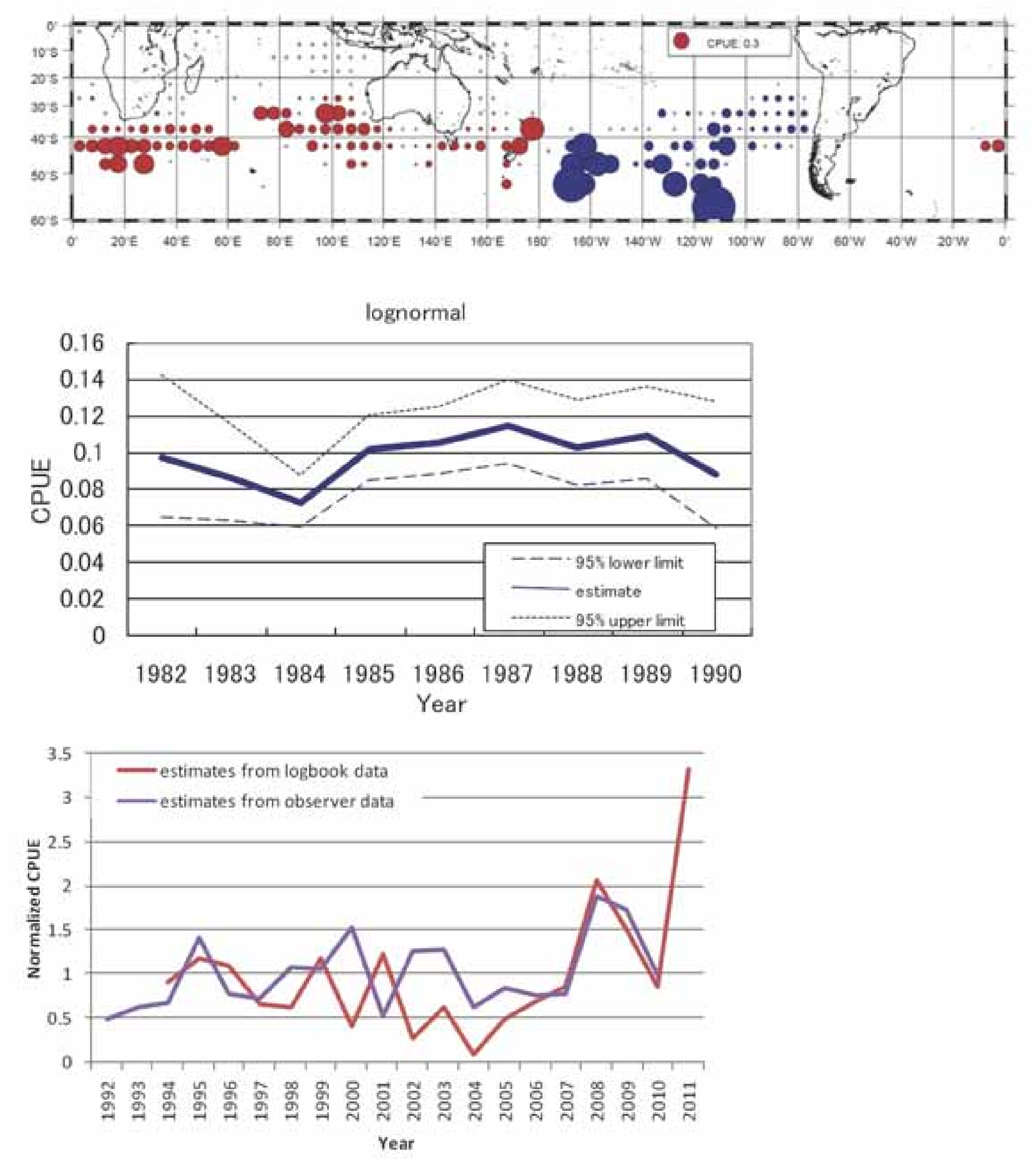
Reported catches of porbeagle shark in the Southern Hemisphere



Source: FAO FishStat.

FIGURE 5

Distribution of the average CPUE of porbeagle caught in the Southern Hemisphere

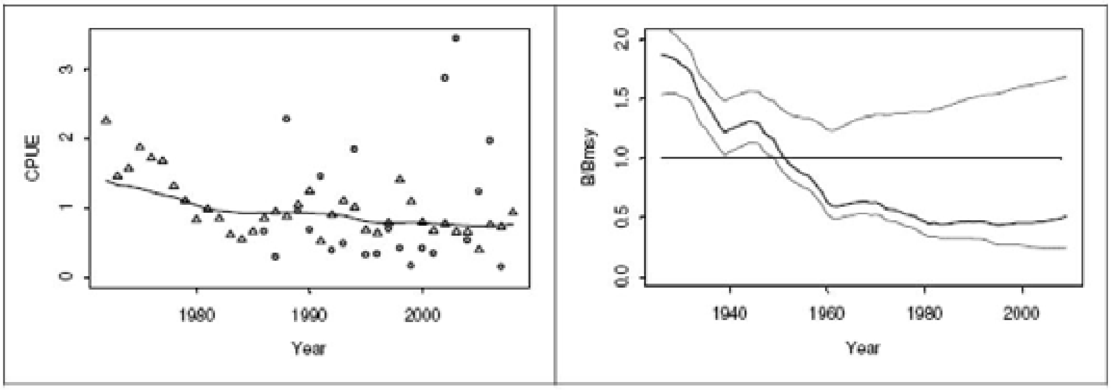


Notes: The upper panel shows the distribution of the average CPUE of porbeagle caught in the Southern Hemisphere by Japanese longliners. Red and blue circles are the CPUE of commercial longliners and exploratory longline research, respectively. The middle panel shows trends in the standardized CPUE of a driftnet survey in 1982–1990. The lower panel shows trends in the standardized CPUE of commercial longline in 1992–2010 (CCSBT observer data) and 1994–2011(logbook data).

Source: Semba *et al*. (2012).

FIGURE 6

Results of a Bayesian surplus production model for the Northeast Atlantic porbeagle stock

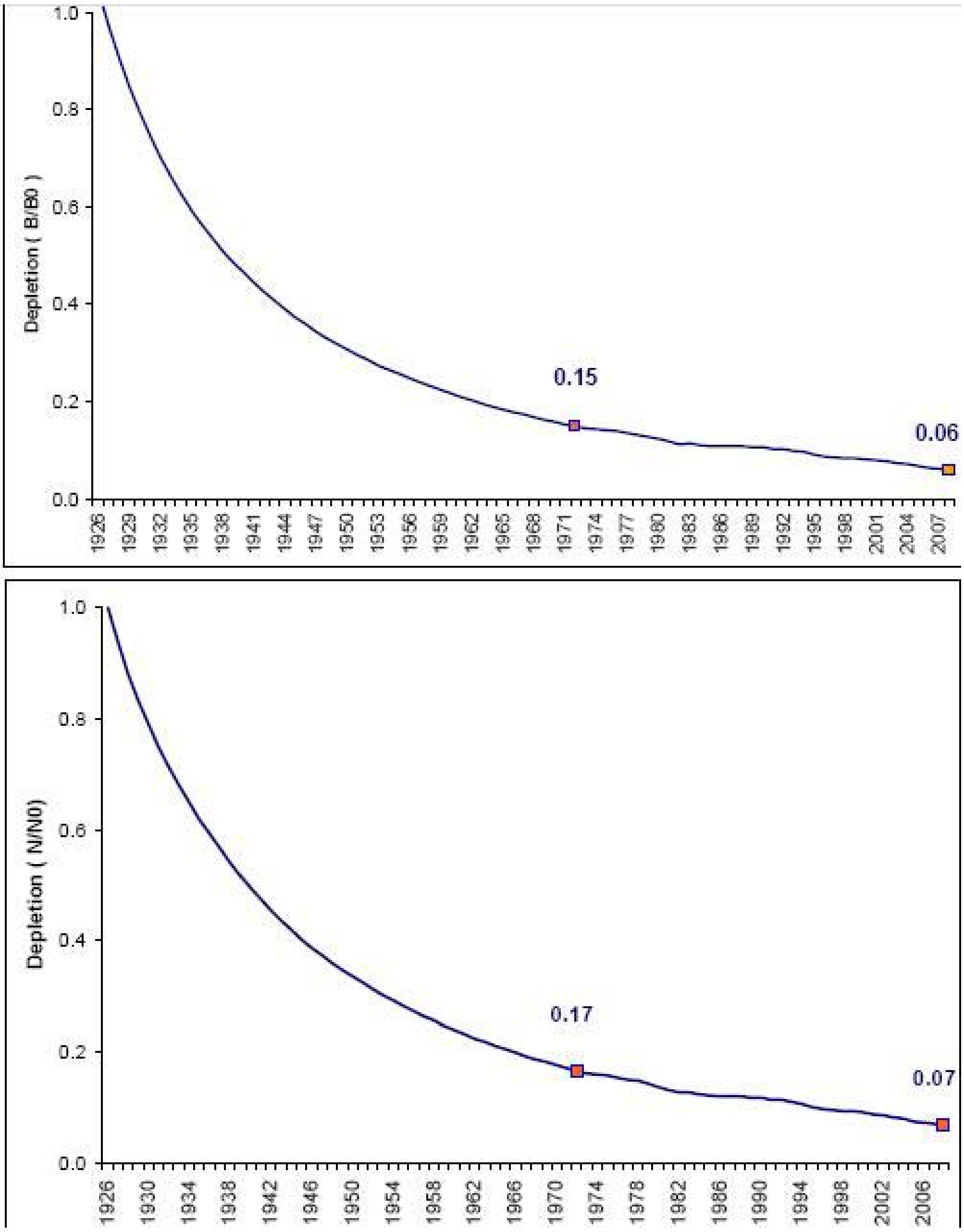


Notes: Left: standardized French and Spanish CPUE (each series divided by its mean) and fitted biomass trend. Right: biomass (B) relative to biomass at MSY (B_{MSY})

Sources: SCRS (2009); proposal (Figure 10).

FIGURE 7

Depletion of total biomass (upper panel) and numbers (lower panel) for an age-structured production model assuming virgin conditions in 1926 for porbeagle shark in the Northeast Atlantic

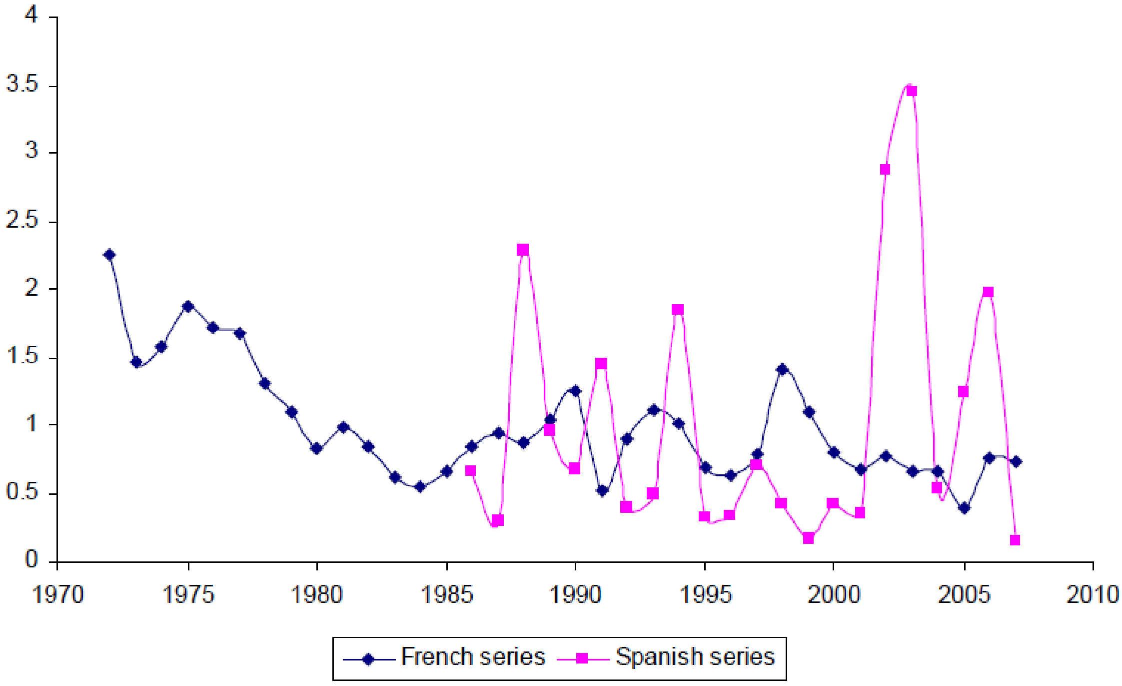


Notes: The dots indicated on the line correspond to depletion at the beginning of the modern period (1972) and recent depletion (2008)

Source: SCRS (2009).

FIGURE 8

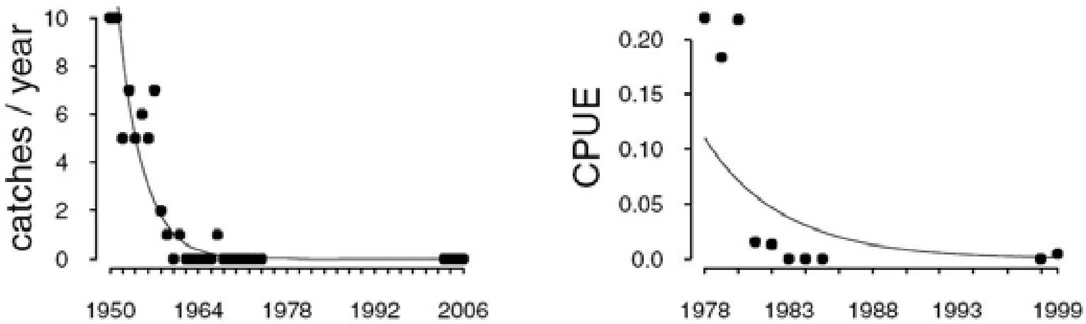
French and Spanish porbeagle CPUE from longline fisheries in the Northeast Atlantic



Source: ICES (2011).

FIGURE 9

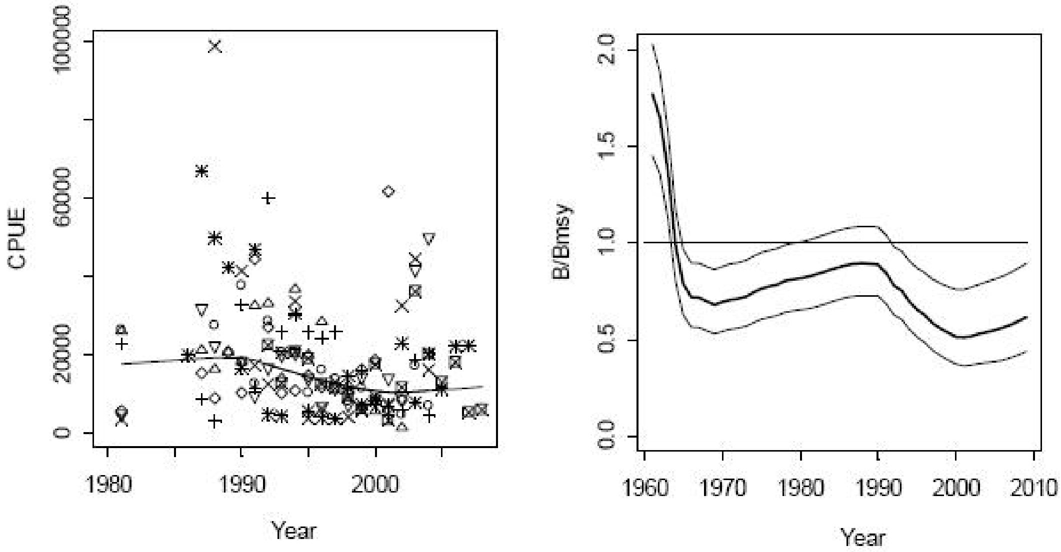
Trends in porbeagle standardized annual catches in tuna traps in the Ligurian Sea (left) and in CPUE (sharks landed per 1 000 hooks) for the Ionian Sea



Source: Ferretti *et al.* (2008).

FIGURE 10

Results of a Bayesian surplus production model of the Northwest Atlantic porbeagle stock

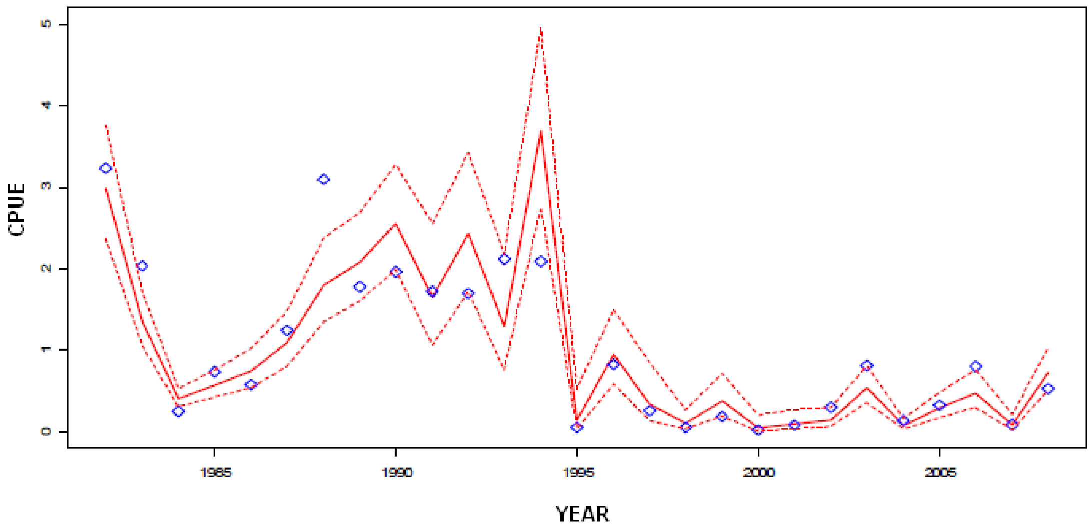


Notes: Left: Canadian, United States and Spanish longline CPUE (each divided by its catchability to be in units of biomass) and fitted biomass trend. Right: biomass (B) relative to biomass at MSY (B_{MSY})

Source: SCRS (2009).

FIGURE 11

Nominal (dots) and standardized catch rates (line) for porbeagle sharks from the Uruguayan pelagic longline fleet, 1982–2008.

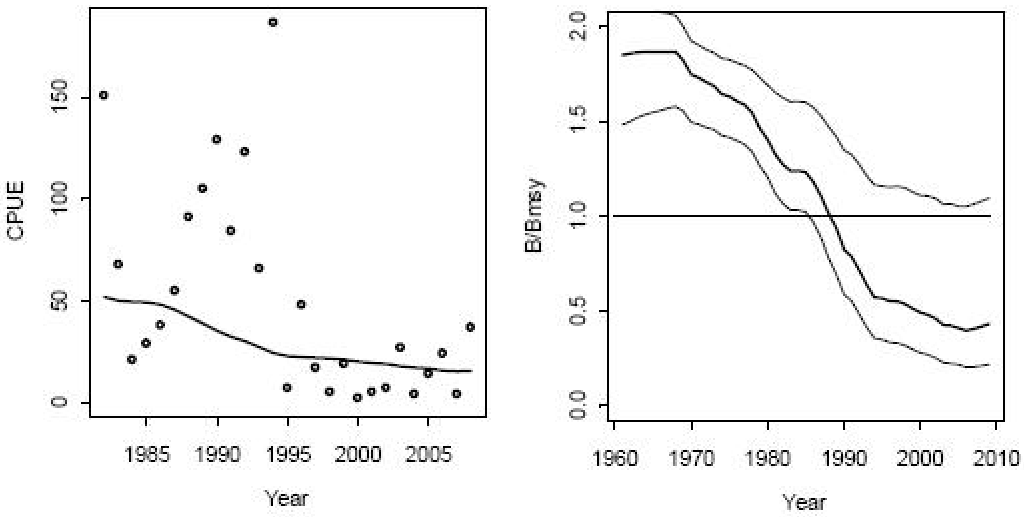


Note: Dotted lines represent 95 percent confidence intervals for the standardized catch rates.

Sources: SCRS (2009); Pons and Domingo (2010).

FIGURE 12

Results of a Bayesian surplus production model of the Southwest Atlantic porbeagle stock, assuming that catches are proportional to effort

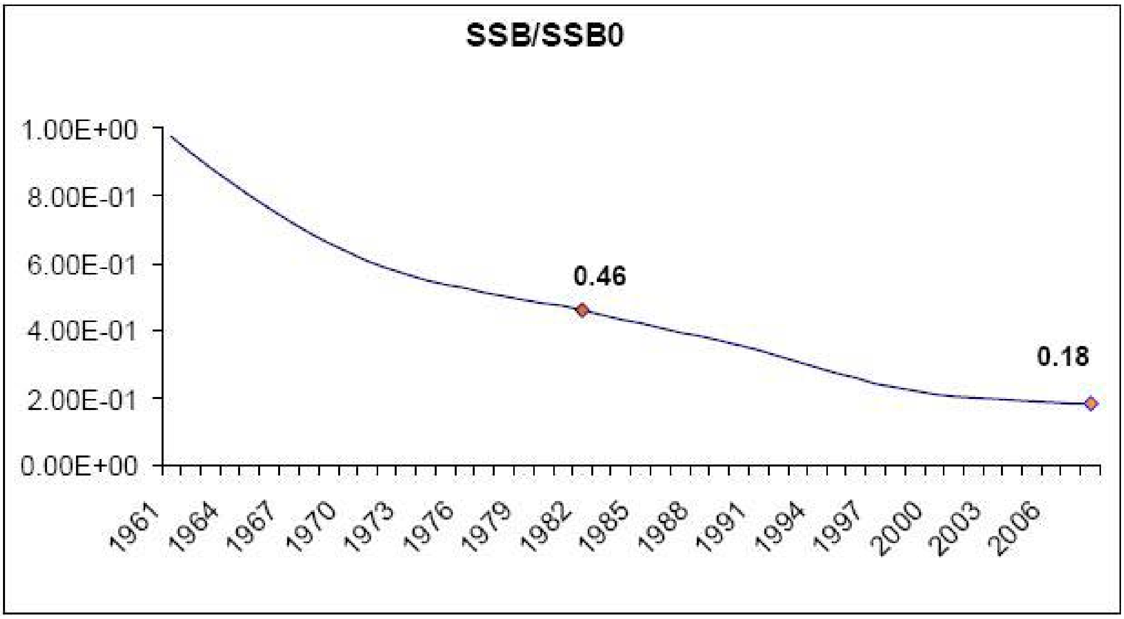


Notes: Left: standardized Uruguayan longline CPUE (unstandardized data was in kilograms per 1 000 hooks) and fitted biomass trend. Right: biomass (B) relative to biomass at MSY (B_{MSY}).

Source: SCRS (2009).

FIGURE 13

Relative spawning stock biomass (SSB) trend for the catch-free age-structured production model assuming virgin conditions in 1961 for Southwest Atlantic porbeagle shark

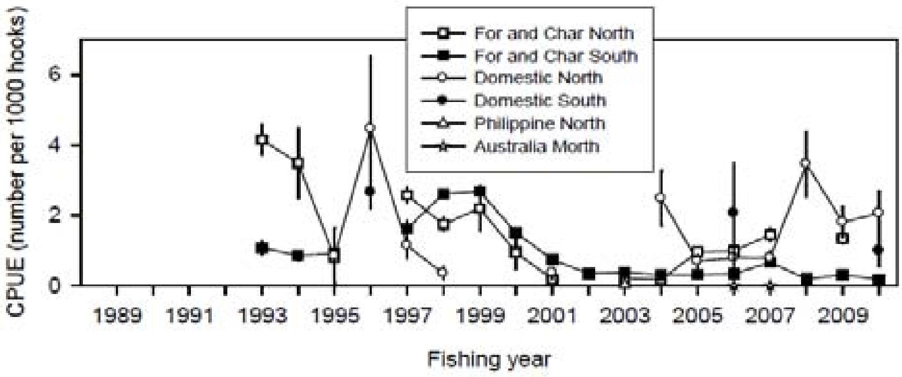


Note: Dots indicate the depletion at the beginning of the modern period (1982) and recent depletion (2008).

Source: SCRS (2009).

FIGURE 14

Unstandardized CPUE indices (number of *Lamna nasus* per 1 000 hooks) for various New Zealand tuna longline fleets based on observer reports

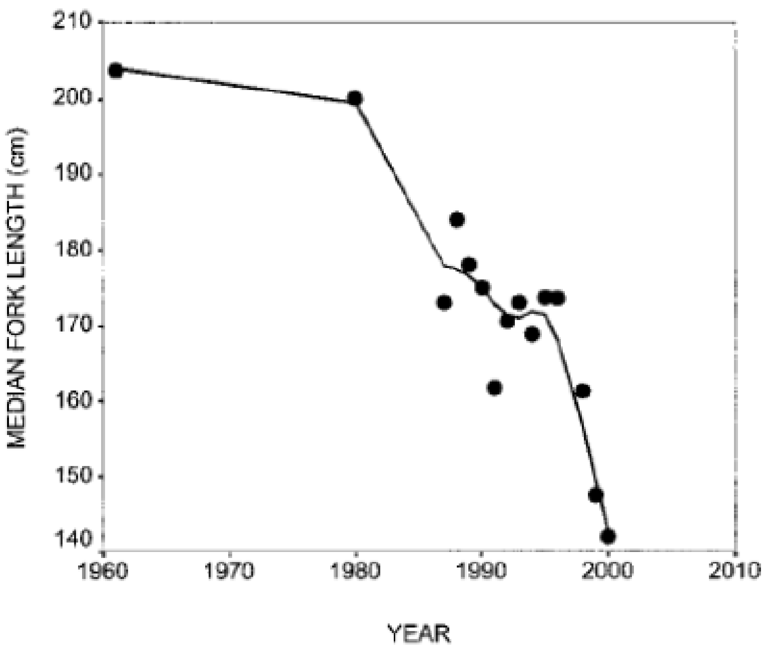


Notes: Values are the mean estimates with 95 percent confidence limits. The CPUE results from the domestic fleet should be interpreted with caution owing to the lower observer coverage of this fleet. CPUE estimates for the charter fleet can be considered reliable from 1993 onwards.

Source: Griggs and Baird (in press), cited in New Zealand Ministry for Primary Industries (2012).

FIGURE 15

Changes in the median fork length of porbeagle in commercial catch in September–November on mating grounds off southern Newfoundland



Note: A LOESS line is fitted to the data.

Source: Campana *et al*. (2001).

FAO Expert Advisory Panel assessment report: freshwater sawfish - CoP16 Proposal 45 -

Species:

Pristis microdon – freshwater sawfish.

Proposal:

Transfer of *Pristis microdon* from Appendix II to Appendix I of CITES in accordance with Article II, paragraph 1, satisfying Criteria A.(i), (v); B.(i), (ii), (iii) and (iv); and C.(i), (ii) in Annex 1 of Resolution Conf. 9.24 (Rev. CoP15).

Basis for proposal:

The proposal indicates that the historical decline in the area of distribution and in the number of individuals have resulted in fragmented populations of *P. microdon*. Also according to the proposal, new evidence of strong female philopatry (individuals remaining or returning to their birth place) indicates that the Australian population is divided into several subpopulations, with limited opportunity for re-establishment. This characteristic makes the species more vulnerable to removals. Vulnerability is also compounded by life-history parameters and by the species susceptibility to multiple threats, including to fisheries bycatch. Considering the above, the proponent considers that commercial exports of *P. microdon* may have a detrimental impact on the survival and recovery of the species and should not be allowed any longer. *P. microdon* is the only species of Pristidae in Appendix II: all other species are listed in Appendix I. The proposal indicates that transfer of the species from Appendix II to Appendix I will ensure maximum conservation benefit for the family and an easier enforcement of the sawfish listings by reducing the ability for "look-alike" or illegal trade.

ASSESSMENT SUMMARY

CITES biological listing criteria

The Panel found the available information indicates that the freshwater sawfish *Pristis microdon* meets the biological criteria for an Appendix I listing. A similar conclusion was reached by FAO (2007) when assessing the proposal for listing all species of Pristidae in Appendix I.

Comments on technical aspects of the proposal

Biology and ecology: The freshwater sawfish *Pristis microdon* was known to occur in the Indo-West Pacific but limited scientific records and other observations suggest abundance has declined to a small fraction of historical levels. Demographic information from other Pristidae species indicates that sawfishes have a low productivity. Recent genetic studies indicate that the population of Northern Australia *P. microdon* has high levels of mtDNA heterogeneity and no nDNA heterogeneity. These results suggest that *P. microdon* may have a male-biased dispersal. While females remain or return to pupping sites, males are more wide-ranging, being responsible for the gene flow across assemblages.

Trade: Sawfish parts and products of all species are already included under Appendix I; only live individuals of *Pristis microdon* can be traded internationally under Appendix II.

Fisheries management: Only a few range States have adopted management measures to control the take of the species, including Australia, Bangladesh, India, Indonesia, and Malaysia. In addition, all shark fishing is banned in Myanmar.

Likely effectiveness of a CITES listing for the conservation of the species: Any trade in freshwater sawfish products is already prohibited by CITES because the current Appendix II listing only allows the export of live specimens under specified circumstances. Retaining live specimens of all species listed under Appendix I could facilitate the implementation of CITES regulations, as identification to the species level would no longer be necessary.

DETAILED PANEL ASSESSMENT

1. Scientific assessment in accordance with CITES biological listing criteria

1.1 Biological aspects

1.1.1 Population assessed

The freshwater sawfish *Pristis microdon* is an elasmobranch species member of the family Pristidae, which includes seven species of sawfish.[6] *P. microdon* was known to occur in coastal and freshwater habitats in the Indo-West Pacific, but recent distribution is uncertain because of presumed localized extirpations across the region (proposal). Northern Australia is believed to be one of the last strongholds of the species (Phillips, 2012). In this area, the species has been recorded from rivers, estuaries and marine environments up to 100 km offshore and up to 400 km upstream (proposal). Adults occur in marine and/or estuarine waters. Females pup near river mouths and juveniles migrate upriver to freshwater nursery habitats, where they remain until they reach sexual maturity (proposal; Thorburn *et al*., 2007; Phillips, 2012).

Recent genetic studies using markers with different modes of inheritance (mitochondrial DNA, which is maternally inherited, and nuclear DNA, which is bi-parentally inherited) showed that the population of northern Australia has high levels of mtDNA heterogeneity and no nDNA heterogeneity (proposal; Phillips, 2012). These results suggest that *P. microdon* may have a male-biased dispersal. While females are strongly philopatric (remain or return to pupping sites), and are distributed in independent maternal assemblages, males are more wide-ranging, being responsible for the gene flow across assemblages. As discussed by Philips (2012), the presence of male-biased dispersal could imply that males are migrating between regions to reproduce or, alternatively, that there is a single breeding ground for the studied assemblages in northern Australia. Although further genetic and tagging studies are required to understand the structure of the population, the results obtained so far are indicative that the population in northern Australia may be divided in distinct subpopulations. Taking into account the available genetic data, five broad management units have been suggested for *P. microdon* in Australia, each comprising a group of river systems (DEHWA, 2009): West Coast of Western Australia; Northern Territory and East Kimberley; Southern and Western Gulf of Carpentaria; North Eastern Gulf of Carpentaria; and East Australian Coast.

1.1.2 Productivity level

Demographic information from other Pristidae species indicates that sawfishes have a low productivity (FAO, 2007). For example, natural mortality was estimated from 0.07 to about. 0.15 per year for *P. pectinata* and *P. perotteti* (Simpfendorfer, 2000). The biology of *P. microdon* is poorly known, especially the marine adult phase, which is the least-sampled life stage. Overall, the available information indicates that the species has also a low level of productivity. *P. microdon* shows fast growth rates during the juvenile phase in freshwater environments, reaching more than 2 m in total length (TL) by age four (Thorburn *et al*., 2007). Males appear to leave the rivers at about 2.5 m TL and females at 2.8 m TL, presumably prior to attaining maturity (Whitty *et al*., 2008). Sexual maturity is reached at approximately eight years of age and longevity is likely to be well above the observed maximum age of 28 years (Phillips, 2012). The species appears to reproduce annually, with an observed average fecundity of 12 pups (DEHWA, 2009).

[6] The taxonomy of sawfish is currently under revision and the number of species may consequently change in the future.

1.1.3 Anthropogenic sources of mortality

P. microdon is susceptible to different sources of anthropogenic mortality in marine and freshwater environments, including those resulting from direct exploitation and bycatch in commercial and recreational fisheries and habitat degradation (proposal). The relative importance of these sources of mortality is generally unknown. Habitat modification and bycatch mortality were considered the most prominent threats in Australia (Phillips, 2012).

Reported catches to FAO of sawfishes in the Indo-West Pacific oscillated around 50 tonnes per year from 1985 to 2005, increasing rapidly after that to a peak of about 200 tonnes per year in recent years (Figure 2). The actual proportion of *P. microdon* in this total is unknown owing to the lack of species-specific catch data.

The degradation of mangroves and estuaries, resulting from urban and industrial development, and the construction of barrages on rivers, which impede the upstream movement of juveniles, are recognized as important habitat modification processes affecting freshwater sawfishes (DEWHA, 2009; Phillips, 2012).

The long toothed rostra makes the species particularly susceptible to entanglement in fishing gear. Bycatch mortality in gillnet and trawl fisheries is recognized as the greatest fishing threat (proposal; DEWHA, 2009). Because some of the products from sawfishes are highly valued (e.g. rostra, teeth, fins), there is an often incentive to illegally retain individuals caught as bycatch, rather than return them to sea. Sawfishes generally survive capture and, if adequately released, would have a high chance of survival (DEWHA, 2009).

The species is also caught in recreational and indigenous fisheries. Sawfishes are cultural and spiritual icons for indigenous people in northern Australia (McDavitt, 2005), and an unknown number of juveniles are removed from rivers as a part of the indigenous harvest (Phillips, 2012). However, the impact of indigenous takes on the population is probably reduced because the fishery is focused on the juvenile phase when natural mortality is high (DEWHA, 2009). Also according to the review carried by DEWHA (2009), the mortality incurred by recreational fishing is probably declining as a result of educational campaigns to release individuals incidentally caught.

Since 2007, international trade in *P. microdon* has been restricted under CITES (Appendix II listed species) “for the exclusive purpose of allowing international trade in live animals to appropriate and acceptable aquaria for primarily conservation purposes”. Between 30 and 40 animals have entered the aquaria trade since 1998 from Australia, 9 being traded after the listing in 2007. Because of the uncertainties in making NDFs for the species (DSEWPaC, 2011), the Government of Australia has recently decided to prohibit international trade in the species from the country (proposal).

1.1.4 Population status and trends

Population size

There are no estimates of the population size of *P. microdon* across its range.

Area of distribution

The species was once distributed throughout the Indo-West Pacific but has become rare or locally extinct in parts of its former range (Table 2). Currently, northern Australia seems to be the last stronghold of the species. The historical distribution of the species range includes Indonesia – Arafura Sea, west, east and central Kalimantan, Indragiri River near Rengat, Sumatra and the Java Sea; Papua New Guinea – Fly River system, Sepik River, Laloki River and Lake Murray; Malaysia – Kinabatangan, Perak and possibly Tembeling and Linggi Rivers; Thailand – possibly from Mae Nam Chaophraya River at Nantauri and above Paknam; Cambodia – Grand Lac; the Philippines – Luzon (Laguna de Bay, Bikol River and Camarines Sur Province), Lake Naujan, Mindoro, Mindanao (Rio Grande and Liguasan Swamp, Cotabato Province, and Agusan River at Moncayo, Davao Province); Myanmar and India – Ganges and Brahmaputra (Compagno, Dando and Fowler, 2005; Compagno and Last, 1999; Last and Stevens, 1994, 2009). The occurrence of *P. microdon* in Sri Lanka, Pakistan, Oman, the Red Sea, Madagascar, Mozambique, and Zimbabwe is dependent on the taxonomic understanding of the genetic relationship with other pristid species (Last and Stevens, 2009). It still occupies a relatively large area of distribution in northern Australia, believed to extend from Eighty Mile Beach in Western Australia to Princess Charlotte Bay on the east coast of Queensland (Phillips, 2012).

Population trend

There are no quantitative estimates of population decline across the species range, but there are many observations of greatly reduced abundance relative to historical levels and of extirpations from substantial parts of historical ranges (Table 2).

The strong decline in global landings reported to FAO of sawfishes (Pristidae) since a global peak of 1 759 tonnes in 1978 is indicative of global declines of the family (Figure 1, proposal). However, global catches are inadequate to infer the status of the population of *P. microdon*, which is restricted to the Indo-West Pacific. Reported landings of sawfishes in this region have followed a different pattern of change (Figure 2). Landings oscillated without a trend from the late 1980s to 2005, increasing rapidly since then to a peak of about 200 tonnes/year in 2009/2010. This pattern of change would not be consistent with a decline in population abundance. However, landings data have a low reliability as an index of population abundance because of the influence of factors such as changes in effort, management measures, market conditions, discarding practices and data reporting. It is particularly inaccurate in this case because landings are reported at the family level.

One of the longest time series of data for Pristidae in Australia is from the Queensland Shark Control Program, which operates bather-protection fishing gear along the east coast of Queensland (Stevens, Pillans and Salini, 2005). The data, based on a continuous sampling effort in the same location, show a marked decline in catches of Pristidae from the late 1960s to the late 1990s off the northern town of Cairns and the practical disappearance of the species in southern towns of Townsville and Rockhampton since the early 1990s (Figure 3). However, the extent of occurrence *P. microdon* in the east coast of Queensland and the actual proportion of the species in these data sets are unknown.

1.2 Assessment relative to quantitative criteria

1.2.1 Small population

There are no estimates of total population numbers for the species. Recent genetic studies indicate the likely existence of female philopatry to pupping grounds (Phillips, 2012). Although the size and capacity of re-establishment of these female assemblages are unknown, the possibility of having assemblages of small size cannot be excluded.

1.2.2 Restricted distribution

No guidelines for restricted area of distribution are provided in the CITES criteria, which indicate that thresholds should be taxon-specific (Conf. Res. 9.24 Rev. CoP15). FAO (2001) recommended that historical extent of decline in area of distribution would be a better measure of extinction risk than absolute value of distributional area, but that if no other suitable information is available and absolute area of distribution has to be used for an exploited fish population, analyses should be on a case-by-case basis as no numeric guideline is universally applicable.

Although the available data indicate that the species has experienced a high level of range contraction, it still occupies a relatively large area of distribution in northern Australia, which is considered one of the last refuge areas of the species.

1.2.3 Decline

Under the CITES criteria for commercially exploited aquatic species (Conf. Res. 9.24 Rev. CoP15), a decline to 15–20 percent of the historical baseline for a low-productivity species would justify consideration for Appendix I.

Scientific records and other observations from many parts of the range suggest that abundance has declined to a small fraction of historical levels (Table 2). In many cases, the evidence is anecdotal and not species-specific. Although few of these estimates are quantified and most have relatively low reliability, overall it seems likely that the species meets the decline criteria for a CITES Appendix I listing, as concluded by FAO (2007) for all species of Pristidae.

Were trends due to natural fluctuations?

The strong correlation between sawfish recruitment and the length of the wet season in Western Australia suggest that long-term climate change may have an impact on *P. microdon* population (DEWHA, 2009). However, there is no indication in the sources available that observed historical declines were due to natural fluctuations.

2. Comments on technical aspects in relation to trade, management and implementation issues

2.1 Trade aspects

Since the species was listed in Appendix II of CITES in 2007, international trade in *P. microdon* has only been allowed to appropriate and acceptable aquaria, primarily for conservation purposes. According to the information presented in the proposal, nine specimens have been legally exported from Australia (and 100 mg of sawfish ear bones for scientific purposes) since 2007. A recent review of the NDF for the species indicated that, because of the impossibility of estimating mortality rates, it was not possible to determine if any level of harvesting for trade would be detrimental to the survival of the species (DSEWPaC, 2011). In view of this conclusion, exports from Australia are no longer permitted (proposal).

FAO (2007) noted that given the rarity of sawfishes and the apparent decline in directed fisheries, products entering international trade may originate mainly from fish incidentally caught. Given the high value of rostra, teeth and fins, there is a concern that products from incidentally caught fish may be illegally traded, despite the CITES listing.

2.2 Fisheries management aspects

As noted in the proposal, a few range States have adopted management measures to control the take of the species, including Australia, Bangladesh, India, Indonesia[7] and Malaysia. In addition, all shark fishing is banned in Myanmar.

In Australia, the main range State, *P. microdon* is legally protected in the three territories of its occurrence (Western Australia, Northern Territory and Queensland). Specific management measures are in place in each territory (proposal). The species cannot be harvested without a permit by commercial and recreational fishers in Queensland. In Northern Territory, retention by commercial fishers is forbidden without a permit, and as of January 2010, retention by recreational fishers is forbidden (DSEWPaC, 2011). All commercial and recreational take, including incidental mortality, has been prohibited since 2005 in Western Australia. Queensland currently permits the removal of a number of *P. microdon* specimens from the Queensland Gulf of Carpentaria for aquarium use. Indigenous harvesting of sawfishes occurs in all three territories, but the level of take is unknown (DSEWPaC, 2011).

While these domestic measures are likely to be reducing mortality, DSEWPaC (2011) noted that the lack of data on the extent of historical decline and on the level of recovery of the population precludes an evaluation of the effectiveness of the measures adopted in Australia. It appears, for example, that

[7] Indonesia has banned fishing for sawfishes since 1999.

IUU fishing is an important threat to the population, particularly in the southern, western and northeastern Gulf of Carpentaria (DEWHA, 2009). There is no information on the effectiveness of the protection mechanisms in place in the other range Sates.
P. microdon was listed in Appendix II of CITES in 2007 for the exclusive purpose of allowing international trade in live animals to appropriate and acceptable aquaria primarily for conservation purposes. All other species of sawfishes were listed in Appendix I in the same year. The impact of the listing on the species conservation is unknown. For one thing, the reported landings of sawfishes increased substantially in the two years following the listing (Figures 1 and 2). Whether catches were for domestic use only or illegally exported is not known. Indonesia and Iran (Islamic Republic of) were the two main countries reporting sawfishes catches in the period. Only a few specimens were exported from Australia since the listing (see section 1.1.3.). In 2011, the revision of the species NDF by the Australian Scientific Authority for Marine Species concluded, “it is currently not possible to conclude with a reasonable level of certainty that any harvest of *P. microdon* for export purposes would not be detrimental to the survival or recovery of the species” (DSEWPaC, 2011). Therefore, trade in sawfish from Australia is no longer allowed.

2.3 Implementation issues

The Panel agrees with relevant observations made by FAO (2007) with respect to the implementation of an Appendix I listing for Pristidae. The following is based on these previous observations.

2.3.1 Basis for findings: legally obtained, non-detrimental

An Appendix I listing means that international trade is only permitted in exceptional circumstances. Both an export and an import permit are required for any shipment. An import permit can only be issued if the import is not for primarily commercial purposes, and also requires a finding that the purpose of the import will not be detrimental. An export permit requires an NDF and a finding that the specimen was legally obtained. Exemptions are in place for personal or household effects (not for sale) in specified circumstances, and for pre-Convention specimens.

The determination that the import of a shipment is not for primarily commercial purposes would essentially eliminate most of the existing international trade in freshwater sawfish products. Examples of trade that may be considered non-commercial might include international movements for non-commercial exhibitions or for scientific purposes.

The exemption for personal and household effects (curios) applies only in specific circumstances. In practice, it is difficult to take advantage of this exemption, particularly for specimens listed in Appendix I, because customs authorities frequently require official proof that it applies. The pre-Convention exemption requires proof that the specimen was obtained prior to entry into force of the listing; some Parties treat this provision as applying to specimens obtained prior to entry into force of the Convention for that individual Party (1975 or later). Methods exist to date specimens of Pristidae but these are expensive to use.

2.3.2 Identification of products in trade

Sawfish parts and products of all species are already included under Appendix I; only live individuals of freshwater sawfish can be traded internationally under Appendix II. Having also live specimens of all species listed under Appendix I could facilitate the implementation of CITES regulations as identification to the species level would no longer be necessary.

Some sawfish products in trade are easily identifiable, in particular rostra and live specimens. Rostra of a similar group, the sawsharks, are easily distinguishable from those of sawfishes with an appropriate identification guide.

Currently, international trade of sawfish products is banned under CITES. Before the CITES ban, rostral teeth were traded in international markets with a variety of similar products for use as cockfighting spurs: deer antler, bones, sting ray spines, sea turtle shell, sea lion teeth.

Before the ban of international trade under CITES, sawfish fins in general were highly valued in international markets and were traded in the China, Hong Kong SAR shark fin market (McDavitt, 1996; Parry-Jones, 1996). Experienced traders in dried shark fins could identify them to the family level, but this would probably be impossible for a non-specialist. Powder derived from dried sawfish rostra and teeth would be very difficult to distinguish from other powders used in traditional medicines. A forensic DNA test is available for *P. pectinata* (Feldheim *et al*., 2010) and should be developed for the remaining species to ensure identification.

2.3.3 "Look-alike" issues

Look-alike issues would not arise with an uplisting of freshwater sawfish: having live specimens of all species listed under Appendix I could facilitate the implementation of CITES regulations as identification to the species level would no longer be necessary.

2.4 Likely effectiveness of a CITES Appendix I listing for conservation

Any trade in freshwater sawfish products is already considered illegal by CITES because the current Appendix II listing only allows the export of live specimens for the aquarium trade. It can therefore be concluded that the transfer of the species from Appendix II to Appendix I will not affect the effectiveness of trade control measures. The Appendix I listing is also unlikely to reduce the anthropogenic sources of mortality affecting the recovery of the species.

3. Conclusion

Although no quantitative data were available to estimate the historical extent of decline of the population, the widespread indications of severe declines in abundance and distribution and of local extirpations, along with recent evidence of genetically distinct subpopulations, suggest that the species has a high vulnerability to exploitation and is threatened with extinction. The available information therefore indicates that the species meets the biological criteria for an Appendix I listing. A similar conclusion was reached by FAO (2007) when assessing the proposal for listing all species of Pristidae in Appendix I.

Currently, the main threats to the species seem to be related to incidental takes in commercial and recreational fisheries and to habitat degradation. Mortality due to habitat degradation would not be affected by a CITES listing.

Any trade in freshwater sawfish products is already prohibited by CITES because the current Appendix II listing only allows the export of live specimens for the aquarium trade under specified circumstances. Retaining live specimens of all species listed under Appendix I could facilitate the implementation of CITES regulations, as identification to the species level would no longer be necessary. As noted by FAO (2007), a CITES Appendix I listing would only be effective for the conservation of the species in combination with strengthened national management. Strengthening management measures where these are in force, and implementing management in other areas, addressing all sources of mortality, would be essential to ensure conservation and recovery of the population of freshwater sawfish.

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TABLES AND FIGURES

TABLE 1

Information for assessing productivity level of freshwater sawfish *P. microdon*

Parameter	**Information**	**Productivity**	**Source**
Natural mortality	0.07–0.14 per year (*P. pectinata*, *P. perotteti*)	Low	Simpfendorfer, 2000 (*apud* FAO, 2007)
Age at maturity	8 years	Low–medium	Phillips, 2012
Maximum age	28–80 years	Low	Peverell, 2008 (*apud* Phillips, 2012)

TABLE 2

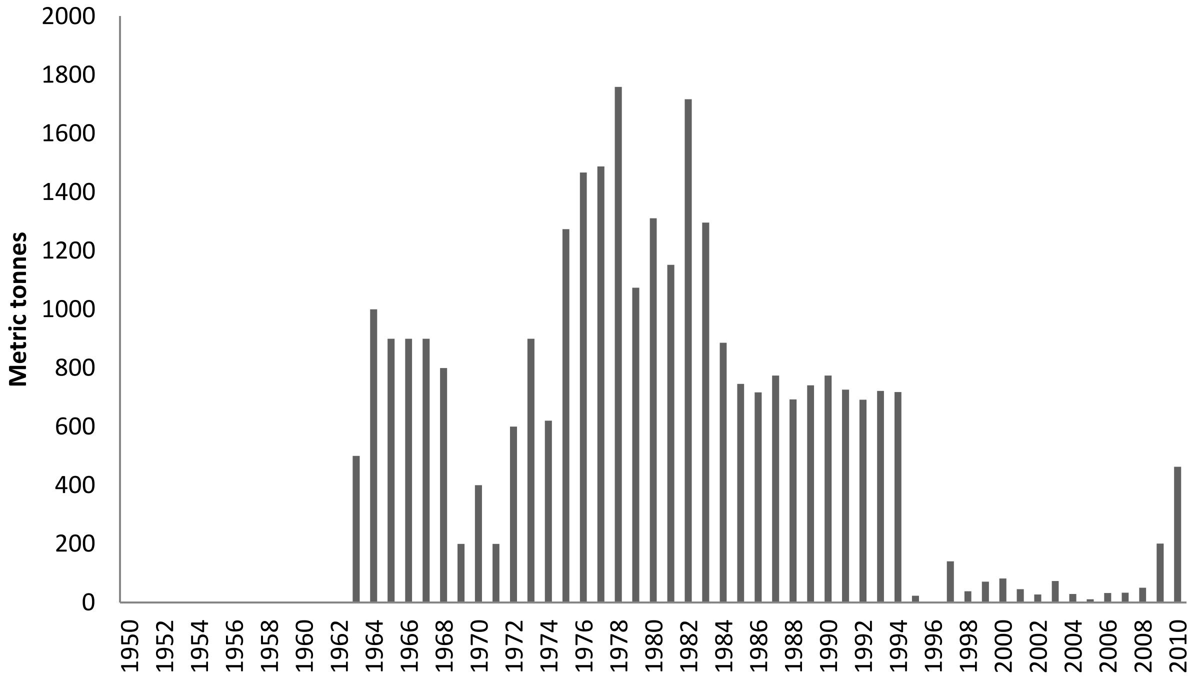
Decline indices for freshwater sawfish

Area	Index	Trend	Basis	Coverage	Reliability	Source
World	Reported catches Pristidae	Decline to 9% of historical	Mean 2005–2010 ca. 132 tonnes/year; mean 1976–1979 ca. 1400 tonnes/ year (maxima)	World reported landings (FAO)	Catch data (2)	Figure 1 proposal; FAO FishStat.
Indo–West Pacific	Reported catches Pristidae	No trend from late 1980s to 2005; increase from 2005 to 2010.	Visual inspection of Figure 2.	Landings from East and West Indian Ocean and West Pacific (FAO)	Catch data (2)	FAO FishStat; Figure 2
Queensland, Australia	Catches Pristidae	Severe decline and disappearance of pristids from late 1960s to 1990s.	Visual inspection, Figure 3	Queensland east coast, Australia	Catch data from Shark Control Programme (2–3)	Stevens, Pillans and Salini (2005)
Queensland, Australia	Bycatch of *P. microdon* in northern prawn fishery (NPF)	Species disappearance from NPF bycatch since 1998.	Species historically caught in NPF. No individuals caught since 1998.	Queensland, Australia	Bycatch data from scientific and fisheries observers (2–3)	DEWHA (2009)
Indonesia	Observations Pristidae	Species have not been recorded for more than 25 years in Indonesian waters	Proposal	Indonesia	Anecdotal impression (1)	White and Kyne (2010) in proposal
Southeast Asia	Catches, observations, *P. microdon*	Greatly reduced, locally extirpated; formerly common	Common in fisheries in 1960s, currently few reported; extirpated from Fly River, New Guinea	Southeast Asia	Catch, observations (1–2)	Compagno *et al*. (2006) in proposal; FAO (2007)
Australia	Observations *P. microdon*	Significant decline (not quantified)	Proposal	Australia	Anecdotal impression (1)	Pillans *et al*. (2009) in proposal
Gulf of Thailand	Trawl surveys, Pristidae	Virtual disappearance of sawfish	Observation based on the analysis of survey data from 1963–1972	Gulf of Thailand	Comparison of data from trawl surveys (5)	Pauly (1979, 1988)
Indonesia, New Guinea	Observations *P. microdon*	Demise of *P. microdon* from Lake Sentani, New Guinea	Observation	Indonesia	Anecdotal impression (1)	Polhemus, Englund and Allen (2004)
Cambodia	Observations *P. microdon*	Species not seen for several decades in Cambodian Mekong	Observations	Cambodia	Anecdotal impression (1)	Rainboth (1996)
South Africa	Observations *P. microdon*	Disappearance since 1990s,	No confirmed sightings since 1990s (proposal)	South Africa	Anecdotal impression (1)	Proposal

Source: Reliability indices are based on FAO (2001).

FIGURE 1

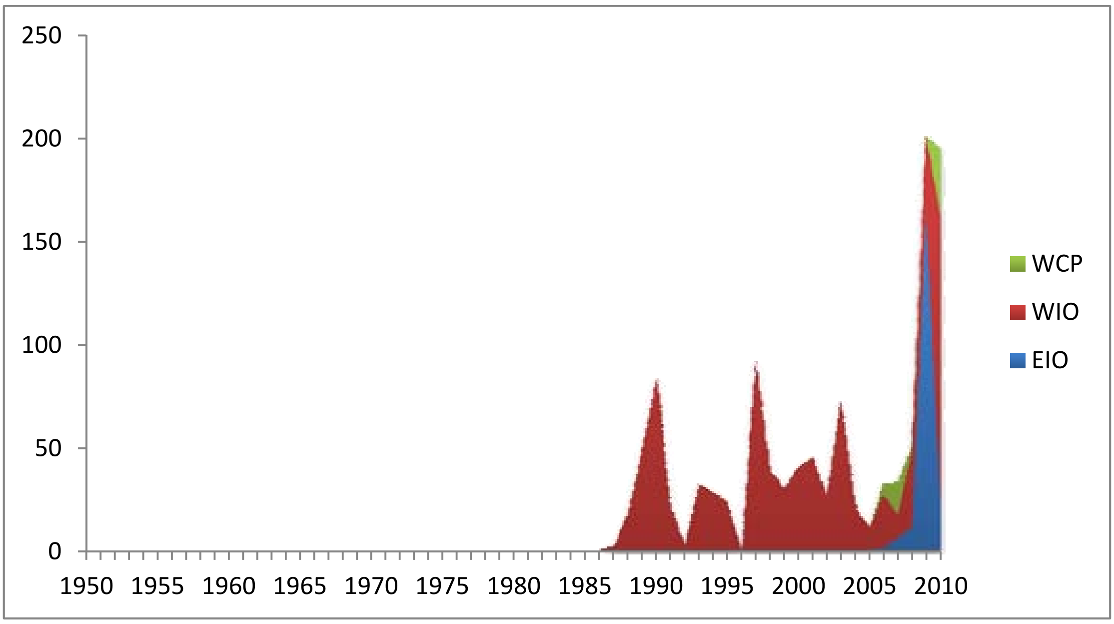
Reported world landings of Pristidae, 1950–2010



Sources: FAO fishery information (2012); proposal.

FIGURE 2

Reported landings of sawfishes (all species) in Indo-West Pacific

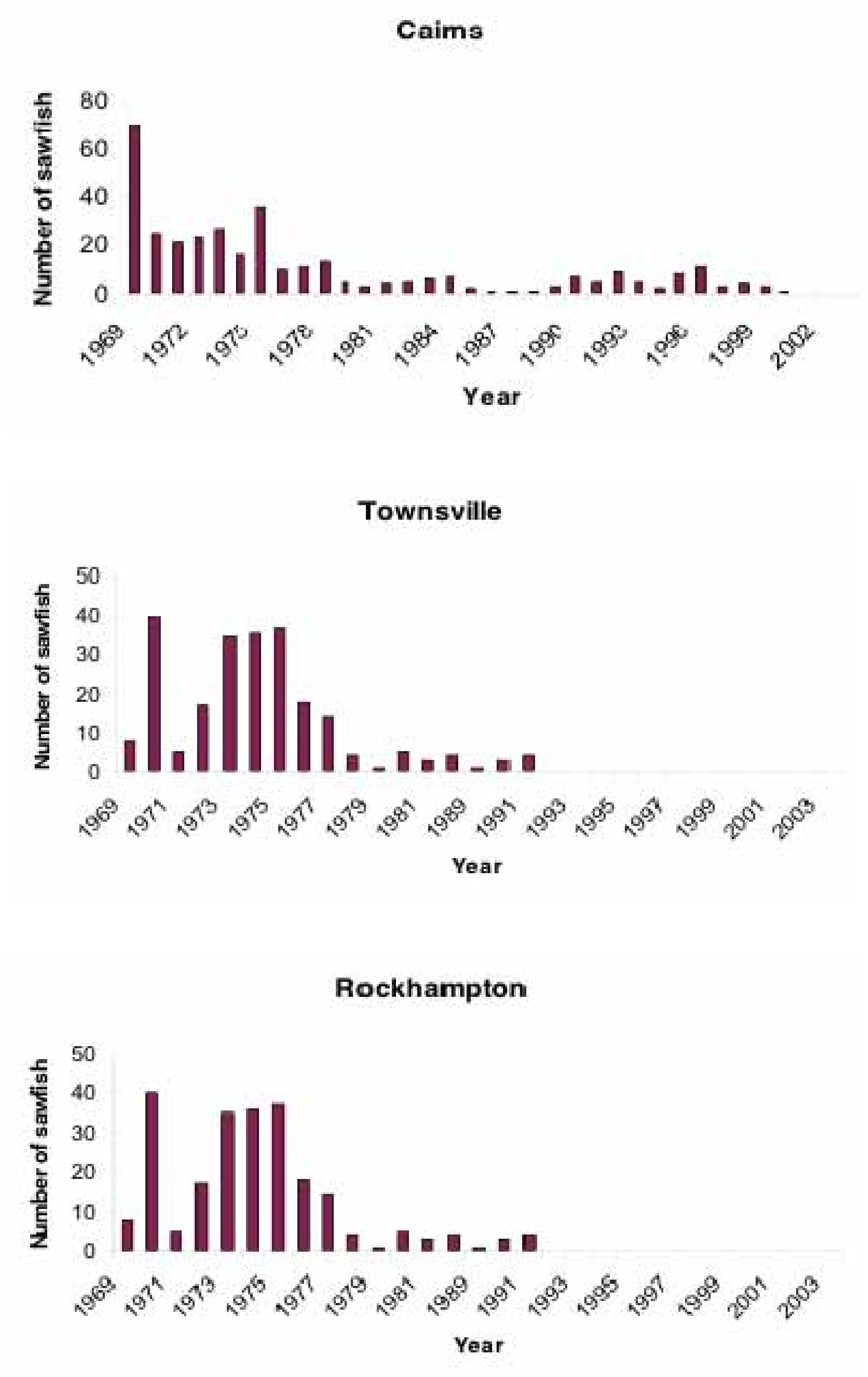


Note: WCP = Western Central Pacific; WIO = Western Indian Ocean; EIO = Eastern Indian Ocean.

Source: FAO FishStat.

FIGURE 3

Catches of unidentified pristids from Cairns, Townsville and Rockhampton in the Queensland Beach Control Program, 1969– 2003



Source: Stevens *et al.* (2005).

FAO Expert Advisory Panel assessment report: manta rays - CoP16 Proposal 46 -

Species:

Genus *Manta*, composed of *M. birostris*, *M. alfredi* and possibly a third species, *Manta c.f. birostris*.

Proposal:

Inclusion of the genus *Manta* in Appendix II in accordance with Article II paragraph 2(a) of the convention and satisfying Criterion A and B in Annex 2a of Resolution Conf. 9.24 (Rev. CoP14).

Basis for proposal:

Annex 2a, Criterion A. "It is known, or can be inferred or projected, that the regulation of trade in the species is necessary to avoid it becoming eligible for inclusion in Appendix I in the near future." The proposal indicates that increasing fishing pressure, driven by international trade in *Manta* gill rakers, has led to significant rates of decline in population sizes in recent years (declines of 56–86 percent in the past 6–8 years). The species in the genus have low productivity and are highly vulnerable to exploitation. According to the proposal, the species are only protected in a few range States and there are no management measures in the range States with the largest documented fisheries. It is also stated that without regulation of international trade, all *Manta* spp. will probably qualify globally for Appendix I listing in the near future.

Annex 2a, Criterion B. "It is known, or can be inferred or projected, that a regulation of trade in the species is required to ensure that the harvest of specimens from the wild is not reducing the wild population to a level at which survival might be threatened by continued harvesting or other influences." The proposal indicates that because of their small and highly fragmented populations, extremely low productivity, and known aggregating behaviour, *Manta* spp. are highly vulnerable to overexploitation. Under current fishing pressure levels, populations will continue to decline, putting the survival of these species at risk.

ASSESSMENT SUMMARY

CITES biological listing criteria

Considering the decline criteria overall and within regions, there is a paucity of reliable information on historical or recent decline of both species of manta. Thus, the Panel was unable to identify reliable information to assess against the decline criteria throughout the range. It also could not comment on the projected trends of the populations as any projections were likely to be speculative. Both species are pan-oceanic in distribution and thus do not qualify under the distribution criterion.

The Panel was unable to assess the situation of the two species against the small population criteria. The abundance of mantas is described in the proposal in terms of aggregation numbers, population numbers and surveys of sightings in an interchangeable manner. These data could not be reasonably integrated to provide an approximate estimate of global population size. Estimates of the population size using life-history characteristics and distribution could not be reconciled with sightings and removals.

Comments on technical aspects of the proposal

Biology and ecology: Manta rays are low-productivity species. The genus *Manta* has recently been split into two species: *Manta alfredi* and *Manta birostris*. The global population size of both species is unknown. Local aggregations are typically estimated as from hundreds to thousands of individuals. *M. birostris* has a circumglobal distribution in tropical, subtropical and temperate waters, while *M. alfredi* is restricted to tropical and subtropical waters. *M. birostris* undergo significant seasonal migrations and are capable of large migrations (> 1 000 km) although movements across ocean basins are presumed rare. *M. alfredi* are more resident to coastal waters, with shorter seasonal migrations. Manta rays are the largest of the rays and both species are planktivores.

Trade: The price of gill rakers is high. The proposal suggests that the value of gill rakers has increased greatly in recent years, leading to an increase in targeted fishing for *Manta* spp. in key range States. No supporting evidence was provided to substantiate these assertions. The lack of commodity codes for the species makes it difficult to verify the extent and trends of the trade in the species products. The current estimates of demand appear to be in the same order of magnitude of catches in the few documented fisheries. The gill raker trade is supplied by both target and bycatch fisheries. These fisheries also supply the domestic meat and international skin market. The Panel concluded that trade is an important driver for the targeted fisheries. In addition, an unknown proportion of the global trade originates from the bycatch in other commercial fisheries.

Fisheries management: Fishery removals are poorly documented. The species are caught in direct fisheries and as bycatch in coastal and offshore fisheries. The proposal suggests that approximately 4 600 individuals are caught annually to supply the trade in gill rakers. Important fishing countries have not adopted specific measures for manta rays, or NPOA-Sharks. Management measures exist, including the banning of the harvesting and/or trade of manta rays in a few range States.

The Panel noted various risk factors for the conservation of manta rays including their low productivity, the seasonal and predictable aggregations, the lack of reliable catch and population information and the lack of management at regional and international levels in most areas.

Likely effectiveness of a CITES listing for the conservation of the species: As there is a proportion of the fishery driven by the international gill raker trade, it is likely that this will be further regulated and monitored if this species is included in Appendix II. The listing would only be effective in addressing concerns about the conservation of the species when combined with strengthened national and international management.

DETAILED PANEL ASSESSMENT

1. Scientific assessment in accordance with CITES biological listing criteria

1.1 Biological aspects

1.1.1 Population assessed

The genus *Manta* is composed of two species, *Manta alfredi* and *Manta birostris*, only recently separated (Marshall, Compagno and Bennett, 2009). The authors also indicated that a third putative species distinct from *M. birostris*, *Manta cf. birostris*, probably exists in the Atlantic but there seems to be not enough evidence to separate this third species. The genus has a circumglobal distribution in tropical, subtropical and temperate waters, with *M. alfredi* restricted to tropical and subtropical waters. The species are commonly encountered along productive coastlines, oceanic islands and offshore seamounts, often aggregating in large numbers to feed, mate, or clean (groomed to remove parasites). Manta rays are mainly planktivorous, but may feed also on small and moderate-sized fishes (Compagno, 1997). *M. birostris* and *M. alfredi* are known to co-occur in some locations, showing distinct habitat uses (Kashiwagi *et al*., 2011). Available satellite tracking data indicate that *M. birostris* undergo significant seasonal migrations and are capable of large migrations (more than 1 100 km); however, movements across ocean basins may be rare (Marshall *et al*., 2011a). *M. alfredi* is more resident to coastal tropical waters, exhibiting shorter seasonal migrations (proposal).

Manta populations seem to be sparsely distributed and highly fragmented within their wide geographical range (Marshall *et al*., 2011a, 2011b; proposal). Evidence of site fidelity to some regions, migratory patterns and the low interchange of tagged individuals among populations appear to support the hypothesis of discrete regional populations. However, this has not been corroborated by genetic studies.

1.1.2 Productivity level

Information on life history parameters for *Manta* spp. is very sparse. The available information (Table 1) indicates that the species fit in low-productivity levels. The genus is considered among the least fecund of all elasmobranchs, bearing one pup one average every two to three years (Couturier *et al*., 2012). There are no available age and growth studies on *Manta* spp. Minimum longevity has been estimated based on the continuous sightings of individuals identified by the ventral markings. There are reported sightings of *M. alfredi* of more than 30 years off Japan (Couturier *et al*., 2012) and Hawaii (Clark, 2010), and of *M. birostris* of up to 20 years (Marshall *et al*., 2011a). Rubin and Kumli (2002) reported an individual seen over a time span of 15 years in the Sea of Cortez, Mexico. Ishihara and Homma (1995) reported a male *M. birostris* of approximately 20 years off Japan. Age at maturity was estimated for male *M. alfredi* in Hawaii at 3–6 years (Clark, 2010). Female *M. alfredi* and *M. birostris* are thought to mature at 8–10 years (Marshall *et al*., 2011a, 2011b). Generation time was estimated by Marshall *et al*. (2011a, 2011b) at 25 years, based on an expected longevity of 40 years and an age of female first maturity of 10 years. Natural mortality rates were estimated at extremely low values (Table 1; Dulvy, Pardo and Simpfendorfer, forthcoming). The inferred low natural mortality, in addition to the habit of aggregating at certain times and their very small numbers observed in the aggregations (from hundreds to thousands), makes the species highly vulnerable. Any increase in fishing mortality would probably have a profound effect on the population.

1.1.3 Anthropogenic sources of mortality

Fisheries catches are likely to be greatest sources of anthropogenic mortality to manta rays. Their large size, aggregative behaviour, predictable habitat use, lack of human avoidance and desirable products (meat, gill rakers) make them an easy and profitable target for exploitation. The species are caught in directed fisheries and also as bycatch (Heinrich *et al*., 2011; Courtier *et al*., 2012). However, the relative importance of directed and incidental catches is unknown.

Targeted fisheries for mantas as well as other Mobulidae have been reported throughout the species range, being conducted by small-scale and multispecies fisheries using several types of gear, including harpoons, gillnets and trawl nets (proposal; Alava *et al*., 1997; Mohanraj *et al*., 2009; Heinrichs *et al*., 2011; Rayos, Santos and Barut, 2012). The species are valued for their meat, skin, cartilage and gill rakers; the latter used in traditional Chinese medicine, fetching a high price in trade (see Trade aspects). Indonesia, Sri Lanka, and India appear to have the largest catches, but targeted fisheries have also been reported in Peru, Mexico, China, Mozambique and Ghana (Heinrichs *et al*., 2011). According to Heinrichs *et al*. (2011), while the fisheries for meat have been conducted for centuries by some communities, the growing demand for gill rakers and the decrease in abundance of other elasmobranch resources have significantly increased fishing effort of the targeted fisheries in the last decade. However, there are no time series of effort or catch data available to evaluate possible changes in fishing pressure or CPUE.

The species are also known to be taken as bycatch in gillnets, longlines, and purse seine fisheries, but catches are poorly documented (proposal; White, Giles and Potter, 2006; Camhi *et al*., 2009). The bulk of the catches reported to FAO are in a generic category “Mantas, devil rays nei” that also includes Mobulidae species. The only reported catches of giant manta were by Ecuador: 5 tonnes in 2007 and 10 tonnes in 2008. Based on data from documented fisheries, Heinrichs *et al*. (2011) estimated that, on average, 3 409 individuals are taken annually in targeted and bycatch fisheries. The numbers taken globally are expected to be much higher.

Manta rays are also potentially threatened by the degradation of coral reefs, interaction with marine debris, marine pollution and boat strikes, but the relative importance of these other sources of anthropogenic mortality are unknown but probably small compared with the effect of fisheries (Heinrichs *et al.*, 2011).

1.1.4 Population status and trends

Population size

No estimates of total population abundance are available. The species are believed to be sparsely distributed, with small subpopulations in the range of 100–6 000 individuals (Heinrichs *et al.*, 2011; Marshall *et al*., 2011a, 2011b). The size of some regional aggregations is reported in the proposal and in Heinrichs *et al.* (2011). Total numbers from all monitored sites would be about 10 000 *M. alfredi*, 2 200 *M. birostris* and > 70 *M. c.f. birostris*. These probably considerably underestimate the total numbers of mantas considering that they comprise only part of the known aggregation sites (25 others have been identified according to the proposal) and only 3 of 62 range States for *M. birostris* and 7 of 39 for *M. alfredi*, according to the data provided in Annex IV of the proposal). In addition to those locations noted above, other locations where the species occur are also likely given their circumglobal distribution.

Area of distribution

No estimate of extent of distribution is available but, considering that both known species are circumglobal, they can be considered to have a very large distribution.

Population trend

Information presented in the proposal regarding population trends from different oceanic regions is summarized below and in Table 2.

Pacific Ocean

Homma *et al.* (1999) claim that there have been population collapses as a result of overfishing along the Pacific coast of Mexico, although no data are presented. Dive sightings data from the Islas Revillagigedos, off the Pacific coast of Mexico, identified 127 *M. birostris* over 20 years (Rubin and Kumli, 2002), some of them re-sighted for 10 years or more. These data suggest that mantas are still present in the Pacific coast of Mexico.

Homma *et al*. (1999) also reported changes in the school sizes of manta rays as observed by a diver off Okinawa Island, Japan. Schools of 50 individuals were observed in 1980, 30 in 1990 and 14–15 individuals in 1997. Homma *et al*. (1999) also commented that this local decline in school size may not be as result of population decline as “young rays and pregnant females also make up the school”. According to the proposal, these data indicate a decline of 70 percent in the population off Japan in 17 years. Without information on the methodology, sampling effort applied in these different periods, or how these observed individuals relate to the total population off Japan, it is difficult to evaluate the reliability of the data or the inferred decline

Indo-Pacific

Declining trends are estimated based on differences in landings of target fisheries in Lamakera and Lombok, Indonesia, and in the Philippines. Sightings data from dive operators are also used to infer trends. Neither source is considered to be a reliable indicator of changes in abundance.

Dewar (2002) conducted an exploratory survey in the traditional whaling and manta ray fishing villages of Lamalera and Lamakera, in the Alor region of eastern Indonesia. In Lamakera, the author reported a recent change in fishing practices and uses of *M. birostris*, apparently driven by trade in skin and gill rakers that had resulted in an increase in fishing effort and catches by an order of magnitude in just a few years. The most important changes observed were the replacement of traditional whaling vessels for smaller vessels powered by 15 hp outboard motors that reduced transit time to fishing ground and the increase in the number of boats from 18 to more than 30. As a result, catches increased from 200–300 individuals per year to an estimated average take of 1 500 manta rays per year (range from 1 050 to 2 400). Dewar (2002) does not provide enough information about the methods and number of interviews to understand the level of reliability of the estimates.

In the proposal, the estimated takes by Dewar (2002) are compared with average catch of about 650 mantas in the same village in 2010 (Setiasih, forthcoming) to infer the rate of decline of the local population. If landings were an indicator of population abundance, the above data would imply a population decline of 37–72.5 percent in the last decade. However, the methodology and analyses used by these studies were found deficient by the Panel.

The proposal compares estimated annual landings of 143 *M. birostris* in a target fishery in Lombok from 2007 to 2012 (Setiasih., forthcoming) to estimated annual landings of 331 individuals caught as bycatch in a skipjack tuna gillnet fishery in 2001–05 (White, Giles and Potter, 2006). These data are used in the proposal to infer a 57 percent decline in the population in 6–7 years. The Panel has concerns about the methodology in the Setiasih. (forthcoming) study and noted that there was no attempt in the proposal to standardize across the different fisheries (target and bycatch) in these studies.

For the area studied by Setiasih. (forthcoming), official catch statistics do show a decline in catch in recent years (Figure 1). However, this trend is not consistent across all of Indonesia.

Alava *et al.* (2002) evaluated the trends in landings of the directed fisheries for whale shark and mobulid rays in the Bohol Sea, Philippines, based on interviews with fishers and visits to landings sites between 1993 and 1995. With regards to manta rays, the authors noted that: “Fishers’ memory of the historical catch of mantas and/ or devilfishes was vague. Fishers had problems segregating catches into species, often confusing mantas and devilfishes. Most often, catches were underestimated. Fishers feared that the Bureau of Internal Revenues would investigate them if they reported higher catches”. Despite such limitations, the reported number of manta rays caught in Pamilacan Island, Blacayon, in the 1960s (100 individuals) was two times higher than the value in 1996 (50 individuals) (Alava *et al*., 2002). This difference in reported landings appears to be used in the proposal to indicate a decline of 50 percent in *Manta* spp. in the region. In contrast, an analysis of the perception of fishers in the locality indicated that 90 percent believed that catches in the 1995 season were the same or higher than previous years (Alava *et al*., 2002).

Marshall *et al.* (2011) reported that 156 manta rays were caught in the Philippines in 2002–03 (no location provided) and that the species is currently rare around the Bohol Sea. A recent study in various landing sites in the Bohol Sea (Rayos, Santos and Barut, 2012) indicated a decline in landings of *M. birostris* from 14 individuals in 2002 to 3 individuals in 2010 (78 percent decline). However, the authors noted that while catches in 2002 originated from fishing boats of Pamilacan Island, these fishers did not fish for rays in the area in 2010. The manta rays found in the markets were apparently bought from fishers from other localities (Rayos, Santos and Barut, 2012).

Since 1998, there has been a ban on taking or catching, selling, purchasing and processing, transporting and exporting of whale sharks and manta rays in the Philippines (Fisheries Administrative Order 193). The ban was lifted in 1999 and apparently re-established after 2002 (Marshall *et al*., 2011; Couturier *et al*., 2012). The difficulty in obtaining reliable catch data for a species that is legally protected should therefore be weighed in the analysis of trends in landings.

The other information used to infer trends in abundance are reports of manta ray sightings by dive operators. One such report suggests that the local population of *Manta* spp. in the Sulu Sea off Palawan Island (Philippines) has fallen by one-half to two-thirds in seven years since the end of the 1980s (proposal).

Indian Ocean

Two main sources of data are used to infer trends in population abundance in the Indian Ocean: changes in fisheries landings; and anecdotal information about sightings by divers, dive operators, fishers and researchers.

In Sri Lanka, Fernando and Stevens (2011) estimated an annual catch of 1 055 *M. birostris* in 2011, based on a survey of the Negombo and Mirissa fish markets. There are no available landings data from previous periods to compare with this data. However, the authors report that the majority of fishers interviewed indicated a decrease in catches in the last five years. Another study by Anderson *et al*. (2010), cited in the proposal as showing a declining trend in manta catches in Sri Lanka, could not be obtained.

In Chennai, India, *M. birostris* was one of the elasmobranch species caught in the multispecies gillnet, trawling and hook and line fisheries between 2002 and 2006 (Mohanraj *et al*., 2009). Annual catches were zero in 2002 and 2003, 12.3 tonnes in 2004, zero in 2005, and 6 tonnes in 2006. The total landings of rays in the period declined from 1 538 tonnes/year to 520 tonnes/year (67 percent decline) (Mohanraj *et al*., 2009). In Mumbai, the catch rate of rays by trawlers decreased by 63 percent in 14 years, from 0.65 kg/h in 1990 to 0.24 kg/h in 2004 (Raje and Zacharia, 2009). The decline in catches of rays and mobulid rays from these studies are used in the proposal to infer declines in manta rays. However, the Panel noted that mantas appear to be a minor component of the catches in both studies (e.g. 0.3 percent in Chennai; Mohanraj *et al*., 2009), and concluded that the decline inferred cannot be reliably used to assess the proposal.

In Mozambique, where a target fishery exists for *M. alfredi* (Marshall *et al*., 2011b), Rohner *et al*. (forthcoming) reported an 86 percent decline in the sightings of *M. alfredi* in a period of 8 years (2003–2011). In the same period, the abundance of *M. birostris* appears to have remained stable.

Personal communications are used in the proposal to claim declines in abundance in the Indian Ocean, including a reported 76 percent decline in *Manta* spp. sightings by dive operators in the Similan Islands, Thailand, from 2006–07 (59 individuals) to 2011–12 (14 individuals).

Overall and across all regions, there is a paucity of reliable information on relative decline, either historical or recent, of both species of manta.

1.2 Assessment relative to quantitative criteria

1.2.1 Small population

There are no global estimates of population numbers of the two species. The argument put forward by the proposal that the species have “small populations with small, highly fragmented, and isolated subpopulations, preventing recruitment and recovery following declines” is not supported by the available data. Numbers of individuals in the few monitored aggregation sites of both species vary in the range from 100 to 6 000 individuals (Heinrichs *et al*., 2011; Marshall *et al*., 2011a, 2011b), but there is no genetic evidence that these represent isolated subpopulations. Instead, data indicate that there is little interchange between regional populations (Couturier *et al*., 2012). *M. alfredi* is more resident than *M. birostris*, preferring inshore coastal waters and undertaking relatively short migrations. *M. birostris* is more oceanic, performing extensive migrations (Couturier *et al*., 2012).

Total numbers from all monitored sites would be about 10 000 *M. alfredi*, 2 200 *M. birostris* and > 70 *M. c.f. birostris*. These are underestimates of the total numbers considering that they cover only part of known aggregation sites (25 additional sites have been identified according to the proposal) and part of the range States (3 out 62 for *M. birostris* and 7 out of 39 for *M. alfredi*, according to data presented in Annex IV of the proposal). In addition, these sites probably represent only a portion of the known population for these circumglobal species. The Panel was therefore unable to assess manta rays against the criterion on small population size.

1.2.2 Restricted distribution

No guidelines for restricted area of distribution are provided in the CITES criteria, which indicate that thresholds should be taxon-specific (Conf. Res. 9.24 Rev. CoP14). FAO (2001) recommended that historical extent of decline in area of distribution would be a better measure of extinction risk than absolute value of distributional area, but that if no other suitable information is available and absolute area of distribution has to be used for an exploited fish population, analyses should be on a case-by-case basis as no numeric guideline is universally applicable.

No estimate of distribution area is available but, considering that both known species are circumglobal, they can be considered to have a very large distribution. In addition, there is no evidence of decline in the area of distribution of the species.

1.2.3 Decline

Under the CITES criteria for commercially exploited aquatic species (Conf. Res. 9.24 Rev. CoP15), a decline to 15–20 percent of the historical baseline for a low-productivity species might justify consideration for Appendix I. For listing on Appendix II, being "near" this level might justify consideration; "near" for a low-productivity species being 20–30 percent of the historical abundance level (15–20 percent + 5–10 percent).

Also according to the CITES guidelines (Conf. Res. 9.24 Rev. CoP15), "in circumstances where information to estimate the extent of decline is limited, the rate of decline over a recent period could itself still provide some information on the extent of decline. For listing in Appendix II, the historical extent of decline and the recent rate of decline should be considered in conjunction with one another. The higher the historical extent of decline, and the lower the productivity of the species, the more important a given recent rate of decline is. A general guideline for a marked recent rate of decline is the rate of decline that would drive a population down within approximately a 10-year period from the current population level to the historical extent of decline guideline (i.e. 5–20 % of baseline for exploited fish species). There should rarely be a need for concern for populations that have exhibited an historical extent of decline of less than 50 %, unless the recent rate of decline has been extremely high".

No overall population decline index is available for comparison with the guidelines. There is a paucity of reliable information on historical or recent decline of both species of manta for evaluation against decline criteria.

In the Pacific, no reliable information was presented to support the claim that the population *M. alfredi* has declined by 70 percent off Japan. Moreover, the suggested population collapse of *M. birostris* in the Pacific coast of Mexico could not be verified; more recent sources confirm the presence of the species in the area.

In the Indo-Pacific, the methodologies and analyses used in the studies by Dewar (2002) and Setiasih *et al*. (forthcoming) were considered deficient by the Panel to infer population declines. For the area studied by Setiasih *et al*. (forthcoming) in Indonesia, official catch statistics do show a decline in catch in recent years; however, this trend is not consistent across all of Indonesia. In the Bohol Sea, the Philippines, a historical extent of decline of 50 percent (estimated based on Alava *et al*. [2002]) and a recent rate of decline of 78 percent (estimated based on Rayos, Santos and Barut [2012]) received a low reliability because of uncertainties in the catch data used as index of population abundance.

In the Indian Ocean, declines in catches and catch rates of rays and mobulid rays were used in the proposal to infer declines in manta rays. However, the Panel noted that mantas appear to be a minor component of the catches in both studies and concluded that the decline inferred cannot be reliably use to assess the proposal. Dive sightings of manta rays off Mozambique indicate recent rates of decline of 76 percent of *M. alfredi*, which is consistent with a decline criterion for Appendix II listing. Data from the same location indicate that the population of *M. birostris* is stable.

Were trends due to natural fluctuations?

There is no evidence that observed trends were due to natural fluctuations.

2. Comments on technical aspects in relation to trade, management and implementation issues

2.1 Trade aspects

Meat, skin, cartilage and the gill rakers are the main products in trade. While the meat is used mainly locally, for human consumption, shark bait or animal feed, the high-value products (skin, cartilage and gill rakers) are mainly traded to other cities or exported (proposal; Dewar, 2002). The skin is used for making shoes and wallets, the cartilage as food supplement, and the gill rakers as traditional medicine in Asia (Dewar, 2002; Heinrichs *et al*., 2011). According to the proposal, the increasing international demand for the gill rakers in China, has been a major driver for the intensification of the targeted fisheries in the last decade. The Panel could not verify this claim, and the lack of commodity codes for the species also makes it impossible to verify the extent and trends of the trade in the species products.

The high value of the gill rakers is one indication of the high demand in the market. Dried gill rakers of large manta rays receive an average price in Guanghzou, China, of USD250/kg (Hilton, 2011; Heinrichs cited in Heinrichs *et al*. [2011]; but not available to the Panel). The dried gill rakers of one manta gives an average of USD140 to fishers (Dewar, 2002). White, Giles and Potter (2006) report the information received from a buyer in Tanjung Luar, Indonesia, who paid approximately USD545 for three adult manta rays and received USD490 for the gill rakers alone. In Sri Lanka, the dried gill rakers are sold by dealers at prices ranging from about USD90/kg to about USD136/kg, and may reach USD228/kg for exceptionally large gill rakers (Fernando and Stevens, 2011).

Based on a market survey conducted in the main centres for the Chinese dried seafood trade (Singapore, China, Hong Kong SAR, China, Macao SAR, Taiwan Province of China and Guangzhou) Hilton (cited in Heinrichs *et al*. [2011]) estimated that an average of 61 000 kg of gill rakers are traded annually, with an estimated 30 percent coming from *M. birostris*. DNA testing applied to market samples found no *M. alfredi* gill rakers at the market. From the weight of gill rakers in trade, it is estimated that approximately 4 652 manta rays would be needed annually to supply the market (proposal). In turn, the total estimated catches in the fisheries documented by Heinrichs *et al*. (2011) is in the order of 3 409 mantas per year. The number of manta rays supplying the gill raker trade is in the same order of magnitude as the catches from the documented fisheries. The fact that *M. alfredi* was absent from market samples should be noted because the species is probably the most abundant and more commonly targeted in tropical and subtropical coastal areas (Couturier *et al*., 2012).

2.2 Fisheries management aspects

Recent compilations of national legislations indicated that some range States have specific laws prohibiting the catch and/or trade of manta rays or protecting the species in marine reserves (proposal; Heinrichs *et al*., 2011; Couturier *et al*., 2012), including States with known important fisheries such as Indonesia, Maldives, Mexico, and the Philippines. The majority of the laws have been enacted within the last decade. Of the top three *Manta* spp. fishing countries (Indonesia, Sri Lanka, and India), which account for 90 percent of estimated catches in all documented fisheries (Heinrichs *et al*., 2011), only Indonesia has adopted specific management measures to regulate the harvest of the species (a 17 760 square mile shark sanctuary in Raja Ampat, declared in 2010, for the protection of sharks, manta rays, mobulas, dugongs and turtles). The country has also adopted an NPOA-Sharks (Fischer *et al*., 2012), providing some political visibility to elasmobranch conservation and sustainable use. According to the last assessment of the IPOA-Sharks by FAO (Fischer *et al*., 2012), Sri Lanka and India have not yet adopted an NPOA-Sharks. As noted by Couturier *et al*. (2012), and also by the proposal, in some cases, legislation for manta rays specifically refers to *M. birostris*, because laws were enacted before the separation of the genus in two species. The revision of legislation will be required in countries were the two species coexist and are targeted by fisheries, such as in the Philippines, to make the law applicable to both species.

M. birostris is known to occur in tropical tuna purse seine, longline and gillnet fisheries (proposal; White, Giles and Potter, 2006; Poisson *et al*., 2012). However, the relative importance of incidental catch in these fisheries compared with direct takes is unknown. Given the lack of manta ray bycatch mitigation measures in tuna RFMOs, the incidental takes of manta rays, especially *M. birostris*, in tuna fisheries regulated by these organizations is currently uncontrolled.

M. birostris was listed in Appendices I and II of the CMS in 2011, meaning that signatory countries agreed to work individually or collectively to protect the species from factors affecting their conservation. Specific agreements or memoranda of understanding concerning the species have not yet been adopted. The recently created CMS Memorandum of Understanding on Sharks does not include *M. birostris*.

2.3 Implementation issues

2.3.1 Introduction from the sea

Of the two species, *M. birostris* is known to occur in the open ocean and in marine environments not under the jurisdiction of any State. Introduction from the sea (i.e. transport of captured specimens from international waters to areas under national jurisdiction) would be expected to occur, at least in fisheries regulated by tuna RFMOs.

Under CITES, such transport of specimens listed on Appendix II would require a certificate from the State to whose jurisdiction the specimens were brought, including an NDF. Exactly how these certification processes would be carried out is still a matter of debate within CITES. Some level of involvement of RFMOs is expected in areas where such organizations have been established with a mandate over shark fisheries.

2.3.2 Basis for findings: legally obtained, non-detrimental

Export permits for Appendix II species must be accompanied by a certificate attesting that the specimens were legally obtained and by an NDF showing that exports are consistent with sustainable harvesting. The majority of the range States have no specific laws concerning manta rays. Few range States have adopted bans on the fisheries and/or trade of the species. Certifying that specimens were legally obtained is not expected to be a problem in these two extreme cases (in some cases, adjustments in national legislations will be needed to account for the recent taxonomic changes). However, jurisdictions that have banned fisheries in specific protected areas will probably face difficulties in identifying the origin of specimens caught, especially in places with weak fisheries monitoring capacity.

Development of an NDF requires appropriate scientific capacity, biological information on the species, and a framework for demonstrating that exports are based on sustainable harvests. Biological information about manta rays is weak in all parts of their range. Substantial improvements in the knowledge base would be required to make scientifically sound NDFs. Methods adapted to data-poor situations will have to be developed and employed in most cases. In this regard, population monitoring by dive sightings seems to be a generally accepted methodology and could be employed to evaluate the effect of harvesting on population numbers in aggregation sites. Some guidance, resources and tools are available to inform countries on the necessary information and steps to be taken in the making of NDFs for fisheries resources (FAO, 2004; Rosser and Haywood, 2002; Anonymous, 2008).

2.3.3 Identification of products in trade

According to the proposal, "*Manta* spp. are often confused with rays of the genus *Mobula*, also in family Mobulidae (Mobulids). The nine species in genus *Mobula* vary widely in body size and geographic distribution (Couturier *et al*., 2012). Fisheries for *Mobula* spp. generally occur in the same locations as for *Manta* spp., in most cases with larger numbers of *Mobula* spp. landed (Fernando and Stevens, forthcoming; White, Giles and Potter, 2006). *Mobula* rays are also targeted for the international trade of their gill rakers, and the trade names, “fish gills” or “peng yu sai”, are used to refer to gill rakers from both genera (Heinrichs *et al*., 2011)”.

The size of the dried gill rakers is an important determinant of price and seems also to be an important aspect considered in separating gill rakers of *M. birostris* from other mobulids. In the market survey described by Heinrichs *et al*. (2011), large gill rakers were associated to *M. birostris* while medium size gill rakers were associated to *Mobula tarapacana* or to juvenile *M. birostris*. Small gill rakers were associated to other mobulid species.

Despite the availability of guides for the identification of gill rakers and live specimens of mobulid from some regions (a guide for the Indo-Pacific is included in the proposal), differentiation of gill rakers from manta and mobula species could be a challenge for customs officers without an identification guide.

Furthermore, if the species are listed in Appendix II, it will be important to develop species-specific commodity codes for meat and other products (gill rakers, skin and cartilage) in order to monitor the origin of products in trade.

2.3.4 “Look-alike” issues

Considering the potential problems related to the identification of products in trade, it is expected that the listing of *Manta* spp. in Appendix II of CITES will potentially create justification for listing other mobulid species with similar products in international trade.

2.4 Likely effectiveness of a CITES listing for the conservation of the species

As there is a proportion of the fishery driven by the international gill raker trade, it is likely that this will be further regulated and monitored if this species in included in Appendix II. The listing would only be effective in addressing concerns about the conservation of the species when combined with strengthened national and international management.

3. Conclusion

Manta rays are low-productivity species. The global population size of both of the manta species is unknown. Local aggregations are typically estimated as from hundreds to thousands of individuals. The proposal suggests that approximately 4 600 individuals are caught annually to supply the trade in gill rakers.

The proposal suggests that the value of gill rakers has increased considerably in recent years, and concern was expressed that this may drive an increase in targeted fishing for *Manta* spp., predominantly *M. birostris*, in key range States. However, no supporting evidence was provided to substantiate these assertions.

Considering the decline criteria overall and across all regions, there is a paucity of reliable information on historical or recent decline of both species of manta. The Panel was thus unable to find reliable information to assess against the decline criteria throughout the range. It also could not comment on the projected trends of the populations.

There is a wide distribution of both species.

Aggregation numbers, population numbers and sighting surveys appear to be used interchangeably in the proposal, and these data cannot be reasonably integrated to provide an approximate estimate of global population size. Thus, the Panel was unable to assess against the small population criterion.

Fishery removals are poorly documented. The species are caught in direct fisheries and as bycatch in coastal and offshore fisheries. Recent management measures exist, banning the harvesting and/or trade of manta rays in few range States. However, important fishing countries have neither adopted any specific measures for manta rays nor adopted NPOA-Sharks.

The Panel notes various risk factors for the conservation of manta rays including their low productivity, the seasonal and predictable aggregations, the lack of reliable catch and population information, lacking management at the regional and international levels in many areas.

Manta rays are harvested for local consumption and also to supply international markets for skin, cartilage and, especially, gill rakers. Gill rakers achieve a high price, and the proposal suggests that this is related to demand. Although the lack of commodity codes for the species makes it impossible to verify the extent and trends of the trade in the species products, the current amount internationally traded appears to be similar to the estimated catches in the few documented fisheries. It is therefore reasonable to conclude that trade is an important driver for the targeted fisheries. Moreover, an unknown proportion of the global catches originate from the bycatch in other commercial fisheries.

Both targeted and bycatch fisheries supply the domestic meat and international gill and skin market. There is a proportion of the fishery driven by the gill raker trade, and it is likely that this will be further regulated and monitored if this species is included in Appendix II. The listing would only be effective in addressing concerns about the conservation of the species when combined with strengthened national and international management.

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TABLES AND FIGURES

TABLE 1

Information for assessing productivity of manta rays

Parameter	Information	Productivity	Source
Longevity	*Manta* spp.: 20–40 years	Low to medium	Marshall *et al*., 2011a, 2011b
Age at maturity	*M. alfredi*, male, Hawaii: 3–6 years	Medium	Clark, 2010
	M. alfredi, female: 8–10 years	Low	Marshall *et al*., 2011b
Natural mortality	0.002–0.004/year	Low	Dulvy, Pardo and Simpfendorfer, forthcoming
Generation time	*Manta* spp.: 25 years	Low	Marshall *et al*., 2011a, 2011b

TABLE 2

Decline indices for manta rays

Area	Index	Trend	Basis	Coverage	Reliability	Source
Pacific Ocean	Diver sightings, likely *M. alfredi*	EOD 70% (?)	*M. alfredi* sightings: school size 50 in 1980; 30 in 1990 and 14–15 in 1997	Okinawa, Japan	Visual observations without sampling effort data (0–1)	Homma *et al*. (1999)
	Landings data (?) and diver sightings.	Suggested collapse followed by apparent recovery.	Reported collapse of the fishery in the 1980s. Diver sightings of 127 individuals off Pacific Mexico coast	Sea of Cortez and Islas Revillagigedos, Pacific coast of Mexico	Visual observations without sampling effort data (1)	Homma *et al*. (1999); Rubin and Kumli (2002)
Indo-Pacific	Landings data *M. birostris*	RRD of 37–72.5% (annual rates 4.5–12%)	Comparison landings 660 individuals in 2010 to 1 050 and 2 400 ind. in 2002.	Lamakera, Indonesia	Landings data (2).	Dewar (2002); Setiasih *et al*., forthcoming; Heinrichs *et al*. (2011)
	Landings data *M. birostris*	RRD 57% (annual rate of 11.5%)	Comparison landings 143 *M. birostris* in 2007–2011 to 331 ind. in 2001–05.	Lombok, Indonesia	Inferred landings data from different fisheries (1).	White *et al*. (2006); Setiasih *et al*. (forthcoming)
	Landings data *Manta* spp.	EOD 50%	EOD inferred by proponent comparing landings of 100 ind. in the 1960s to 50 ind. in 1996	Bohol Sea, Philippines	Landings data derived from interviews (2-).	Proposal using Alava *et al*. (2002)
	Fishers' perceptions	No change or increasing	Perception of fishers regarding current and past manta ray fishing seasons	Bohol Sea, Philippines	Scientifically-designed, structured interviews (3)	Alava *et al*. (2002)
	Landings data	RRD 78% (annual rate of 17%)	Comparison landings 14 ind. in 2002 and 3 ind. in 2010	Bohol Sea, Philippines	Landings data without information on effort (2-).	Rayos, Santos and Barut (2012)

Area	Index	Trend	Basis	Coverage	Reliability	Source
	Sightings dive operators	EOD 50–66%	Decline in dive sightings in seven years from the end of 1980s	Sulu Sea, Philippines	Anecdotal impressions (1)	Proposal
Indian ocean	Interviews	Decline	Decline in catches in the past 5–10 years	Sri Lanka	Interviews at landings sites/markets (2-).	Fernando and Stevens (2011); Anderson et al. 2010 (cited in the proposal).
	Landings of all ray species	RRD 67%	Decline in total landings of all rays from 1 538.3 tonnes in 2002 to 520.6 tonnes in 2006	Chennai, India	Landings data (aggregated) without information on effort (1).	Mohanraj et al. (2009)
	CPUE rays in trawlers	EOD 63%	Decline in catch rates for all rays of trawler from 0.65 kg/h in 1990 to 0.24 kg/h in 2004.	Mumbai, India	Unstandardized CPUE data from ray fishery (1)	Raje and Zacharia (2009)
	Sightings *M. alfredi*	EOD 86%	Decline in sightings of *M. alfredi* from 2003 to 2011.	Mozambique	Scientifically-designed, dive sightings (3)	Rohner et al. (forthcoming)
	Sightings *M. birostris*	No change	No change in sightings of *M. birostris* from 2003 to 2011.	Mozambique	Scientifically-designed, dive sightings (3)	Rohner et al. (forthcoming)
	Sightings by dive operators	RRD 76% (annual rate of 25%)	Decline in *Manta* spp. Sightings between 2006–07 (59 ind.) and 2011–12 (14 ind.).	Similan Islands, Thailand	Anecdotal impressions (1)	Proposal

Notes: EOD = historical extent of decline; RRD = recent rate of decline.

Source: Reliability indices based on FAO (2001).

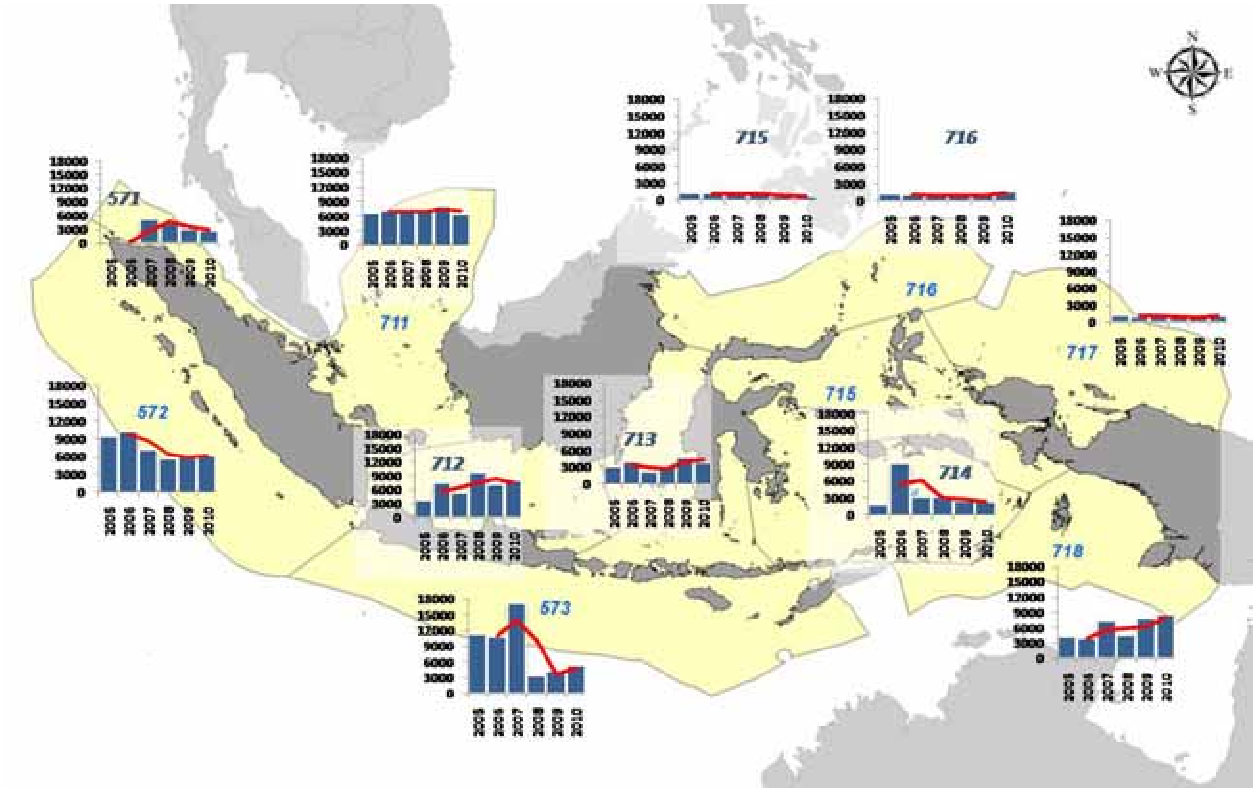
FAO. 2013. *Report of the fourth FAO Expert Advisory Panel for the Assessment of Proposals to Amend Appendices I and II of CITES Concerning Commercially-exploited Aquatic Species, Rome, 3–8 December 2012*. FAO Fisheries and Aquaculture Report No. R1032. Rome, FAO. 161 pp.

CORRIGENDUM

After publication of the report, the FAO Secretariat was informed that Figure 1 of the manta ray assessment (page 119) should be labeled: *Trends in catches of sharks in Indonesia*. In the view of the FAO Secretariat, the Panel's findings and conclusions are not affected by this change.

FIGURE 1

Trend in catches of manta rays in Indonesia



Source: Dharmadi (personal communication).

FAO Expert Advisory Panel assessment report: ceja river stingray - CoP16 Proposal 47 -

Species:

Paratrygon aiereba – ceja river stingray

Proposal:

Inclusion of the ceja river stingray *Paratrygon aiereba* in Appendix II in accordance with Article II paragraph 2(a) and satisfying Criterion B in Annex 2a of Resolution Conf. 9.24 (Rev. CoP15).

Basis for proposal:

Annex 2a, Criterion B. "It is known, or can be inferred or projected, that a regulation of trade in the species is required to ensure that the harvest of specimens from the wild is not reducing the wild population to a level at which survival might be threatened by continued harvesting or other influences." The proposal indicates that the harvesting of ceja river stingray for the ornamental fish trade constitutes one of the main threats to the species and a cause of reduction of populations in the wild. According to the proposal, the inclusion of the species in Appendix II aims to ensure that commercial harvesting will be sustainable. The proposed listing would also contribute to the monitoring and control of the legal activities, reduce illegal trade and support management in the range States.

ASSESSMENT SUMMARY

CITES biological listing criteria

The Panel noted that the supporting statement of the proposal included many unsubstantiated claims, making evaluation difficult. There is no information available to infer population status and trends. Thus, it was not possible to evaluate whether the populations meet the biological criteria for a CITES Appendix II listing under decline. The species is widely distributed (not meeting the restricted area criterion) and the populations are not believed to meet the criterion of a small population.

Comments on technical aspects of the proposal

Biology and ecology: *P. aiereba* is the only species of the genus *Paratrygon*. The species occurs across a large area of the Amazon and Orinoco river basins. It is considered a higher trophic predator, with low fecundity and a large potential maximum size, compared with other freshwater stingrays. *P. aiereba* is a low-to-medium-productivity species.

Trade: The available data indicate that *P. aiereba* is traded internationally for ornamental use and possibly for consumption but the extent of this trade and the effects on the populations are unknown.

Fisheries management: In addition to international trade, the species is also harvested for other purposes, including domestic consumption and removal to reduce local populations to avoid incidents with tourists (population control). The relative importance of these sources of mortality is unknown. Overall, considering that the capture of the species for the ornamental fish trade is prohibited in Brazil and that the number of specimens legally traded from Colombia according to the proposal is very low, it seems unlikely that harvesting for the ornamental fish trade can be considered as a significant cause of any population change.

There are specific regulations to control ornamental harvest and trade in Colombia and Brazil, but there are no specific management measures in other range States. Specific regulations concerning other uses (food, recreational, population control, etc.) appear to be lacking across the region. This factor as well as the existence of illegal cross-border trade and the unregulated fisheries constitute risk factors for the sustainable use of the species.

Likely effectiveness of a CITES listing for the conservation of the species: The Panel did not find any supporting evidence that a CITES Appendix II listing will probably have an impact on the conservation of the species. Strengthening management by range States will be required in order to address properly the existing concerns about the conservation and sustainable use of the species.

DETAILED PANEL ASSESSMENT

1. Scientific assessment in accordance with CITES biological listing criteria

1.1 Biological aspects

1.1.1 Population assessed

Paratrygon aiereba belongs to the family of freshwater stingrays (Potamotrygonidae), the only group of elasmobranchs fully restricted to the freshwater environment (Compagno and Cook, 1995). *P. aiereba* is the only species of the genus *Paratrygon.* The species is common in shallow waters close to river banks, occurring in any type of water, but never in the floodplains. It is distributed in the Amazon and Orinoco river basins, having Colombia, Venezuela (Bolivarian Republic of), Brazil, Ecuador, Peru and Bolivia (Plurinational State of) as range States (proposal). Similar to other freshwater stingrays, *P. aiereba* occupies a high trophic level in the ecosystem, preying on other fishes and aquatic invertebrates (Araujo and Rincon, 2009). The species is reported to feed primarily on fish and at night (Barbarino and Lasso, 2005; Charvet, 2006) but other food items have also been observed (Lasso, Rial and Lasso-Alcala, 1997).

Mejía-Falla *et al*. (2009) reported that the common names for *Paratrygon aiereba* are: raya de río, manta raya de disco, raya manzana and raya ceja. In Brazil, the common names for this species in the main channel of the Amazon River were indicated as: arraia-nari-nari, arraia-aramaçá, arraia-arumaçá, arraiamaramaçá and arraia-vermelha (Charvet-Almeida and Almeida, 2008).

In a recent thorough revision of the data available on the biology, fishery and trade of freshwater stingrays for ornamental purposes in Colombia, it was pointed out that there are few data available for Colombia (Mejía-Falla *et al*., 2009). Nevertheless, their high potential as an economic resource is also indicated and the need of further studies is highlighted (Mejía-Falla *et al*., 2009).

Based on the genetic divergence of the populations from the Xingu, Araguaia, and the Solimões and Amazon-Estuary system in Brazil, Frederico *et al*. (2012) indicated that there is more than one species within what currently is considered *P. aiereba*. The study also revealed that the *P. aiereba* populations are structured within each river, with no or almost non-existent gene flow occurring between rivers.

1.1.2 Productivity level

The biology of *P. aiereba* is poorly known. The species is considered the largest freshwater stingray, with adults reaching more than 60 kg in weight (Charvet-Almeida, Araujo and Almeida, 2005). Maximum sizes reported in different studies cited in the proposal range from 80 cm disc width (DW) to 157 cm DW. It is suggested that *Paratrygon* specimens could attain greater sizes (DW) in larger rivers and in certain streams of the Colombian Orinoquía could be relatively common (Mejía-Falla *et al*., 2009). Sexual maturity was estimated at 40–60 cm DW for males and 72 cm DW for females in Brazil (Charvet-Almeida, Araujo and Almeida, 2005; Charvet-Almeida, 2006) but there seems to be some variation according to each population (Charvet, personal communication). In Venezuela (Bolivarian Republic of), sexual maturity is reached at sizes larger than 41 cm DW for males and at 37 cm DW for females (Lasso, Rial and Lasso-Alcala, 1997). Reproductive information from Venezuela (Bolivarian Republic of) indicates a 2:1 proportion of females, low fecundity (1–8 embryos but an average of 1) and in the llanos, reaching sexual maturity at about 37–50 cm DW (males) and of more than 61 cm DW (females) (Lasso *et al*., 1996; Barbarino and Lasso, 2005). The reproductive mode is matrotrophic viviparity with trophonemata, and the duration of the reproductive cycle is about two years (Araujo and Rincon, 2009). Fecundity can vary from 1 to 8 pups per gestation, but it is usually between 1 and 2 pups (proposal; Charvet-Almeida, Araujo and Almeida, 2005; Araujo and Rincon, 2009).

The productivity of the species was considered by the Panel to be low to medium (Table 1).

1.1.3 Anthropogenic sources of mortality

Among the main sources of threats to the species of Potamotrygonidae are: fish harvesting for different purposes (subsistence/artisanal fisheries for food, artisanal fisheries for ornamental fish, bycatch in other commercial fisheries and recreational fisheries); population control in areas of interest to tourism (owing to fear of injury); and habitat destruction caused by other sectors, including those resulting from the construction of hydroelectric dams, fluvial ports, mining and oil exploration (proposal; Araujo *et al*., 2004a; Junk, Soares and Bayley, 2007; Araujo and Rincon, 2009). The relative importance of these anthropogenic sources of mortality is unknown owing to the lack of accurate data on fisheries takes and information on the status and trends of populations in areas affected by the different threats. The proposal reported that in Colombia overexploitation for commercial and ornamental use was the main threat to the species (Lasso and Sanchez-Duarte, 2012) cited in the proposal); however, this was not supported by the information available to the Panel, and considering the export numbers provided (216 individuals between 2007 and 2011), the Panel concluded that trade is not currently a substantial threat.

In Brazil, the capture of *P. aiereba* for the ornamental fish trade is illegal, but some level of illegal catch and trade with other South American countries is known to occur (Araujo and Rincon, 2009).

P. aiereba is caught for consumption in artisanal and commercial fisheries in Brazil and Venezuela (Bolivarian Republic of) (proposal). The target food fishery in Brazil started in about 2000 (Charvet-Almeida and Almeida, 2008) and has been continuously increasing since then (Figure 1). Different types of gear are used in Brazil, including hook and line, longline and harpoon (Araujo *et al*, 2004; SBEEL, 2005). The bycatch in trawling fisheries has been reported along the Solimões-Amazon River (Araujo *et al*., 2004a). The species is a target of ornamental fisheries in Colombia, Peru and Venezuela (Bolivarian Republic of) (proposal). Scoop nets, tidal traps, snorkel and the “rayero” in Colombia, are common types of gear used in the ornamental fisheries (SBEEL, 2005; Anjos *et al*., 2009).

It is reported that the Venezuelan fisheries of *P. aiereba* as a food source started in 1996, in an artisanal way, as an alternative fishery in periods when high waters (flooding) made it difficult to catch more valuable fishes. The total catches for 1996–2002 were reported as 2.7 tonnes (Barbarino and Lasso, 2005; Mejía-Falla *et al*., 2009).

There are no catch statistics for freshwater stingrays in FAO FishStat. In Brazil, landings are reported at the family level (Potamotrygonidae), and also include daysiatid species (Araujo *et al*., 2004a). The available data show an increasing trend in landings from 2001 to 2010, with production in recent years in the order of 750 tonnes/year (Figure 1). The exact proportion of *P. aiereba* in this total is unknown, but the species is known to be caught in food fisheries in Brazil, along with *Potamotrygon orbignyi, P. scobina, P. motoro* and *Plesiotrygon iwamae* (Araujo *et al*., 2004a; SBEEL, 2005).

Export data from Colombia indicate that 216 specimens of *P. aiereba* were exported between 2007 and 2011 (proposal). This represents a small proportion of the total number of Potamotrygonidae exported from the country, which increased from about 5 000 individuals in 1995 to a peak of more than 60 000 in 2008, decreasing afterwards to about 20 000 per year (proposal, Figure 2).

The control of populations by ecotourism companies has been carried out in Brazil to avoid accidents with tourists. This activity, which appears to be unregulated, removed an estimated 21 000 individuals from the wild populations of the various species, including *P. aiereba* (Araujo *et al*., 2004a).

1.1.4 Population status and trends

Population size

No estimates of total population abundance are available.

Area of distribution

The species is widely distributed in the Amazon and Orinoco river basins, which cover an estimated total area of about 6.8 million km^2 (International River Basin Registry, Oregon State University, available at www.transboundarywaters.orst.edu/database/interriverbasinreg.html).

Population trend

There is no available information on the overall trends on the abundance of populations. There is information from two studies that can be used to infer declines in two locations. According to unpublished data from a survey conducted in Estrella Fluvial de Inirida (Colombia), no *P. aiereba* were detected in the surveyed area (252 943 ha), where the species used to be abundant. In Brazil, a decline in catches in Santarém and adjacent region was suggested (Charvet-Almeida and Almeida, 2008).

The International Union for Conservation of Nature (IUCN) classifies the species as data deficient (Araujo and Rincon, 2009). However, in Colombia, the species was considered vulnerable owing to population decline driven by overexploitation in the last ten years (Lasso and Sanchez- Duarte, 2012, cited in the proposal). This reference could not be accessed by the Panel to verify the basis for this conclusion. Only *Potamotrygon yepezi* was listed under the category vulnerable in the Catatumbo region, Colombia (Mojica, Usam and Vásquez, 2002).

Landings data from Brazil show an increasing trend from 2001 to 2010, from 25 to 788 tonnes when considering regions that most probably include dasyatids in the sampling (Figure 1). When considering only data from the Santarem region, one of the main landing sites for potamotrygonids and where no dasyatids are caught, landings increased from 7 to 104 tonnes between 2001 to 2004 (Figure 1). It is estimated that *P. aireba* comprises approximately 70 percent of the landings in Santarem.

There is no information in the proposal about population trends in other range States.

1.2 Assessment relative to quantitative criteria

1.2.1 Small population

There are no estimates of population numbers for the species. Recent studies indicate that *P. aiereba* populations are structured within each river, with no or nearly non-existent gene flow occurring between rivers (Frederico *et al*., 2012). The possibility of having subpopulations of a small size (fewer than 5 000 individuals, according to CITES Conf. Res. 9.24 Rev. CoP15) in some of the tributaries of the Orinoco and Amazon Rivers cannot therefore be excluded.

1.2.2 Restricted distribution

No guidelines for restricted area of distribution are provided in the CITES criteria, which indicate that thresholds should be taxon-specific (Conf. Res. 9.24 Rev. CoP15). FAO (2001) recommended that historical extent of decline in area of distribution would be a better measure of extinction risk than absolute value of distributional area, but that if no other suitable information is available and absolute area of distribution has to be used for an exploited fish population, analyses should be on a case-by-case basis as no numeric guideline is universally applicable.

Considering that the species occurs in two of the largest river basins in South America, it can be considered to have a large distribution. Some level of reduction in habitat is expected to have occurred in the past as a result of impacts from other sectors (deforestation, mining, etc.).

1.2.3 Decline

Under the CITES criteria for commercially exploited aquatic species (Conf. Res. 9.24 Rev. CoP15), a decline to 15–20 percent of the historical baseline for a low-productivity species might justify consideration for Appendix I. For a listing in Appendix II, being “near” this level might justify consideration, “near” for a low-productivity species being 20–30 percent of the historical abundance level (15–20 percent + 5–10 percent). For a medium-productivity species, the Appendix I level would be 10–15 percent of the baseline, the Appendix II (“near”) level 15–25 percent.

No overall population decline index is available for comparison with the guidelines. Local declines in abundance appear to have occurred in Colombia and Brazil, but these have not been quantified.

Were trends due to natural fluctuations?

There is no indication in the sources available that declines were due to natural fluctuations.

2. Comments on technical aspects in relation to trade, management and implementation issues

2.1 Trade aspects

The main products in trade are meat and live specimens for the ornamental fish trade (proposal). *P. aiereba* is consumed domestically in Brazil and Venezuela (Bolivarian Republic of). Meat is reported to be exported from Brazil to Japan and the Republic of Korea (Ramos, 2009). *Dasyatis* spp. meat is regularly exported from the northern coast of Brazil and it is likely that potamotrygonid meat is being included along with it (Charvet, personal communication). The proportion of catches consumed internally or exported is unknown because of the lack of specific catch and trade data. Colombia and Peru export live specimens for ornamental use (Araujo and Rincon, 2009). According to data presented in the proposal, the species comprises a small proportion of the ornamental trade of freshwater stingrays from Colombia; a total of 216 specimens were exported between 2007 and 2011. The main importers were Thailand, China and Japan (Annex 3 of the proposal). In Brazil, the harvest of the species for ornamental use is prohibited (see Fisheries management aspects). The possible illegal harvest of the species in Brazil to be exported to other range States is recognized as a potential management issue (Ramos, 2009).

Similar to other freshwater stingrays, *P. aiereba* is sold at a high price in ornamental fish markets. For example, the retail price in an online fish shop in the United States of America is USD200 per specimen.

In parallel to the trade in wild-caught specimens, it appears that large-scale captive breeding of freshwater stingrays in Asian countries is supplying both domestic and export markets. This is not the case for ceja river stingrays.

2.2 Fisheries management aspects

In Brazil, Decree IBAMA No. 204/2008 establishes the norms for the harvest and trade of Potamotrygonidae species for ornamental purposes. Because *P. aiereba* is not in the list of allowed species, the capture and trade of the species for ornamental use is prohibited.

According to the information presented in the proposal, since 2007, Colombia has established legal and administrative frameworks for regulating ornamental fisheries, including for freshwater stingrays. Resolución 3532/2007 includes *P. aiereba* in the list of species allowed to be harvested for ornamental use. Permits are required for the harvesting and trade of the species. Resolución 0301/2011 establishes a global annual harvest quota of 23 000 specimens of Potamotrygonidae for 2012, valid for *P. aiereba* and seven other species of freshwater stingrays included in the list of allowed species in Resolución 3532/2007.

P. aiereba was assigned a high priority in Colombia's NPOA-Sharks (Caldas *et al*., 2010). The plan outlines a series of activities and priorities for the Potamotrygonidae species, including the establishment of complementary management measures, improvement of knowledge about population dynamics, fisheries and trade, the development of awareness campaigns and capacity development activities for communities dependent on the ornamental fisheries.

There are no specific regulations for the ornamental use of freshwater stingrays in the other range Sates (proposal; Table 2, CITES AC24 Doc. 14.2). In addition to Colombia, NPOA-Sharks have been adopted by Venezuela (Bolivarian Republic of) and Ecuador (Fischer *et al*., 2012). However, freshwater stingrays are not mentioned in these plans.

Therefore, it appears that the management systems in place in Brazil and Colombia provide some limits for the ornamental fisheries and establish mechanisms for controlling and monitoring catch and trade. However, specific regulations concerning other uses (food, recreational, population control, etc.) appear to be lacking across the region.

The level of compliance and the actual effectiveness of the systems are not discussed in the proposal or in other publications analysed during this review. However, the reported illegal cross-border trade among range States (proposal; CITES AC24 Doc. 14.2) evidenced the need to strengthen regional cooperation in order to improve management effectiveness for the ornamental fisheries.

In this context, building on the outcomes of the South American Freshwater Stingray Workshop, Geneva, 15–17 April 2009 (CITES AC24 Doc. 14.2), the CITES Animals Committee elaborated the following recommendations (AC24 Decision 15.85) for Parties that are range States of Potamotrygonidae:

- Range States increase their efforts to improve data collection on the scale and impact of the threats facing stingray species and populations from collection for ornamental trade, commercial fisheries for food, and habitat damage.
- Range States consider implementing or reinforcing national regulations regarding the management and reporting of capture and international trade of freshwater stingrays for all purposes, including commercial fisheries for food and ornamental trade, and standardizing these measures across the region, for example through existing South American intergovernmental bodies.
- Range States be encouraged to consider the listing of endemic and threatened species of freshwater stingrays (Potamotrygonidae) in CITES Appendix III to support domestic management measures for species entering international ornamental trade and to improve and enhance trade data collection.

The recommendations stress some key points for improving management effectiveness: the need for better data on harvest for different uses; the need for better regulatory frameworks for the different uses; the need for better data on trade (thus the recommended Appendix III listing); and the need for harmonizing national regulations among range States.

2.3 Implementation issues

2.3.1 Basis for findings: legally obtained, non-detrimental

Export permits for Appendix II species must be accompanied by a certificate attesting that the specimens were legally obtained and by NDFs showing that exports are consistent with sustainable harvesting. Colombia and Brazil have specific laws concerning the harvest and trade of freshwater stingrays and are not expected to have problems in certifying that specimens were legally obtained. For range States without specific regulations, the main issue will be to determine whether individuals being exported were captured in the country or whether they were illegally transported across the border between range States. These States will need to adjust their national legislations in the event of Appendix II listing of the species.

Development of an NDF requires appropriate scientific capacity, biological information on the species, and a framework for demonstrating that exports are based on sustainable harvests. If NDFs are to be meaningful and accurate, the findings will need to apply to the specific substock (i.e. by river) from which the harvest was made. Failure to take the stock structure into account could lead to local extirpations even where the harvest from a range State overall was estimated to be sustainable. Substantial improvements in the knowledge about the stock structure, population dynamics and fisheries of freshwater stingrays will be required in order to make scientifically sound NDFs. However, the experience in Brazil and Colombia shows that the lack of data is not an impediment for establishing management systems based on precautionary measures (bans, harvest quotas and permits) and mechanisms for monitoring and control of catches and trade. Compliance with these measures can be the basis for making NDFs while better data are not available.

2.3.2 Identification of products in trade

P. aiereba belongs to a monotypic genus. The identification of the live specimens in trade is straightforward because of the unique external morphological characteristics. As for the trade in meat, identification will probably be only possible with the use of DNA techniques.

2.3.3 "Look-alike" issues

The Panel could not identify any "look-alike" species for *P. aiereba* in live trade, but these may exist for meat trade.

2.4 Likely effectiveness of a CITES listing for the conservation of the species

The Panel did not find any supporting evidence that a CITES Appendix II listing will probably have an impact on the conservation of the species. Strengthening management by range States will be required to address properly the existing concerns about the conservation and sustainable use of the species.

3. Conclusion

The Panel noted that the supporting statement of the proposal included many unsubstantiated claims, making evaluation difficult.

P. aiereba is a low-to-medium-productivity species. There is no information available to infer population status and trends. Consequently, it was not possible to evaluate whether the population meets the biological criteria for a CITES Appendix II listing.

The available data indicate that *P. aiereba* is traded internationally for ornamental use and possibly for consumption, but the extent of this trade and the effects on the populations are unknown.

The species is also harvested for other purposes including for domestic consumption, and removal for avoiding incidents with tourists, and is also likely to be affected by human-induced habitat changes. The relative importance of these sources of mortality is unknown. Overall, considering that the capture of the species for the ornamental fish trade is prohibited in Brazil and that, according to the proposal, the number of specimens legally traded from Colombia is very low, it seems unlikely that harvesting for the ornamental fish trade is a cause of any population change.

While there are specific regulations to control ornamental harvest and trade in Colombia and Brazil, there are no specific management measures in other range States. Specific regulations concerning other uses (food, recreational, population control, etc.) appear to be lacking across the region. This factor as well as the existence of illegal cross-border trade and the unregulated fisheries constitute risk factors for the conservation and sustainable use of the species.

The Panel did not find any supporting evidence that a CITES Appendix II listing will probably have an impact on the conservation of the species at this time. Strengthening management by range States will be required to address properly the existing concerns about the sustainability of the species.

The Panel noted that the recommendation in paragraph c of Decision 15.85 (to list the species in Appendix III) has not been acted upon by range States. The Panel considers that the implementation of this recommendation will improve trade data, which at present are inadequate.

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TABLES AND FIGURES

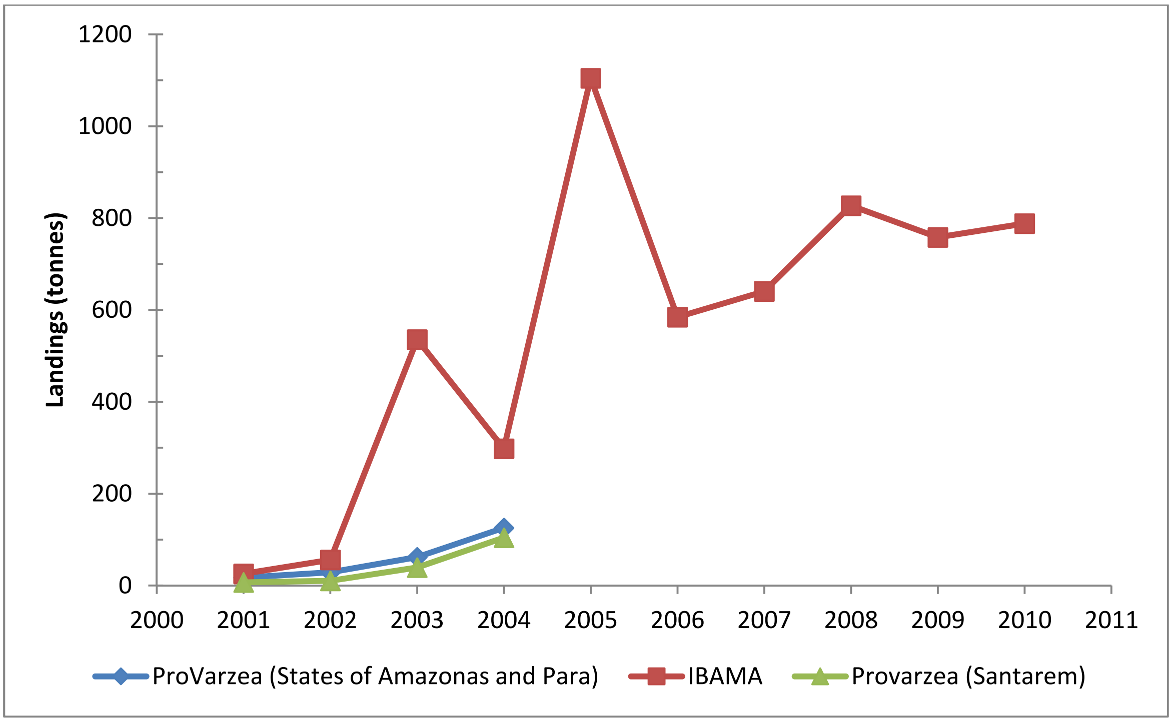
TABLE 1

Information for assessing productivity of the ceja river stingray

Parameter	Information	Productivity	Source
Natural mortality	0.233–0.346/year	Medium	Araujo, 2011
Maximum age	28.2 years	Low	Araujo, 2011
Age at maturity	5.4 years (male) 5.6 years (female)	Medium Medium	Araujo, 2011
K	0.068–0.22/year (male) 0.078–0.24/year (female)	Low – medium Low – medium	Araujo, 2011
r	0.075–0.284	Low – medium	Araujo, 2011

FIGURE 1

Landings of freshwater stingrays (Potamotrygonidae) in Brazil



Note: Landings according to three data sources: ProVarzea, covering only the northern states of Amazonas and Para; ProVarzea, covering only Santarem; and IBAMA covering all states.

Sources: Ruffino (2002, 2005, 2006); Thome-Souza (2007); IBAMA (2003, 2004a, 2004b, 2005a, 2005b, 2007, 2008); MPA (2010, 2012).

FIGURE 2

Exports of specimens of Potamotrygonidae from Colombia, 1995–2012

Source: Proposal.

FAO Expert Advisory Panel assessment report: ocellate river stingray and rosette river stingray - CoP16 Proposal 48 -

Species:

Potamotrygon motoro (ocellate river stingray) and *Potamotrygon schroederi* (rosette river stingray).

Proposal:

Inclusion of the freshwater stingrays *Potamotrygon motoro* and *Potamotrygon schroederi* in Appendix II in accordance with Article II paragraph 2(a) and satisfying Criterion B in Annex 2a of Resolution Conf. 9.24 (Rev. CoP14).

Basis for proposal:

Annex 2a, Criterion B. "It is known, or can be inferred or projected, that a regulation of trade in the species is required to ensure that the harvest of specimens from the wild is not reducing the wild population to a level at which survival might be threatened by continued harvesting or other influences." The proposal indicates that the harvesting of freshwater stingrays for the ornamental fish trade constitutes the main threat for the species and a cause of reduction of populations in the wild. According to the proposal, the inclusion of the species in Appendix II aims to ensure that commercial harvesting will be sustainable. The proposed listing would also contribute to the monitoring and control of the legal activities, reduce illegal trade and support management in the range States.

ASSESSMENT SUMMARY

CITES biological listing criteria

Evidence of decline in abundance is reported for Colombia, but not to the extent required for consideration in Appendix II. In Brazil, the available information indicates that populations showed no trend. The data available are not sufficient to determine whether the species qualify globally under the decline criteria. The two species are distributed across a large area of South America, although different for each species (thus, they cannot be considered under the restricted area criterion) and the populations do not appear to meet the criterion of a small population.

Comments on technical aspects of the proposal

Biology and ecology: The biology of *P. motoro* has been extensively studied while *P. schroederi* is less studied, resulting in less information being available. Both species occur in the various freshwater environments, including large and small rivers, floodplains and lakes in South America. *P. motoro* and *P. schroederi* have different distribution areas and habitat preferences, with the distribution of *P. schroederi* being less extensive and limited to the Amazon and Orinoco river basins.

The population dynamics of both species are poorly known and very few data are available to infer their productivity, status and trends. However, the available information suggests that *P. motoro* has a medium productivity whilst the productivity of *P. schroederi* is probably lower than that of *P. motoro*.

Trade: Considering the high prices of these freshwater stingrays in the ornamental fish trade and the number of individuals exported, it seems that trade is one of the drivers of exploitation. Export data for Colombia and Brazil indicate that at least 99 000 *P. motoro* and 15 000 *P. schroederi* were exported from the two countries between 1999 and 2011. Exports of *P. motoro* from Peru varied from 7 800 to 30 000 individuals per year between 2000 and 2005. Legal exports from Brazil in the last decade have fluctuated in response to changes in national regulations on international trade. It is likely that the increase in captive breeding may be reducing dependence on wild stocks.

Fisheries management: *P. motoro* and *P. schroederi* are harvested for the ornamental trade and food production. In addition, a negative fishery exists (a fishery that removes stingrays to reduce interaction with tourists). The relative importance of these sources of mortality is unknown. There are specific regulations to control ornamental harvest and trade in Colombia and Brazil (the two main exporters). There are no specific management measures in other range States. This factor as well as the illegal cross-border trade of individuals and the unregulated fisheries for other uses constitute risk factors for the sustainable use of the species.

Likely effectiveness of a CITES listing for the conservation of the species: A CITES Appendix II listing might enhance the existing measures to control harvest for the ornamental trade that are partially implemented by some of the exporting countries. Harvesting for other uses, including for food and population control, will not be affected by a CITES listing. At present, the Panel is not in a position to assess the relative importance of international ornamental trade *vis-à-vis* other sources of mortality. Strengthening management at country level will be required in order to address the existing concerns about the sustainability of the species.

The Panel noted that the recommendation in paragraph c of Decision 15.85 (to list the species in Appendix III) has not been acted upon by range States. The Panel considers that the implementation of this recommendation will improve trade data, which at present are inadequate.

The potential difficulty in identifying the species in trade will be the main implementation issue of a possible listing, especially considering that this family has polychromatism (wide inter- and intra-specific colour variation) and hybrids are in international trade.

DETAILED PANEL ASSESSMENT

1. Scientific assessment in accordance with CITES biological listing criteria

1.1 Biological aspects

1.1.1 Population assessed

Potamotrygon motoro and *P. schroederi* belong to the family of freshwater stingrays (Potamotrygonidae), the only group of elasmobranchs fully restricted to the freshwater environment (Compagno and Cook, 1995). Mejía-Falla *et al*. (2009) reported that the common names for *Potamotrygon motoro* are: raya ocelada, raya motora and raya naranja. In Brazil, the common names for this species in the main channel of the Amazon River are indicated as: arraia-de-fogo, arraia-pintada, arraia-de-bola and arraia-bolinha (Charvet-Almeida and Almeida, 2008).

Both species occur in the different freshwater environments, including large and small rivers, floodplains and lakes in South America. *P. motoro* and *P. schroederi* have different distribution areas and habitat preferences (proposal; Drioli and Chiaramonte, 2005; IUCN). *P. motoro* has a wider distribution, occurring in major South American river basins, such as the Amazon, Orinoco, Parana, Paraguay, Uruguay and La Plata. The species is found in Colombia, Venezuela (Bolivarian Republic of), Guyana, Suriname, French Guiana, Brazil, Ecuador, Bolivia (Plurinational State of), Peru, Paraguay, Uruguay and Argentina. It occurs in the main watercourse of black water and white water rivers as well as in floodplain lakes. *P. schroederi* has a more restricted distribution, being reported in the Amazon, Orinoco and Negro river basins, in Colombia, Brazil and Venezuela (Bolivarian Republic of). The species is mainly found in the main watercourse of clear water and black water rivers. Both species occupy high trophic levels, preying on other fishes and aquatic invertebrates (proposal). In the Marajó Island, Brazil, *P. motoro* showed a variable diet according to sampling locations, with insects, molluscs, crustaceans, annelids and fish included among its food items (Almeida *et al*., 2010).

In a recent thorough revision of the data available on the biology, fishery and trade of freshwater stingrays for ornamental purposes in Colombia, it was pointed out that there are few data available for Colombia (Mejía-Falla *et al*., 2009). Nevertheless, their high potential as an economic resource is also indicated and the need of further studies is highlighted (Mejía-Falla *et al*., 2009).

It seems that the taxonomy of freshwater stingrays has not been resolved yet, partly because of the highly variable colour pattering of many species (CITES AC24 Doc. 14.2). In fact, as discussed in the proposal, *P. motoro* presents colour patterns similar to four other species: *P. boesemani*, *P. brachyura*, *P. henlei* and *P. ocellata*. *P. schroederi* can have a very similar colour to *P. tigrina*, which occurs in the Peruvian Amazon (proposal).

There is no information on the population structure or migratory patterns of the species.

1.1.2 Productivity level

The biology of freshwater stingrays is poorly known and life-history parameters are very sparse. The reproductive mode is matrotrophic viviparity (with trophonemata), with annual reproductive cycles closely synchronized with the hydrologic cycle of the river basins (Charvet-Almeida, Araujo and Almeida, 2005). Fecundity varies from 4 to 15 pups for *P. motoro* and from 1 to 3 pups for *P. schroederi* (proposal; Charvet-Almeida, Araujo and Almeida, 2005; Drioli and Chiaramonte, 2005). Sexual maturity in Brazil was estimated at 39 cm and 44 cm DW, respectively, for male and female *P. motoro*, and at 42 cm and 44 cm DW for male and female *P. schroederi* (Charvet-Almeida, Araujo and Almeida, 2005). Smaller sizes at maturity for *P. motoro* have been reported in other basins (e.g. 31 cm for males and 35 cm for females; proposal). Maximum sizes of up to 50 cm and 54 cm DW are reported for *P. motoro* and P. *schroederi*, respectively (proposal; Araujo, 2009). The maximum size attained by *P. schroederi* has been reported to vary from 40 to 60 cm DW (Mejía-Falla *et al*., 2009). Apart from the information on reproduction (Charvet-Almeida, Araujo and Almeida, 2005), there seem to be no further data on the biology of this species.

With respect to the age at first maturity, Charvet-Almeida, Araujo and Almeida (2005) noted that: “Until now, Castex (1963b) and Achenbach and Achenbach (1976) were the only ones that estimated the age of first maturity for *P. motoro* in the wild as being in the latter part of the second year of life, at approximately 20 months, or around three years, respectively. Thorson *et al*. (1983) suggested that the age of first maturity for *P. motoro* could be around 4 years of age, however, they observed that in captivity a pair first reproduced at the age of approximately 71⁄2 years”. Referring to the study of Charvet-Almeida, Araujo and Almeida (2005), the proposal indicates that *P. motoro* reaches sexual maturity in Brazil at 3.5 years, which seems to be a mid-range figure from the values reported in the paper. According to Drioli and Chiaramonte (2005), *P. motoro* reaches sexual maturity at the age of three.

Considering the range of age at maturity of wild populations, it is possible to infer that the species has a medium level of productivity (Table 1). Minimum estimates of the growth parameter K, obtained from the available maximum size, size and age at maturity, indicate that the species grow at rates also consistent with a medium-productivity species (Table 1). Giving the lower fecundity, *P. schroederi* has a lower productivity than *P. motoro*.

1.1.3 Anthropogenic sources of mortality

Among the main sources of threats to the species are: fish harvesting for different purposes (subsistence/artisanal fisheries for food, artisanal fisheries for ornamental fish, bycatch in other commercial fisheries and recreational fisheries); population control in tourist areas; and habitat destruction caused by other sectors, including those resulting from the construction of hydroelectric dams, fluvial ports, mining and oil exploration (proposal; Araujo *et al*., 2004a). The relative importance of these anthropogenic sources of mortality is unknown owing to the lack of accurate information on fisheries takes and the status and trends of populations in areas affected by the different threats. In Colombia, overexploitation for commercial and ornamental use was identified as the main threat to the species (Lasso and Sanchez-Duarte, 2012a, 2012b, cited in the proposal but unavailable to the Panel).

Different types of gear are used in the target food fisheries, including hook and line, longline and harpoon (Drioli and Chiaramonte, 2005; SBEEL, 2005). Bycatch in trawling and gillnet fisheries has been described in Brazil (Araujo *et al*., 2004a, Pereira, 2005) and in bottom longline fisheries in Uruguay (Domingo *et al*., 2008). The capture of live animals for the ornamental trade is conducted with different techniques. In Colombia, the fishery is conducted with a specially designed type of gear ("rayero"; proposal). The rayero consists of a shaft of 1.20 m, with one end attached to a round container in which there is an opening leading to a bag-shaped net of 40 cm. The rayero is placed on top of the stingray so that the animal is led into the bag net (Prada-Pedreros, Gonzalez and Mondragon, 2009). In Brazil, the use of "puça", tidal traps and snorkel is common (SBEEL, 2005). While the main target ornamental fisheries are specimens of small size, mostly juveniles, the food fisheries are more focused on larger sizes (SBEEL, 2005; Anjos *et al*., 2009). Live adults are also harvested to supply the demand of captive breeding facilities in Asian countries (Caldas *et al*., 2010).

There are no catch statistics for South American freshwater stingrays in FAO FishStat. In Brazil, landings are reported at the family level (Potamotrygonidae), and used to include dasyatid species as well (Araujo *et al*., 2004a). The available data show an increasing trend in landings from 2001 to 2010, with production in recent years of the order of 750 tonnes/year (Figure 1). However, these numbers include sampling localities where dasyatids specimens are landed (Charvet, personal communication). The exact proportion of *P. motoro* and *P. schroederi* in this total is unknown. According to Araujo *et al*. (2004a) and SBEEL (2005), five species of Potamotrygonidae are known to be caught in food fisheries in northern Brazil: *Potamotrygon orbignyi*, *P. scobina*, *P. motoro*, *Plesiotrygon iwamae* and *Paratrygon aiereba*. Catches of *P. schroederi* in the state of Amazonas were only for the ornamental trade (Araujo *et al.*, 2004b).

Export data of ornamental fish from Brazil and Colombia indicate that at least 98 913 *P. motoro* and 15 130 *P. schroederi* were exported between 1999 and 2011 (proposal). Exports of *P. motoro* from Colombia show an increasing trend from 1999, peaking at 20 000 individuals in 2008 and declining to 10 000 in 2011 (proposal, Figure 2). Data for Brazil between 2003 and 2005 show annual exports between approximately 3 000 and 7 000 individuals (proposal, Figure 2). The numbers exported do not correspond to the actual number caught because some of the individuals caught may be traded illegally or in domestic markets, being unaccounted for in the export statistics. In addition, the losses and mortality before exportation can be high. According to SBEEL (2005), of the total number of individuals caught for the ornamental trade, only 60 percent end up being exported.

The control of populations by tourism companies has been carried out in Brazil to avoid accidents with tourists (this is a form of negative fishery where stingrays are killed and discarded). This activity, which appears to be unregulated, removed an estimated 21 000 individuals from the wild populations of the various species in Brazil (Araujo et al., 2004a). It remains uncertain how many *P. motoro* and *P. schroederi* are killed.

1.1.4 Population status and trends

Population size

No estimates of total population abundance are available.

Area of distribution

The species are widely distributed in some of the major South American river basins. The Amazon, Orinoco and La Plata basins alone cover an estimated total area of about 10 million km^2 (International River Basin Registry, Oregon State University, available at www.transboundarywaters.orst.edu/database/interriverbasinreg.html).

Population trend

Besides the time series of export data (Figure 2), the proposal does not include any other type of data that could be used to infer population trends. Trade data are a poor indicator of population abundance because they only cover part of the total takes, and are affected by demand and trade regulations in place (such as those described for Brazil in section 2.1). Catch data of the food fisheries would be a slightly better indicator of population trend, but no species-specific catch data are available for these stingray species. Landings data from Brazil show an increasing trend from 2001 to 2010, from 25 to 788 tonnes when considering regions that most probably include dasyatids in the sampling. When considering only data from the Santarém region, one of the main landing sites for potamotrygonids where no dasyatids are caught, landings increased from 7 to 104 tonnes between 2001 and 2004 (Figure 1). From the Santarém region, it is estimated that approximately 15 percent of the landings correspond to *P. motoro*. Total catches of freshwater stingrays in Brazil show an increasing trend for the past decade (Figure 1).

The proposal indicates that in Colombia *P. motoro* and *P. schroederi* were categorized as vulnerable according to IUCN criteria A4d, which refers to the rapid reduction of population size by 30 percent due to overexploitation (Lasso and Sanchez-Duarte, 2012a, 2012b, cited in the proposal). The papers could not be accessed to verify the basis for this conclusion. The IUCN classifies both species as data deficient (www.iucnredlist.org). Only *Potamotrygon yepezi* was listed under the category vulnerable in the Catatumbo region in Colombia (Mojica *et al*., 2002).

The low CPUE of *P. motoro* in a fisheries survey conducted in 2007 in Cano Negro, an important area for the collection of ornamental fish in Colombia (Prada-Pedreros, Gonzalez and Mondragon, 2009), is used in the proposal as an indicator of population decline. However, Prada-Pedreros, Gonzalez and Mondragon (2009) reported that the low CPUE of *P. motoro* is explained by the territorial behaviour of the species, which are caught usually in low numbers but with high frequency in the ornamental fishery with the “rayera”. The absence of the species in more recent surveys in Cano Negro and in Estrella Fluvial de Inirida (at the confluence of the Orinoco, Guaviare, Inirida and Atapabo Rivers) is also reported in the proposal based on unpublished data.

Based on the analysis of CPUE data of the ornamental fishery in Amazonas State, Brazil, from 1998 to 2001, Araujo *et al*. (2004b) concluded that there was no evidence of reduction in the abundance of *P. motoro*. The population was considered in equilibrium by SBEEL (2005). There is no information on population trends of *P. schroederi*. The species appears to be rare in the export trade because of high post-capture mortality and the higher costs involved in its exploitation for the aquarium trade (Araujo *et al*., 2004a).

There is no information in the proposal about population trends in other range States.

1.2 Assessment relative to quantitative criteria

1.2.1 Small population

There are no estimates of population numbers for the two species. Considering the number of stingrays exported in the last decade, and that an additional undetermined number are harvested for other purposes, population numbers cannot be below the general guideline (5 000) for small population size provided in the CITES definitions (CITES Conf. Res. 9.24 Rev. COP14).

1.2.2 Restricted distribution

No guidelines for restricted area of distribution are provided in the CITES criteria, which indicate that thresholds should be taxon-specific (Conf. Res. 9.24 Rev. CoP14). FAO (2001) recommended that historical extent of decline in area of distribution would be a better measure of extinction risk than absolute value of distributional area, but that if no other suitable information is available and absolute area of distribution has to be used for an exploited fish population, analyses should be on a case-by-case basis as no numeric guideline is universally applicable.

Considering that *P. motoro* is widely distributed in some of the largest river basins in South America, it can be considered to have a large distribution range. In addition, there is no evidence of decline in the area of distribution of the species. The distribution of *P. schroederi* is more limited but still comprises large river areas in Colombia, Brazil and Venezuela (Bolivarian Republic of).

1.2.3 Decline

Under the CITES criteria for commercially exploited aquatic species (Conf. Res. 9.24 Rev. CoP14), a decline to 10–15 percent of the historical baseline for a medium-productivity species might justify consideration for Appendix I. For a listing in Appendix II, being "near" this level might justify consideration; "near" for a medium productivity species being 15–25 percent of the historical abundance level (10–15 percent + 5–10 percent).

No overall population decline index is available for comparison with the guidelines. Populations in Colombia were considered vulnerable based on a presumed recent decline of 30 percent in abundance (Lasso and Sanchez-Duarte, 2012a, 2012b, cited in the proposal). Although this source could not be verified, a 30 percent decline is not within the decline criteria for an Appendix II listing. Other sources of information are used to indicate the current low abundance of the species in important ornamental fishing areas in Colombia, but these do not provide evidence of decline. Overall, the available information from Colombia leaves doubts about the extent of decline of populations.

In Brazil, CPUE data from Amazonas State show no trend in abundance of *P. motoro* in an ornamental fishing area from 1998 to 2001. There are no recent abundance data to evaluate the extent of decline in the last decade. Moreover, there is no information on the past and present trend of the population of *P. schroederi*. Considering that both species have been under an export quota control since 2003, the risk of overfishing caused by the ornamental fisheries is probably low in Brazil. In Colombia, there is an overall number of potamotrygonids that can be exported per year, and the risk of decline is probably under control. Nevertheless, illegal ornamental trade could potentially cause decline. However, harvest for other uses, including by food fisheries, has probably increased during the last decade, as demonstrated by the increasing trend in landings of Potamotrygonidae (Figure 1).

In conclusion, the available data do not show declines in population numbers consistent with Appendix II listing criteria.

Were trends due to natural fluctuations?

Araujo *et al*. (2004b) found a strong relationship between fishing pressure and the water level of the rivers. In very dry years (during El Niño events), when the water level of the rivers is low, the species habitat is less accessible to fishers, resulting in lower CPUE values. Therefore, trends in CPUE may in part be explained by fluctuations in river flooding regimes and its impacts on fisheries rather than only by population abundance. However, there is no indication in the sources available that observed declines were due to natural fluctuations.

2. Comments on technical aspects in relation to trade, management and implementation issues

2.1 Trade aspects

The freshwater stingrays *P. motoro* and *P. schroederi* are traded internationally as ornamental fish. *P. motoro* is one of the most common freshwater stingrays used by aquarists. Brazil, Colombia and Peru are the main exporting countries, although in Brazil they are not the main species of Potamotrygonidae harvested for the ornamental trade (effort seems to be concentrated on *Potamotrygon cf. hystrix* [Araujo *et al.*, 2004b] and on *P. leopoldi* [Charvet-Almeida, 2006]). Harvesting of the species for different purposes (including ornamental trade) is minor in Venezuela (Bolivarian Republic of), Ecuador, Paraguay and Uruguay (CITES AC24 Doc. 14.2). Among the main importing countries are Germany, the United States of America, Japan, Malaysia and China (proposal; Anjos *et al*., 2009).

Export data from Colombia and Brazil indicate that at least 98 913 *P. motoro* and 15 130 *P. schroederi* were exported from the two countries between 1999 and 2011 (proposal; Figure 2). Exports of *P. motoro* from Peru varied from 7 848 to 30 139 individuals per year between 2000 and 2005 (CITES AC24 Doc. 14.2). Legal exports from Brazil in the last decade have been fluctuating in response to changes in trade regulations in the country. Statistics on ornamental fish trade from Brazil are available since 1998, when Decree IBAMA/AM No. 023/1998 established a system of permits in Amazonas States, the main supplier of freshwater species (Araujo *et al*., 2004a; SBEEL, 2005). Specific annual quotas for *P. motoro* and *P. schroederi* were established from 2003 to 2005 at 5 500 *P. motoro* and 1 500 *P. schroederi* (Decree IBAMA No. 36/2003). Approximately 5 000 *P. motoro* were traded in 2002 and 2003 from Amazonas State alone; in the same two years, 962 and 169 *P. schroederi* were traded from the state (Anjos *et al*., 2009). There are no records of legal exports of the species from Brazil in 2006 and 2007 (IBAMA, 2006, 2007a) because of a ban on the ornamental fisheries and trade in these years. Harvest quotas for the species were again established in 2008 and 2009 (Decree IBAMA No. 204/2008a). The total allowable quota was 907 *P. motoro* and 167 *P. schroederi* in 2008, and 5 200 *P. motoro* and 1 000 *P. schroederi* in 2009 (IBAMA Web site: www.ibama.gov.br).

According to Araujo *et al.* (2004a), *P. schroederi* is a rare stingray in export data from Brazil in part because of the high rates of post-capture mortality, which demands special care and increases the cost–benefit ratio for ornamental fishers. The authors also report that some companies can export *P. cf. hystrix* as *P. schroederi*.

The species are among the most valuable fish species in the ornamental trade from Brazil, reaching an export price of USD 16.2 per individual *P. motoro* (compared with USD 0.06–1.04 per unit for the top ten fish species exported from Amazonas State; Anjos *et al*., 2009). Retail prices in the importing countries average USD200 per unit (proposal).

There are no data on the volume of stingrays traded in the domestic ornamental fish market of the range States. In Brazil, approximately 86 percent of the trade of ornamental fish is directed to international markets (Anjos *et al*., 2009). The illegal trade of individuals across the border of range States, although not quantified, appears to be an important issue for some species in some areas, including in Brazil, Colombia, Ecuador, Peru and Venezuela (Bolivarian Republic of) (Araujo *et al*., 2004a).

Large-scale captive breeding of freshwater stingrays occurs in Asian countries, supplying both domestic and export markets. According to the information presented at the South American Freshwater Stingray Workshop, Geneva, 15–17 April 2009 (CITES AC24 Doc. 14.2), captive-breeding operations are providing a wide range of colour patterns from hybrids and domesticated morphs and distributing individuals at competitive prices, owing to lower transportation costs from Asian centres compared with the cost of transportation from South American production areas. The workshop concluded that the development and expansion of these activities has decreased the dependence of the global ornamental fish industry on the wild-caught fishes. This phenomenon appears to be occurring also with other ornamental freshwater fish species exported from Brazil (Anjos *et al*., 2009).

2.2 Fisheries management aspects

Araujo *et al*. (2004a) indicated that Brazil was the only range State with specific regulation controlling the harvesting of freshwater stingrays for the ornamental trade (limited quota per species). However, according to the information presented in the proposal, since 2007 Colombia has also established legal and administrative frameworks for the ornamental fisheries of freshwater stingrays. Resolución 3532/2007 includes *P. motoro* and *P. schroederi* in the list of species allowed to be harvested for ornamental use (general quota for the family). Permits are required for the harvesting and trade of the species. Resolución 0301/2011 establishes a global annual harvest quota of 23 000 specimens of Potamotrygonidae for 2012, valid for the eight species of freshwater stingrays included in the list of allowed species in Resolución 3532/2007.

P. motoro and *P. schroederi* were assigned a very high priority in the Colombian NPOA-Sharks (Caldas *et al.*, 2010). The plan outlines a series of activities and priorities for the Potamotrygonidae species, including the establishment of complementary management measures, improvement of knowledge about population dynamics, fisheries and trade, the development of awareness campaigns and capacity development activities for communities dependent on their commercial exploitation.

There are no specific regulations for the ornamental use of freshwater stingrays in the other range States (proposal; Table 2, CITES AC24 Doc. 14.2). In addition to Colombia, NPOA-Sharks have been adopted by Venezuela (Bolivarian Republic of), Ecuador, Argentina and Uruguay (Fischer *et al.*, 2012). Nonetheless, the level of attention to freshwater stingrays is minimal in these plans. Only Uruguay and Argentina recognize the occurrence of the species in their continental waters, with Argentina highlighting the need to improve knowledge on the status of the species occurring in the La Plata river basin.

In Brazil, Decree IBAMA No. 204/2008 establishes the regulation for harvest and trade of six Potamotrygonidae species for ornamental purposes, including *P. motoro* and *P. schroederi*. The harvest can only occur in the Amazon and Araguaia-Tocantins basins, in the states of Amazonas and Pará. From the total quota, individual, non-transferable selling quotas are distributed among licensed ornamental trade companies and cooperatives. Documentation with the proof of origin of the individuals is required for every company or person transporting or reselling live stingrays. An export registry or import licence is required for the international trade. Although Decree 204/2008 does not specify how quotas are established, they are fixed based on previous studies and on life-history parameters. .

Therefore, it appears that the management systems in place in Brazil and Colombia establish some limits for the ornamental fisheries and mechanisms for controlling and monitoring catch and trade. However, specific regulations concerning other uses (food, recreational, population control, etc.) appear to be lacking across the region.

The level of compliance and the actual effectiveness of the systems are not discussed in the proposal or in other publications analysed during this review. However, the reported illegal cross-border trade among range States (proposal; CITES AC24 Doc. 14.2) highlighted the need to strengthen regional cooperation in order to improve management effectiveness for the ornamental fisheries.

In this context, building on the outcomes of the South American Freshwater Stingray Workshop, Geneva, 15–17 April 2009 (CITES AC24 Doc. 14.2), the CITES Animals Committee elaborated the following recommendations (AC24 Decision 15.85) for Parties that are range States of Potamotrygonidae:

- Range States increase their efforts to improve data collection on the scale and impact of the threats facing stingray species and populations from collection for ornamental trade, commercial fisheries for food, and habitat damage.
- Range States consider implementing or reinforcing national regulations regarding the management and reporting of capture and international trade of freshwater stingrays for all purposes, including commercial fisheries for food and ornamental trade, and standardizing these measures across the region, for example through existing South American intergovernmental bodies.
- Range States be encouraged to consider the listing of endemic and threatened species of freshwater stingrays (Potamotrygonidae) in CITES Appendix III to support domestic management measures for species entering international ornamental trade and to improve and enhance trade data collection.

The recommendations stress some key points for improving management effectiveness: the need for better data on harvest for different uses; the need for better regulatory frameworks for the different uses; the need for better data on trade (thus the recommended Appendix III listing); and the need for harmonizing national regulations among range States in view of the possible occurrence of shared stocks and also to combat illegal trade.

2.3 Implementation issues

2.3.1 Basis for findings: legally obtained, non-detrimental

Export permits for Appendix II species must be accompanied by a certificate attesting that the specimens were legally obtained and by NDFs showing that exports are consistent with sustainable harvesting. Colombia and Brazil, the key exporting countries, have specific laws concerning the harvest and trade of freshwater stingrays and are not expected to have problems in certifying that specimens were legally obtained. For range States without specific regulations, the main issue will be to determine whether individuals were captured in the country or whether they were illegally transported across the border between range States. These States will need to adjust their national legislations in the event of Appendix II listing of the species.

Development of an NDF requires appropriate scientific capacity, biological information on the species, and a framework for demonstrating that exports are based on sustainable harvests. Improvements in the knowledge about the population dynamics and fisheries of freshwater stingrays will be required in order to make scientifically sound NDFs. However, the experience in Brazil and Colombia shows that the lack of detailed data for the species is not an impediment for establishing management systems based on precautionary measures (harvest quotas and permits) and control mechanisms. The compliance with these measures can be considered the basis for making NDFs while better data are not available.

2.3.2 Identification of products in trade

The correct identification of the species in the ornamental trade is expected to be the main implementation challenge of this proposal. The species are normally recognized by their dorsal colour pattern. The variability of colour patterns of some species (inter- and intra-specific polychromatism) can complicate their identification. For example, *P. motoro* presents colour patterns similar to four other species: *P. boesemani*, *P. brachyura*, *P. henlei* and *P. ocellata* (proposal), and also some juveniles of *P. leopoldi* present a similar colour pattern. *P. schroederi* can have a very similar colour to *P. tigrina* (proposal). It is necessary to discuss whether good identification guides will suffice to allow the correct identification of species by enforcement officers.

2.3.3 "Look-alike" issues

Considering the potential problems related to the identification of the species in trade, it is possible that the listing of *P. motoro* and *P. schroederi* in Appendix II of CITES will potentially create justification for listing other Potamotrygonidae species also traded internationally.

2.4 Likely effectiveness of a CITES listing for the conservation of the species

A CITES Appendix II listing might enhance the existing measures to control harvest for the ornamental trade that are partially implemented by some of the exporting countries. Harvesting for other uses, including for food and population control, will not be affected by a CITES listing. At present, the Panel is not in a position to assess the relative importance of international ornamental trade *vis-à-vis* other sources of mortality. Strengthening management at country level will be required in order to address the existing concerns about the sustainability of the species.

3. Conclusion

The biology of *P. motoro* has been extensively studied while there are limited data available for *P. schroederi*. The population dynamics of both species are poorly known and very few data are available to infer their productivity, status and trends. However, the available information suggests that *P. motoro* has a medium productivity. While the productivity of *P. schroederi* could not be assessed, it is probably lower than that of *P. motoro*.

Evidence of decline in abundance are reported for Colombia, but not to the extent required for consideration in Appendix II. In Brazil, the available information indicates that populations have been stable in the recent past. Overall, the limited information does not show any trends in population abundance to evaluate with CITES Appendix II decline criteria.

P. motoro and *P. schroederi* are harvested for the ornamental trade and food production. In addition, a negative fishery exists – a fishery that removes stingrays to reduce interaction with tourists. The relative importance of these sources of mortality is unknown. Nonetheless, considering the high prices of freshwater stingrays in the ornamental fish trade and the number of individuals exported, it seems that trade is one of the drivers of exploitation.

While there are specific regulations to control harvest and trade in Colombia and Brazil (the two main exporters), there are no specific management measures in other range States. This factor as well as the illegal cross-border trade of individuals and the unregulated fisheries for other uses constitute risk factors for the sustainable use of the species. However, it is likely that the increase in captive breeding may be reducing the dependence on wild stocks.

A CITES Appendix II listing might enhance the existing measures to control harvest for the ornamental trade that are at best are partially implemented by some of the exporting countries. In addition, it is noted that harvesting for other uses, including for food and removal for population control, will not be affected by a CITES listing. At present, the Panel is not in a position to assess the relative importance of international ornamental trade *vis-à-vis* other sources of mortality. Strengthening management at country level will be required in order to address properly the existing concerns about the sustainability of the species.

The potential difficulty in identifying the species in trade would be the main implementation issue of a listing, especially considering that this family is polychromatic (inter- and intra-specific colour variations) and hybrids are in international trade.

The Panel noted that the recommendation in paragraph c of Decision 15.85 (to list the species in Appendix III) has not been acted upon by range States. The Panel considers that the implementation of this recommendation will improve trade data, which at present are inadequate.

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TABLES AND FIGURES

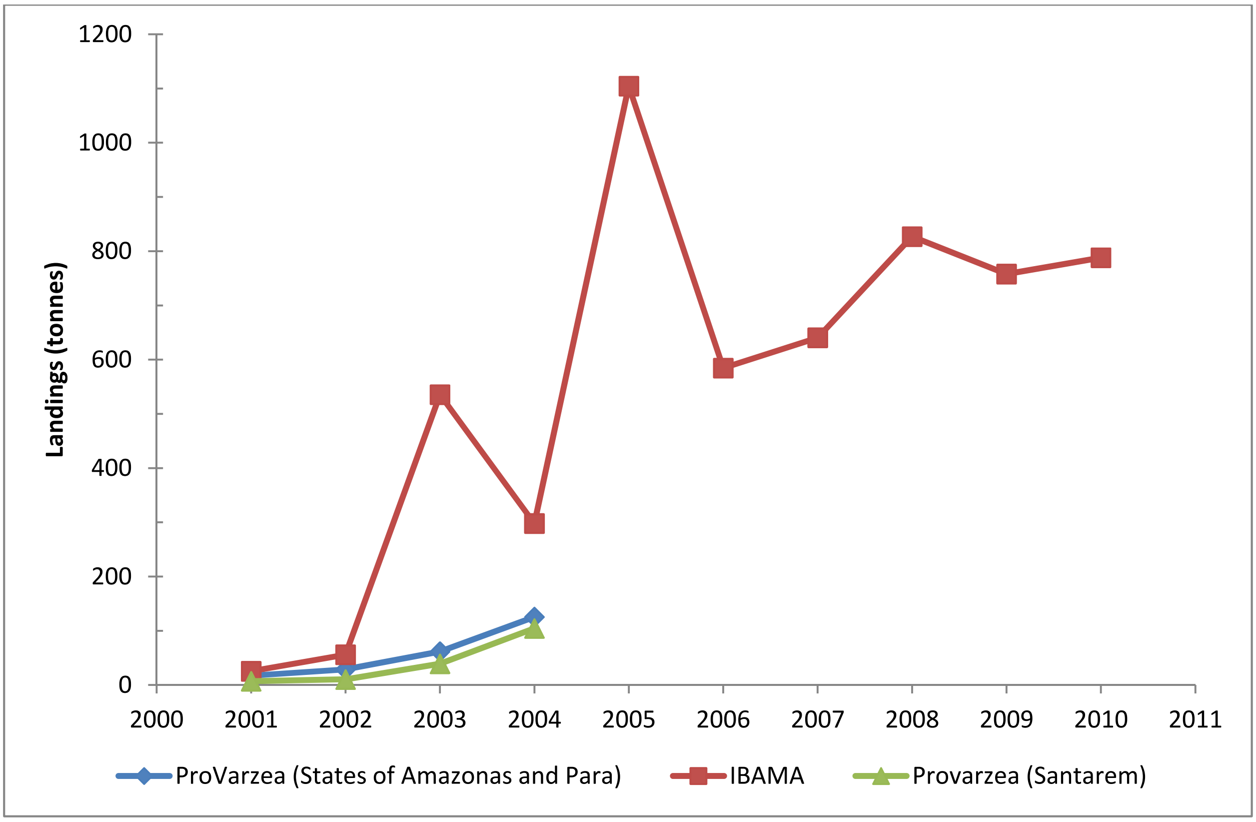
TABLE 1

Information for assessing productivity of South American freshwater stingrays

Parameter	Information	Productivity	Source
Age at maturity	*P. motoro:* 2–4 years	Medium	Charvet-Almeida, Goes de Araujo and Almeida (2005); Drioli and Chiaramonte (2005)
K (year–1)	*P. motoro:* > 0.24	Medium	Estimated assuming K = –ln (1–L_m/L_{inf})/(t_m–t_0); 31 cm <L_m> 44 cm; 2 years <t_m>4 years; L_{inf} = 50 cm; t_0 =0

FIGURE 1

Landings (tonnes) of freshwater stingrays (Potamotrygonidae) in Brazil

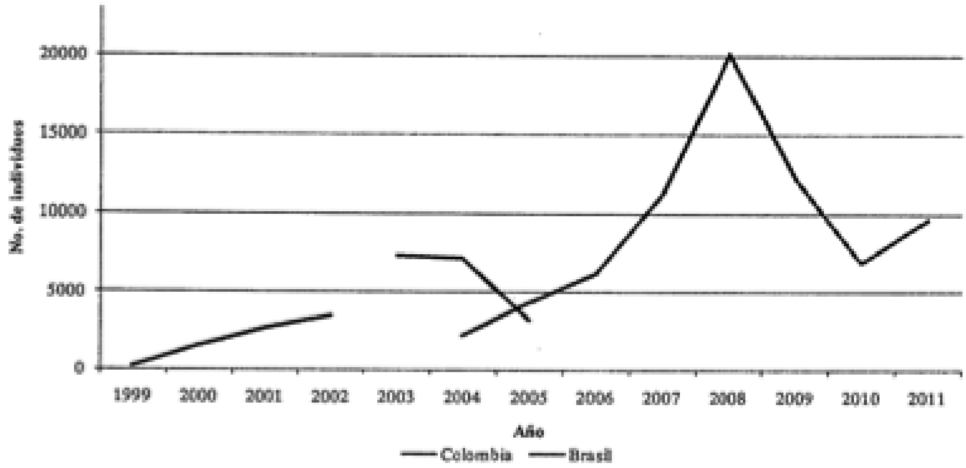


Note: Landings according to three data sources: ProVarzea covering only the northern states of Amazonas and Para, ProVarzea covering only Santarem, and IBAMA covering all states.

Sources: Ruffino (2002, 2003, 2005, 2006); Thome-Souza (2007); IBAMA (2003, 2004a, 2004b, 2005a, 2005b, 2007b, 2008b); MPA (2010, 2012).

FIGURE 2

Exports of *P. motoro* from Colombia and Brazil, 1999–2011



Source: Proposal.

Appendix A

Terms of reference for an “Ad Hoc Expert Advisory Panel for Assessment of Proposals to CITES”[8]

1. FAO will establish an Ad Hoc Expert Advisory Panel for the Assessment of Proposals to Amend CITES Appendices I and II.

2. The Panel shall be established by the FAO Secretariat in advance of each Conference of the Parties, according to its standard rules and procedures and observing, as appropriate, the principle of equitable geographical representation, drawing from a roster of recognized experts, to be established, consisting of scientific and technical specialists in commercially-exploited aquatic species.

3. The Panel members shall participate in the Panel in their personal capacity as experts, and not as representatives of governments or organizations.

4. The Panel will consist of a core group of no more than 10 experts, supplemented for each proposal by up to 10 specialists on the species being considered and aspects of fisheries management relevant to that species.

5. For each proposal the Panel shall:

- assess each proposal from a scientific perspective in accordance with the CITES biological listing criteria, taking account of the recommendations on the criteria made to CITES by FAO;

- comment, as appropriate, on technical aspects of the proposal in relation to biology, ecology, trade and management issues, as well as, to the extent possible, the likely effectiveness for conservation.

6. In preparing its report, the Panel will consider the information contained in the proposal and any additional information received by the specified deadline from FAO Members and relevant regional fisheries management organizations (RFMOs). In addition, it may ask for comments on any proposed amendment, or any aspect of a proposed amendment, from an expert who is not a member of the Panel if it so decides.

7. The Advisory Panel shall make a report based on its assessment and review, providing information and advice as appropriate on each listing proposal. The Panel shall finalize the advisory report no later than ?? days[9] before the start of the CITES Conference of the Parties where the proposed amendment will be addressed. The advisory report shall be distributed as soon as it is finalized to all Members of FAO, and to the CITES Secretariat with a request that they distribute it to all CITES Parties.

8. The general sequence of events will be as follows:

- Proposals received by CITES
- Proposals forwarded by CITES Secretariat to FAO
- FAO forwards proposals to FAO Members and RFMOs and notifies them of deadline for receipt of comments
- Member and RFMO comments and input received by FAO

[8] Taken from Appendix E of the Report of the twenty-fifth Session of COFI, FAO, Rome, 24-28 February 2003
[9] To be discussed with CITES.

- Panel meets and prepares advisory report on each proposal
- Panel report reviewed by FAO Secretariat and forwarded to FAO Members, RFMOs and CITES Secretariat.

Appendix B

Agenda

Monday, 3 December 2012

1. Welcome by Mr Arni Mathiesen, Assistant Director-General, Fisheries and Aquaculture Department
2. Introduction of participants
3. Selection of Panel Chairperson
4. Panel terms of reference, objectives and work programme for the meeting
5. Overview of the CITES listing criteria (Res. Conf. 9.24 [Rev. CoP15])
6. Presentation of the proposal on whitetip shark by the FAO consultant and by the proponent (Colombia)
7. Presentation of the two proposals on stingrays by the FAO consultant and by the proponent (Colombia)
8. Presentation of the proposal on hammerheads by the FAO consultant and by the proponent (Brazil)
9. Discussion with proponents

Tuesday, 4 December 2012

10. Presentation of the proposal on manta rays by the FAO consultant and by the proponent (Ecuador)
11. Presentation of the proposal on freshwater sawfish by the FAO consultant and by the proponent (Australia)
12. Presentation of the proposal on porbeagle by the FAO consultant and by the proponent (Denmark)
13. Discussion with proponents

Wednesday, Thursday and Friday, 5-7 December 2012

14. Preparation of draft reports

Saturday, 8 December 2012

15. Finalization of report on all the seven proposals
16. Clearance and adoption of the report by Panel

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Appendix D
SC 62 Doc.39: Criteria for the inclusion of species in appendices I and II

Original language: English SC62 Doc. 39

CONVENTION ON INTERNATIONAL TRADE IN ENDANGERED SPECIES
OF WILD FAUNA AND FLORA

Sixty-second meeting of the Standing Committee Geneva (Switzerland), 23-27 July 2012

Interpretation and implementation of the Convention

Trade control and marking

CRITERIA FOR THE INCLUSION OF SPECIES IN APPENDICES I AND II*

1. This document has been submitted by the Chair of the Animals Committee.

2. At CoP15 (Doha, 2010) Parties agreed to Decision 15.29, which is directed to the Animals Committee as follows:

   The Animals Committee shall:

   a) on receipt of any or all of the reports referred to in Decision 15.28, and having sought the participation of representative(s) of the Plants Committee, IUCN, TRAFFIC, the Food and Agriculture Organization of the United Nations and other appropriate experts, develop guidance on the application of criterion B and the introductory text of Annex 2 a of Resolution Conf. 9.24 (Rev. CoP15) to commercially exploited aquatic species proposed for inclusion on Appendix II;
   b) recommend the best way to incorporate the guidance for use when applying Resolution Conf. 9.24 (Rev. CoP15) to commercially exploited aquatic species, without affecting the application of Resolution Conf. 9.24 (Rev. CoP15) to other taxa; and
   c) submit its conclusions and recommendations at the 62nd meeting of the Standing Committee.

3. Regarding the application of Annex 2a criterion B and the introductory text to commercially exploited aquatic species, the Animals Committee noted that:

   a) While there are diverse approaches to the application of Annex 2a criterion B, there is commonality in that all Parties and those reviewing listing proposals should take a taxon-specific approach that is sensitive to species vulnerabilities and they should be mindful of the precautionary approach as outlined in Annex 4 of Resolution Conf. 9.24 (Rev.

* *The geographical designations employed in this document do not imply the expression of any opinion whatsoever on the part of the CITES Secretariat or the United Nations Environment Programme concerning the legal status of any country, territory, or area, or concerning the delimitation of its frontiers or boundaries. The responsibility for the contents of the document rests exclusively with its author.*

CoP15).

b) Vulnerability is defined in Annex 5 of Resolution Conf. 9.24 (Rev. CoP15) as the susceptibility to intrinsic or external effects which increase the risk of extinction, and examples of intrinsic and extrinsic factors are provided. Further, the footnote to decline in Annex 5 reiterates that "account needs to be taken of taxon- and case- specific biological and other factors that are likely to affect extinction risk."

c) When considering whether a species qualifies for listing on CITES Appendix II, Parties and those reviewing listing proposals should be aware that, where numerical guidelines or thresholds are provided, they are presented only as examples, since it is impossible to give numerical values that are applicable to all taxa because of differences in their biology.

d) When considering whether a species qualifies for listing on CITES Appendix II, the analysis done by Parties and those reviewing listing proposals are influenced by their level of risk tolerance, which itself is informed by the quality and quantity of available information, their objectives, and their experiences. The variability in Parties' and those reviewing proposals' risk tolerance may be more pronounced when considering commercially exploited aquatic species.

e) The foregoing points are useful to consider and be mindful of when preparing or evaluating proposals to list commercially exploited aquatic species on Appendix II.

4. The Animals Committee finds that there are diverse approaches to the application of Annex 2a criterion B in Resolution Conf. 9.24 (Rev. CoP15). The Animals Committee finds that it is not possible to provide guidance preferring or favouring one approach over another. The Animals Committee recommends that Parties, when applying Annex 2a criterion B when drafting or submitting proposals to amend the CITES Appendices, explain their approach to that criterion, and how the taxon qualifies for the proposed amendment.

5. When drafting and submitting proposals to amend the CITES Appendices with respect to commercially- exploited aquatic species, the Animals Committee encourages Parties to elucidate the vulnerabilities, as defined in Annex 5 of Resolution Conf. 9.24 (Rev. CoP15), and mitigating factors including, but not limited to, large absolute numbers, refugia and fisheries management measures that they have considered.

6. The Animals Committee notes the lack of a definition of commercially-exploited aquatic species in the existing body of CITES documentation, and further notes that FAO documentation indicates that commercially-exploited aquatic species refer to fish and invertebrate species found in marine environments or in large freshwater bodies and subject to commercial exploitation (FAO 2001)[†]

7. The Animals Committee noted the issue of how to determine whether a commercially exploited aquatic species qualifies for listing on CITES Appendix II when that species is found in multiple stocks or subpopulations with varying statuses. The issue was raised in the papers provided by the CITES Secretariat, FAO and IUCN/TRAFFIC (AC25 Doc. 10), further discussed by Germany (AC25 Inf. 10) and referenced in discussions of the Animals Committee working group on criteria. There was recognition of the complexity of the issue and differing views on how to approach this matter. The Animals Committee invites the Standing Committee to consider the merit of continuing a discussion on this matter within CITES.

[†] *Second technical consultation on the suitability of the CITES criteria for listing commercially-exploited aquatic species, www.fao.org/docrep/meeting/003/Y1455E.htm.*

Appendix E

Extract from a FAO proposal for the 2017 revision of the Harmonized System of the World Customs Organization

The FAO proposal covers fishery, agriculture, forestry products and agricultural inputs (fertilizers and agricultural machinery). It has been developed in cooperation with the National Oceanic and Atmospheric Administration (NOAA), the U.S. Fish and Wildlife Service, and the US Department of State; Eurostat, ITTO (International Tropical Timber Organization) and UNECE (United Nations Economic Commission for Europe); VDMA (Verband Deutscher Maschinen und Anlagenbau, the German Machinery and Equipment Federation) and ABIMAQ (Associação Brasileira da Indústria de Máquinas e Equipamentos, the Brazilian Agricultural Machinery Association).

The proposal is still under review and therefore not publicly available. Here the following information from the proposal with regard to the following elasmobranch products can be disclosed.

- Fillets and meat of dogfish/sharks and of rays and skates.
- Shark fins in frozen and in prepared and preserved form.
- Distinction for fins of hammerhead sharks, oceanic whitetip shark, blue shark and porbeagle shark in cured form.

In detail the proposed codes are the following:

0303.92 -- Shark fins (frozen)

0304.47 -- Dogfish and other sharks (Fresh or chilled fillets)

0304.48 -- Rays and skates (*Rajidae*) (Fresh or chilled fillets)

0304.56 -- Dogfish and other sharks (Fresh or chilled meat)

0304.57 -- Rays and skates (*Rajidae*) (Fresh or chilled meat)

0304.88 -- Dogfish, other sharks, rays and skates (*Rajidae*) (Frozen fillets)[FYI, no more codes have available to introduce separately for sharks/rajidae]

0304.96 -- Dogfish and other sharks (frozen meat)

0304.97 -- Rays and skates (*Rajidae*) (frozen meat)

0305.73 -- Dried, whether or not salted, fins of hammerhead sharks (*Sphyrnidae*), with skin and cartilage

0305.74 -- Dried, whether or not salted, fins of oceanic whitetip shark (*Carcharhinus longimanus*), with skin and cartilage

0305.75 -- Dried, whether or not salted, fins of blue shark (*Prionace glauca*), with skin and cartilage

0305.76 -- Dried, whether or not salted, fins of porbeagle shark (*Lamna nasus*), with skin and cartilage

0305.77 -- Other shark fins (cured)

1604.18 -- Shark fins (prepared and preserved)

The fourth FAO Expert Advisory Panel for the Assessment of Proposals to Amend Appendices I and II of CITES Concerning Commercially-exploited Aquatic Species was held at FAO headquarters from 3 to 8 December 2012. The Panel was convened in response to the agreement by the twenty-fifth session of the FAO Committee on Fisheries (COFI) on the terms of reference for an expert advisory panel for assessment of proposals to the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), and to the endorsement of the twenty-sixth session of COFI to convene the Panel for relevant proposals to future CITES Conference of the Parties. The objectives of the Panel were to: i) assess each proposal from a scientific perspective in accordance with the CITES biological listing criteria (Resolution Conf. 9.24 [Rev. CoP13]; and ii) comment, as appropriate, on technical aspects of the proposal in relation to biology, ecology, trade and management issues, as well as, to the extent possible, the likely effectiveness for conservation. Seven proposals were evaluated by the Panel: (1) CoP16 Prop. 42. Proposal to include *Carcharhinus longimanus* (oceanic whitetip shark) in Appendix II in accordance with Article II paragraph 2(a); (2) CoP16 Prop. 43. Inclusion of *Sphyrna lewini* in Appendix II in accordance with Article II 2(a) and inclusion of *S. mokarran* and *S. zygaena* in Appendix II in accordance with Article II 2(b); (3) CoP16 Prop. 44. Inclusion of *Lamna nasus* (Bonnaterre, 1788) in Appendix II in accordance with Article II 2(a); (4) CoP16 Prop. 45. Transfer of *Pristis microdon* from Appendix II to Appendix I of CITES in accordance with Article II, paragraph 1; (5) CoP16 Prop. 46. Inclusion of the genus *Manta* in Appendix II in accordance with Article II paragraph 2(a); (6) CoP16 Prop. 47. Inclusion of the ceja river stingray (*Paratrygon aiereba*) in Appendix II in accordance with Article II paragraph 2(a); and (7) CoP16 Prop. 48. Inclusion of the freshwater stingrays *Potamotrygon motoro* and *P. schroederi* in Appendix II in accordance with Article II paragraph 2(a). This report includes the assessment of each of the seven proposals by the Panel.