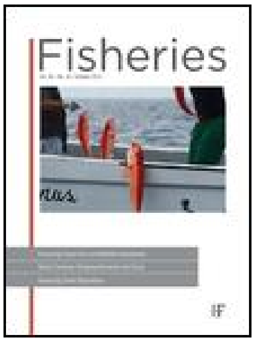


Fisheries

ISSN: 0363-2415 (Print) 1548-8446 (Online) Journal homepage: http://www.tandfonline.com/loi/ufsh20

Is a Global Quantitative Assessment of Shark Populations Warranted?

Matias Braccini

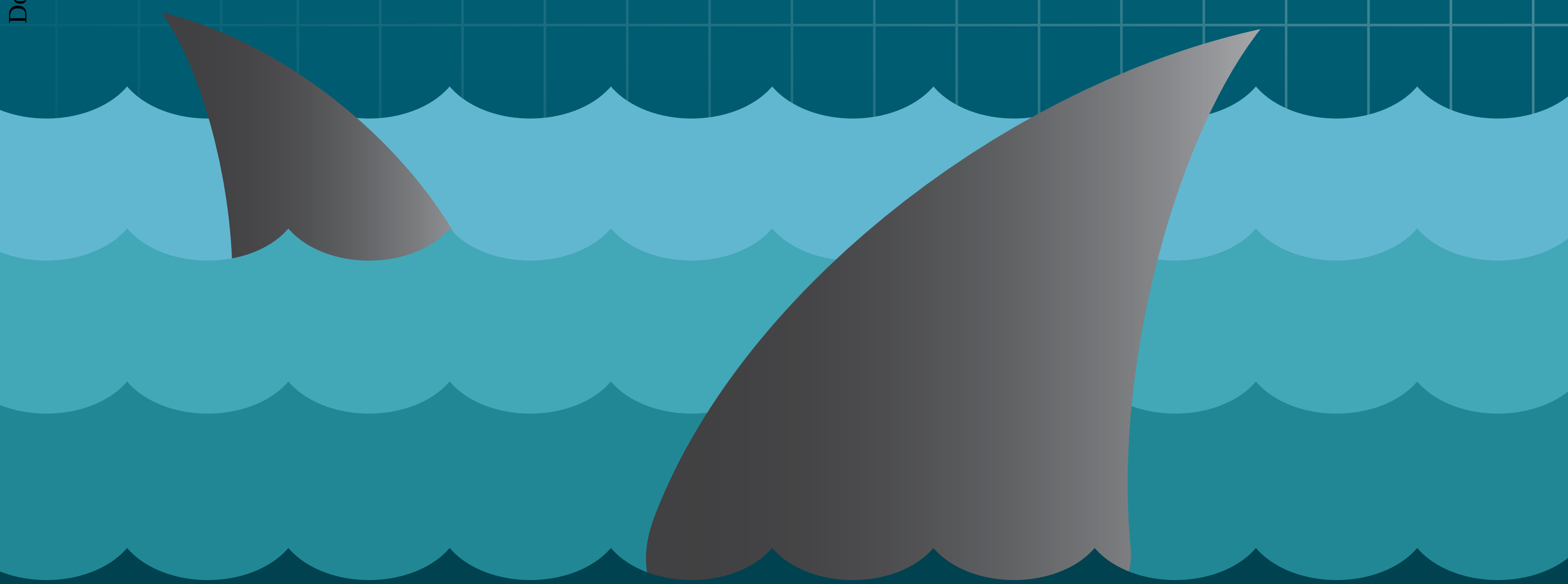
To cite this article: Matias Braccini (2015) Is a Global Quantitative Assessment of Shark Populations Warranted?, Fisheries, 40:10, 492-501, DOI: 10.1080/03632415.2015.1080689

To link to this article: http://dx.doi.org/10.1080/03632415.2015.1080689

FEATURE

Is a Global Quantitative Assessment of Shark Populations Warranted?

Matias Braccini
Western Australian Fisheries and Marine Research Laboratories, PO Box 20, North Beach, WA 6920, Australia. E-mail: Matias.Braccini@fish.wa.gov.au



A recent global quantitative assessment suggested that "the majority of shark populations will continue to decline under current fishing pressure" (Worm et al. 2013:198) and concluded that global shark mortality needs to be drastically reduced to rebuild populations and restore marine ecosystems with functional top predators. The high exploitation rates inferred by the authors are alarming and, if accurate, justify the increased concern of the global conservation community. To assess the generality and accuracy of this work, I critically evaluate the assumptions and validity of the extrapolations made by the authors. This global study provided a valuable overall perspective on the highly relevant topic of shark conservation; however, the generalizations made carry substantial uncertainty that was not accounted for. My review aims to place the conclusions drawn by the authors into perspective, highlighting numerous factors that, having been considered, would have significantly affected their claims.

¿Está garantizada una evaluación cuantitativa global de poblaciones de tiburón?

Una evaluación cuantitativa global sugiere que la mayor parte de las poblaciones de tiburones continuarán decreciendo bajo la presión actual de pesca, concluyendo que la mortalidad global de tiburones necesita ser drásticamente reducida con el fin de recuperar las poblaciones y restaurar a los ecosistemas marinos de las funciones que cumplen los depredadores tope. Las altas tasas de explotación que los autores infieren son alarmantes y, de ser ciertas, justifican la creciente preocupación de la comunidad global de conservación. Con el fin de evaluar la generalidad y precisión de dicho trabajo, aquí hago una evaluación de las suposiciones y validez de las extrapolaciones hechas por los autores. Este estudio global ofrece una valiosa perspectiva general de un tema relevante como lo es la conservación de los tiburones; sin embargo, las generalizaciones hechas por los autores tienen una fuerte carga de incertidumbre, las cual no fue considerada. Esta revisión tiene como objetivo poner en perspectiva las conclusiones a las que llegaron los autores, subrayar numerosos factores que, si se tomasen en cuenta, podrían afectar significativamente sus aseveraciones.

Est une évaluation quantitative globale des populations de requins justifiée?

Une évaluation quantitative mondiale récente a suggéré que «la majorité des populations de requins va continuer à décliner sous la pression actuelle de la pêche» et a conclu que la mortalité globale des requins doit être considérablement réduite pour reconstituer les populations et repeupler les écosystèmes marins de grands prédateurs fonctionnels. Les taux d'exploitation élevés constaté par les auteurs sont alarmants, et s'ils sont exacts, ils justifient la préoccupation accrue de la communauté mondiale de la conservation. Pour évaluer la généralité et l'exactitude de ce travail, j'évalue de manière critique les hypothèses et la validité des extrapolations faites par les auteurs. Cette étude mondiale a fourni une vue d'ensemble précieuse sur le sujet très pertinent qu'est la conservation des requins ; toutefois, les généralisations faites portent une incertitude substantielle qui n'a pas été prise en compte. Mon analyse vise à mettre les conclusions tirées par les auteurs en perspective, en soulignant de nombreux facteurs qui, s'ils avaient été pris en considération, auraient considérablement affecté leurs affirmations.

INTRODUCTION

Overall, sharks have low biological productivity and limited capacity to sustain high exploitation rates (e.g., Musick et al. 2000). As a result, some populations have drastically declined due to overfishing (Stevens et al. 2000), particularly the pelagic and some of the reef and coastal species (e.g., Walker 1998; Dulvy et al. 2008; Nadon et al. 2012). However, dramatic population declines and collapses have only been documented for a fraction of the several populations of the more than 500 living shark species (Eschmeyer 2014). This is mostly due to a combination of three factors. Firstly, not all shark populations are exposed to unsustainable fishing pressure. Secondly, the productivity of sharks varies widely, and population declines are not consistent for all species (Burgess et al. 2005). Hence, as with any other taxa, sharks can be harvested sustainably (Walker 1998), and there are examples of this for short-lived (Walker 1998; SEDAR 2007) and long-lived (McAuley et al. 2007; Kulka et al. 2012) species. Finally, gathering the information needed for quantifying population trends is difficult; due to their low economic value, sharks are generally not a research or management priority (Walker 1998). As a result, this information is generally lacking for sharks, hindering the application of standard quantitative methods for assessing population status. Alternatively, global assessments of shark populations have used Red List Categories and Criteria developed by the International Union for the Conservation of Nature (IUCN) to determine relative extinction risks (Dulvy et al. 2014). For the 465 shark species analyzed, only 74 were considered "vulnerable," "endangered," or "critically endangered," and as the vast majority of the species (209) were "data deficient," predictive models were used to determine their relative risk (Dulvy et al. 2014). This comprehensive global assessment highlights the lack of quantitative information on trends in abundance and fishery exploitation for most shark populations and, therefore, the reason why quantitative methods for estimating population declines and exploitation rates have not been extensively applied on sharks.

Dramatic population declines and collapses have only been documented for a fraction of the several populations of the more than 500 living shark species.

A recent study by Worm et al. (2013), however, calculated the total exploitation rate of all sharks (i.e., all populations from all shark species) from estimates of total global mortality and biomass. The authors then compared this single estimate of global exploitation rate against the average intrinsic rebound potential from 62 species to infer the conservation status of all shark populations (Figure 1). The analysis of Worm et al. (2013) suggests that the majority of shark populations will continue to decline under current fishing pressure and that global shark mortality needs to be drastically reduced. These findings attracted considerable attention (e.g., it was *Marine Policy*'s most downloaded article in 2013) and generated great concern within the conservation community and mainstream society.

We do know that most shark species are very vulnerable to

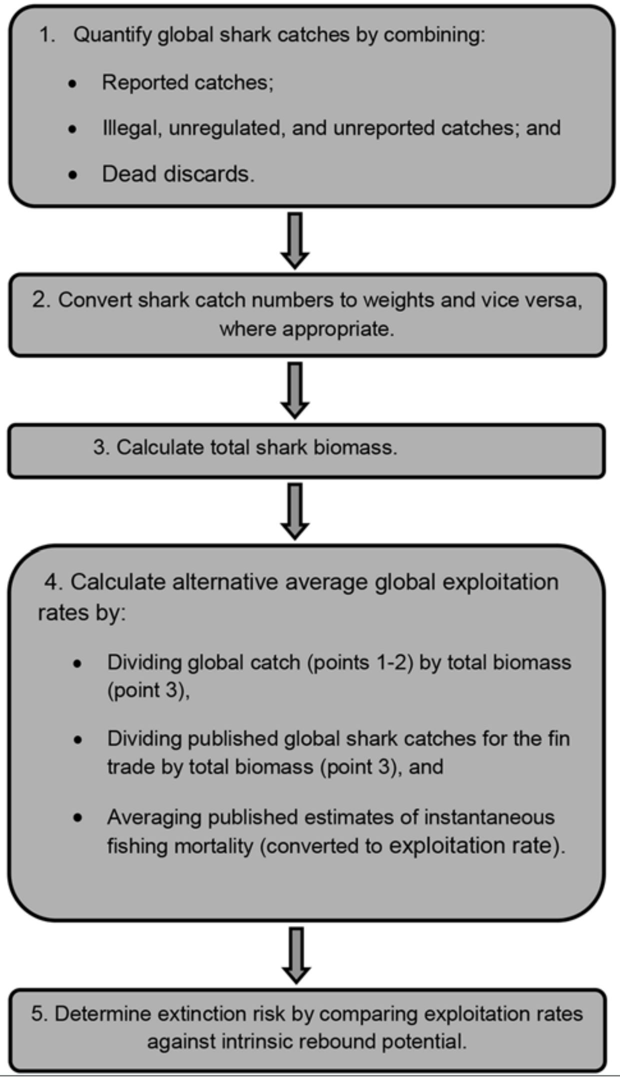


Figure 1. Diagram of the steps undertaken by Worm et al. (2013) for the quantitative assessment of all shark populations.

overfishing due to their biological characteristics (Walker 1998; Musick et al. 2000). However, whether this high vulnerability leads to overfishing depends on the actual exploitation rate exerted on the population. The high exploitation rates inferred by Worm et al. (2013) are alarming and, if accurate, justify the increased concern of the global conservation community. Their assessment thus requires careful consideration of the approach used for deriving such claims. To assess the generality and accuracy of the conclusions from Worm et al. (2013), I critically evaluate the assumptions and validity of the extrapolations made by these authors. I also highlight numerous factors that, having been considered, would have significantly affected the authors' conclusions.

In order to undertake a global quantitative assessment, Worm et al. (2013) needed to make substantial assumptions for estimating the global catch and biomass of shark populations. The authors, however, did not evaluate how these assumptions influenced their results. Hence, I consider a range of alternative assumptions (Table 1) in a sensitivity analysis to illustrate the influence of these assumptions when deriving exploitation rates and assessing all shark populations (Table 2).

TOTAL GLOBAL SHARK CATCHES

Worm et al. (2013) derived total shark mortality from a point estimate of total global shark catches. This estimate was obtained by combining information on reported catches; illegal, unregulated, and unreported (IUU) catches; and discards (accounting for finning and post-release mortality [PRM]).

Reported Catches

Worm et al. (2013) used global databases (Watson et al. 2004; FAO 2012) to derive reported shark catches for 2010. Because global databases report catches by species groups, the authors considered the following chondrichthyan (sharks, rays, and chimaeras) catch classes: (1) large coastal and pelagic sharks; (2) small coastal sharks; (3) deepwater sharks; (4) undifferentiated sharks, rays, and chimaeras (mixed-species group); (5) rays, skates, and chimaeras (separate groups); and (6) undifferentiated rays and skates. Worm et al. (2013) excluded rays, skates, and chimaeras from their analyses. Hence, to calculate the total take of sharks only, Worm et al. (2013) assumed that the proportion of sharks from the mixed-species group was the same as the proportion in the differentiated groups, calculating the reported global annual catch at 392,226 tons (Table 1). The validity of this assumption is arguable. Sharks generally compose the greatest proportion of the elasmobranch (sharks, rays, and skates) catch of longline and net fisheries, whereas skates and rays dominate the elasmobranch catch of trawl fisheries (Bonfil 1994). For the differentiated groups, sharks comprised a large proportion of the elasmobranch catch (mostly taken by longlines and nets). However, when sharks are reported together with rays, skates, and chimaeras as a "mixed group," it is likely that sharks were caught together with these other chondrichthyans in trawl fisheries. In such a case, the proportion of sharks from these fisheries would be smaller than from longline or net fisheries. Worm et al. (2013) did not report the values for the different chondrichthyan catch classes. Hence, for the sensitivity test, I assumed that the proportion of sharks in the undifferentiated group was smaller than in the differentiated group and arbitrarily set the reported global annual catch at 370,000 tons (Table 1).

Illegal, Unregulated, and Unreported Catches

Worm et al. (2013) derived IUU shark catches by averaging the low and high (11 and 26 million tons, respectively) global IUU estimates for all taxa combined reported by Agnew et al. (2009). Given that Agnew et al. (2009) did not explicitly report the IUU catch for sharks, Worm et al. (2013) assumed that the proportion of sharks in the IUU catches was the same as in the reported catches, estimating the global annual IUU shark catch at 111,000 tons (Table 1). The validity of this assumption is also difficult to assess, but it is worth noting that global IUU estimates show considerable variability among species groups (Agnew et al. 2009). For example, high-value demersal fish, lobsters, and shrimps/prawns had the highest level of illegal fishing. Other less valuable groups had much lower levels. Agnew et al. (2009) also reported considerable variability among oceanic regions with the Eastern Central Atlantic and the Southwest Pacific showing the highest and lowest IUU catch levels, respectively. The IUU catch level is also related to the fishery type because some fisheries are more prone to illegal activities (e.g., pelagic fishing in the high seas) than others (e.g., highly regulated fisheries within Exclusive Economic Zone waters; OECD 2005). Failing to consider all of this information

Table 1. Summary of the assumptions made by Worm et al. (2013) and an alternative approach for the calculation of global shark catch and biomass.

Quantity	Value (tons)	Worm et al. (2013) assumptions	Value (tons)	Alternative assumptions
Catch				
Reported	392,226	The proportion of sharks from the undifferentiated chondrichthyan group is the same as the proportion in the differentiated groups.	370,000	The proportion of sharks from the undifferentiated chondrichthyan group is smaller than the proportion in the differentiated groups.
Illegal, unregulated, and unreported (IUU)	111,000	Shark IUU catch is the average global annual IUU catch multiplied by the proportion of sharks (0.006) in the reported global catches.	33,000	Shark IUU catch is the low estimate of the global annual IUU catch multiplied by 0.003 (i.e., half of the proportion of sharks in the reported global catches assumed by Worm et al. [2013]).
			132,000	Shark IUU catch is the low estimate of the global annual IUU catch multiplied by 0.012 (i.e., double of the proportion of sharks in the reported global catches assumed by Worm et al. [2013]).
			78,000	Shark IUU catch is the high estimate of the global annual IUU catch multiplied by 0.003.
			312,000	Shark IUU catch is the high estimate of the global annual IUU catch multiplied by 0.012.
Global discards (from pelagic longline, trawl, and net fisheries catch minus the global reported landings)	1,135,000	The pelagic longline catch is the reported unweighted average shark catch rate scaled up by total pelagic longline effort. The shark discarding ratios from trawl and net fisheries are equal to the ratios from pelagic longline fisheries. The average weight of sharks caught in pelagic longlines and other gears is 36 kg.	475,000	The pelagic longline catch is the reported effort-weighted average shark catch rate scaled up by total pelagic longline effort. The shark discarding ratios from trawl and net fisheries are 25% the ratios from pelagic longline fisheries. The average weight of sharks caught in pelagic longlines is 36 kg, whereas the average weight of sharks caught in other gears is 9 kg.
Global discard mortality	942,000	The average finning proportion of global shark discards is 0.8, and the average post-release mortality of non-finned discarded sharks is 15%.	450,000	The average finning proportion of global shark discards is 0.8 for pelagic longline fisheries but 0.1 for other fisheries, and the average post-release mortality of nonfinned discarded sharks is 90%.
			246,000	The average finning proportion of global shark discards is 0.8 for pelagic longline fisheries but 0.1 for other fisheries, and the average post-release mortality of nonfinned discarded sharks is 10%.
Current biomass				
Pelagic and coastal sharks	21,565,000	Current global shark biomass is the average unfished global elasmobranch biomass estimated by Jennings et al. (2008) divided by two (half of elasmobranchs are sharks) and multiplied by a depletion level of 50%.	6,470,000	Current global shark biomass is half the biomass level assumed by Worm et al. (2013) multiplied by a depletion level of 30%.
			12,939,000	Current global shark biomass is half the biomass level assumed by Worm et al. (2013) multiplied by a depletion level of 60%.
			25,878,000	Current global shark biomass is double the biomass level assumed by Worm et al. (2013) multiplied by a depletion level of 30%.
			51,756,000	Current global shark biomass is double the biomass level assumed by Worm et al. (2013) multiplied by a depletion level of 60%.
Deepwater sharks	0	The biomass of deepwater sharks is negligible compared to the biomass of pelagic and coastal sharks.	216,000	The biomass of deepwater sharks is 1% of the global shark biomass assumed by Worm et al. (2013).

by simply using an unweighted IUU global average can produce highly inaccurate estimates. For the sensitivity tests, I used the low and high global IUU catch estimates and assumed that the proportion of sharks in the IUU catches was half and double the proportion assumed by Worm et al. (2013). This yielded a global annual IUU shark catch of between 33,000 and 312,000 tons (Table 1).

Catch Conversions

Catches reported in numbers were converted to weight and vice versa. This was done using published estimates of average weights for species within four species groups: pelagic (e.g., Blue Shark *Prionace glauca*, Mako Shark *Isurus oxyrinchus*), large coastal (e.g., Tiger Shark *Galeocerdo cuvier*, Bull Shark *Carcharhinus leucas*), small coastal (e.g., squalidae, *Squatina* spp.), and deep water (Gulper Shark *Centrophorus granulosus*, Deepwater Catshark *Apristurus profundorum*). Substantial

Table 2. Effects of the alternative assumptions about global shark catch and biomass on the annual exploitation rate and assessment of sharks. "Percentage exceeding" is the percentage of the derived exploitation rate exceeding the rebound potential of the 26 shark populations analyzed by Smith et al. (1998). Exploitation rate values larger than the value reported by Worm et al. (2013) (0.067) are shaded grey to assist comparisons.

Global catch (tons)			Current biomass (tons)			
Reported	IUU	Discard mortailty	Pelagic and coastal sharks	Deepwater sharks	Exploitation rate	Percentage exceeding
370,000	33,000	450,000	6,470,000	216,000	0.128	96
370,000	33,000	450,000	12,939,000	216,000	0.065	78
370,000	33,000	450,000	25,878,000	216,000	0.033	26
370,000	33,000	450,000	51,756,000	216,000	0.016	0
370,000	33,000	246,000	6,470,000	216,000	0.097	89
370,000	33,000	246,000	12,939,000	216,000	0.049	59
370,000	33,000	246,000	25,878,000	216,000	0.025	7
370,000	33,000	246,000	51,756,000	216,000	0.012	0
370,000	132,000	450,000	6,470,000	216,000	0.142	100
370,000	132,000	450,000	12,939,000	216,000	0.072	85
370,000	132,000	450,000	25,878,000	216,000	0.036	37
370,000	132,000	450,000	51,756,000	216,000	0.018	4
370,000	132,000	246,000	6,470,000	216,000	0.112	93
370,000	132,000	246,000	12,939,000	216,000	0.057	74
370,000	132,000	246,000	25,878,000	216,000	0.029	22
370,000	132,000	246,000	51,756,000	216,000	0.014	0
370,000	78,000	450,000	6,470,000	216,000	0.134	96
370,000	78,000	450,000	12,939,000	216,000	0.068	81
370,000	78,000	450,000	25,878,000	216,000	0.034	37
370,000	78,000	450,000	51,756,000	216,000	0.017	4
370,000	78,000	246,000	6,470,000	216,000	0.104	89
370,000	78,000	246,000	12,939,000	216,000	0.053	67
370,000	78,000	246,000	25,878,000	216,000	0.027	11
370,000	78,000	246,000	51,756,000	216,000	0.013	0
370,000	312,000	450,000	6,470,000	216,000	0.169	100
370,000	312,000	450,000	12,939,000	216,000	0.086	89
370,000	312,000	450,000	25,878,000	216,000	0.043	52
370,000	312,000	450,000	51,756,000	216,000	0.022	7
370,000	312,000	246,000	6,470,000	216,000	0.139	100
370,000	312,000	246,000	12,939,000	216,000	0.071	85
370,000	312,000	246,000	25,878,000	216,000	0.036	37
370,000	312,000	246,000	51,756,000	216,000	0.018	4

variability is expected from such an averaging exercise, given the natural variability in body weight of the species grouped. For example, within deepwater species, there are some of the smallest (*Etmopterus* spp.) and largest (*Somnious* spp.) sharks (Last and Stevens 2009). Despite this, the number of species used for deriving average weights was very limited and variable (see Table 2 in Worm et al. 2013), with the authors assuming that the median weight of sharks caught in pelagic longlines (36 kg) was representative of the weight of sharks captured by other fishing gears. Furthermore, there are considerable differences in species size composition for most fisheries worldwide. This results from a combination of a range of factors such as fishing gear selectivity, species size segregation, fisher targeting behavior, management regulations, and environmental changes. For example, due to gear selectivity (a gillnet of 6.5/7-inch mesh size is used), the shark fisheries of Western Australia catch mostly neonate and one- to two-year-old Dusky Shark *C. obscurus* but mostly large juvenile and adult Whiskery Shark *Furgaleus macki* (Braccini et al. 2013). As a sensitivity test (Table 1), I assumed an average weight of 36 kg for sharks captured in pelagic longlines but an average weight of 9 kg (the median across the nonpelagic shark species reported in Table 2 of Worm et al. 2013) for sharks captured in other gears.

Discards

Worm et al. (2013) calculated the global shark discards from all fisheries. For this, they used catch rate information of pelagic sharks collected by onboard observer programs from longline fisheries. The authors calculated a total average catch rate for the Pacific, Atlantic, and Indian oceans by pooling catch rate information from different pelagic shark species, time periods (e.g., 1990–1999, 1991–1992, 2006–2007 for the Atlantic Ocean), and fisheries targeting different species (swordfish, tuna, Mahi Mahi *Coryphaena hippurus*, billfish, shark). Such an averaging exercise disregards the variability in catch rates among fisheries, years, shark species, etc., which is commonly reported in catch rate standardisation studies (see Maunder and Punt 2004 for a review). Even for the reported averages (see Table 1 in Worm et al. 2013), the considerable variability within ocean basins (coefficient of variation = 140%, 111%, and 76% for the Pacific, Atlantic, and Indian oceans, respectively) was ignored. In addition, onboard observer programs cover only a fraction (small, in most cases) of the effort exerted by a fishery, so spatiotemporal patterns and differences in fleet fishing practices can be underrepresented (e.g., Gilman et al. 2012). Hence, when scaling up catch rates to total catch, it is imperative to propagate this uncertainty.

To calculate total discards, Worm et al. (2013) multiplied the ocean-specific average catch rate by the total longline effort exerted in each ocean in the year 2000 (note that Worm et al. 2013 did not explain the reasoning behind why this year was chosen). If average catch rate is multiplied by total effort, more accurate discard figures are obtained by multiplying total effort by an effort-weighted catch rate rather than by an unweighted catch rate. For the Indian, Pacific, and Atlantic oceans, total discards were calculated using the unweighted rate of 4.3, 16.5, and 21.2 individuals per 1,000 hooks, respectively (see Table 1 in Worm et al. 2013), whereas the effort-weighted rate is 1.3, 6.8, and 39.6 individuals per 1,000 hooks, respectively. Hence, when multiplied by the total effort exerted in each ocean basin, the weighted catch rate yields different total catch levels (924,000 tons) from the unweighted catch rate (852,000 tons).

The total pelagic shark catch calculated by Worm et al. (2013) was then used to calculate the total shark catch from other fishing gears (net, trawl, and troll). The authors assumed that the proportion of longline shark catch in the total global shark catch was the same as the proportion of large pelagic sharks (52%) in the total reported catch. Hence, Worm et al. (2013) calculated the total shark catch from other gears at approximately 786,000 tons (i.e., 852,000 tons × 0.48/0.52). Total discards were then calculated by subtracting the landed catch calculated above (503,000 tons). This yielded a rounded total for shark discards of 1,135,000 tons (i.e., 852,000 tons + 786,400 tons − 503,000 tons; Table 1). The authors acknowledged that longlines have a high proportion of shark bycatch and discards but assumed that other gears have the same shark discarding ratios as pelagic longlines. Other gears, however, do not necessarily meet this assumption because longlines generally have higher shark bycatch proportions than trawl and net gears (e.g., almost bl50% for pelagic longlines [Francis et al. 2001] but <5% for trawls [Stobutzki et al. 2001] and nets [Baeta et al. 2010]). For non-longline gears, mostly nonpelagic sharks are captured (e.g., McAuley and Simpfendorfer 2003; Clarke et al. 2005) and the proportion of shark bycatch and discards is not necessarily high (Alverson et al. 1994). Had these discarding-pattern differences been considered, the authors' calculations of total discards would have been smaller. For example, for an assumed shark discarding ratio of other gears versus pelagic longline of 0.25, total shark discards were calculated at 475,000 tons (Table 1).

To account for the survival of discarded sharks, an average shark finning proportion (0.8) was applied to the estimated 1,135,000 tons annual discards, yielding 908,000 tons of finned (i.e., dead) sharks. This proportion was obtained from four studies on pelagic species taken in longline fisheries. Finning practices are notorious in fisheries targeted at pelagic species; in fact, the shark fin trade consists of mostly fins from pelagic sharks, particularly Blue Shark (Clarke et al. 2006). Other fisheries do not necessarily remove the fins of discarded sharks, which are mostly discarded whole (e.g., Mandelman and Farrington 2007; Braccini et al. 2012). However, the 0.8 finning proportion was applied to the total discard estimate. This estimate includes the discards from all fishing gears, so discard mortality due to finning was overestimated. For the remainder (227,000 tons of discards), a 15% PRM was applied, so 15% of released sharks died (34,000 tons) and 85% survived (193,000 tons). However, the 15% rate used by Worm et al. (2013) is not representative of all sharks discarded in all fisheries. This PRM estimate was obtained from only two studies on six pelagic species taken in pelagic fishing gears. Other studies for nonpelagic species discarded in nonpelagic fishing gears shows that PRM can range from 0% to 100% depending on the taxon, fishing gear, exposure time, and temperature (e.g., Frick et al. 2010; Braccini et al. 2012). For example, bottom-dwelling species have negligible PRM (<10%), whereas pelagic species have very high PRM (>90%) in demersal gillnet fisheries (Braccini et al. 2012), and PRM of sharks may be nearly 100% in deepwater trawl and large shrimp trawl fisheries. Hence, PRM can be much higher or lower depending on a range of factors. In addition, cryptic and delayed PRM are virtually unknown for most species. Worm et al. (2013) estimated the global annual mortality of discarded sharks at 942,000 tons (Table 1), based on the 0.8 finning proportion and 15% PRM of live discarded sharks. Alternatively, using fishery-specific finning proportions and two extreme PRM values, the global annual mortality of discarded sharks was estimated between 246,000 and 450,000 tons (Table 1).

Irrespective of the validity of the assumptions made for calculating discards, it must be noted that total discards comprised the bulk of the estimated global shark catch (see Figure 2 in Worm et al. 2013). The substantial uncertainty in the estimation of discards highlighted in this review considerably affects the derived exploitation rates and hence the status of shark populations (see below).

Estimation of Total Shark Biomass

Total biomass information was required for assessing shark populations. This is arguably one of the weakest parts of the analysis. For the best-studied species, single-species stock assessment models reconstruct stock biomasses with considerable uncertainty (Walters and Martell 2004). Hence, the uncertainty in the estimation of global shark biomass would be so dramatic that the actual point estimate would be almost meaningless.

Worm et al. (2013) derived total shark biomass from the global-scale biomasses predicted by Jennings et al. (2008). Based on theoretical concepts of macro-ecology, life history, and food-web ecology, Jennings et al. (2008) predicted global unfished elasmobranch biomass at 86,260,000 tons. Worm et al. (2013) then assumed that half of this corresponded to sharks. As expected for estimates derived from global predictions based on theoretical concepts, the predictions carry substantial uncertainty. Jennings et al. (2008) acknowledged the limitation of their predictions, particularly for inshore areas where they may have underestimated biomass. They also showed that their predictions were sensitive to the assumptions about transfer efficiency and predator–prey mass ratios, affecting elasmobranch biomass estimates by sevenfold (Jennings et al. 2008). This was not considered in the analyses by Worm et al. (2013).

Biomass predictions by Jennings et al. (2008) are for the unfished condition, so Worm et al. (2013:195) assumed a 50% depletion level for all shark populations based on a global depletion estimate of current exploited fish stocks by Worm and Branch (2012). This yielded a global current shark biomass of 21,565,000 tons (Table 1). Worm et al. (2013) argued that "50% is a conservative assumption for a highly exploited group." Some shark species, such as many pelagic species (Dulvy et al. 2008), have been highly exploited; however, the depletion level of the vast majority of shark species and populations is unknown. In addition, depletion level is expected to vary considerably among species and population, so assuming a single depletion level for all shark species is misleading. In addition, the assumption of 50% depletion made by Worm et al. (2013) is not supported by the recent global IUCN assessment, where only 25% of sharks, rays, and chimaeras are estimated to be threatened (>50% depletion; Dulvy et al. 2014). Ultimately, if a certain depletion level were to be assumed, exploring the effect of this assumption would be a key aspect of the analysis. Hence, for the sensitivity tests, I assumed depletion levels of 30% and 60% and half and double the biomass levels assumed by Worm et al. (2013) because these fall within the biomass estimates reported by Jennings et al. (2008). This yielded a global current shark biomass of between 6,470,000 and 51,756,000 tons (Table 1).

Jennings et al. (2008) did not consider the biomass of deepwater elasmobranchs on the assumption that the abundance of this group is very low. The biomass of sharks in continental and insular slope waters from 200 to 1,000 m often rivals teleost biomass and is likely far greater than in the pelagic realm (e.g., O'Driscoll et al. 2011). Jennings et al. (2008), however, based their assumption of low deep-sea shark biomass on previous deepwater studies that underestimated biomass because most gears historically used in the deep sea did not sample sharks well. As the deep ocean is increasingly studied and biomass estimates for deepwater sharks are becoming available for parts of the world (e.g., Heymans and Howell 2011; O'Driscoll et al. 2011), it seems that the global biomass of deepwater sharks is not as "insignificant." Ignoring this underestimates the total global biomass of sharks. For the sensitivity analysis, I assumed that deepwater shark biomass was 1% of the global shark biomass assumed by Worm et al. (2013; see Table 1).

Exploitation Rates and Assessment of All Sharks

Sharks' global exploitation rate was calculated by dividing the estimate of total global catch by the estimate of total global biomass. Two independent catch estimates were used: the reconstructed catches (1,445,000 tons) and the median total shark catches (1,700,000 tons) estimated from the fin trade by Clarke et al. (2006). These estimates were divided by the calculated current global shark biomass (21.6 Mt). This assumption was based on the exploitation of commercial finfish stocks and the declines reported for the few shark populations for which quantitative abundance information is available. As pointed out above, quantitative estimates of population depletion for the vast majority of shark species are not available. At a global scale, the IUCN assessment (Dulvy et al. 2014) provides a broader perspective on depletion levels; if only 25% of sharks, rays, and chimaeras are considered vulnerable, then for most species population depletion would be less than 50%.

Worm et al. (2013) calculated a third exploitation rate by averaging available estimates of shark instantaneous fishing mortality (F; 21 populations from 15 species). It is worth noting that the list of F values summarized in Table 5 of Worm et al. (2013) is incomplete and some values do not match those reported in the base run scenarios of the original assessments (e.g., $F = 0.026$ for the Blue Shark 2008 ICCAT assessment versus the $F = 0.020$ reported by Worm et al. 2013). The median global exploitation rates were then compared against the population intrinsic rebound potential (r) of 62 shark species. Shark populations where fishing exploitation exceeded their r were deemed at risk of further depletion and extinction.

Matching the exploitation rate exerted on a population to its biological productivity is the core of a quantitative population assessment. Worm et al. (2013) did this for the 21 shark populations for which exploitation rate information was available and found that half of these populations are at risk as the exploitation rate exceeds r (see Figure 3 in Worm et al. 2013). The authors, however, concluded that "the majority of shark populations will continue to decline under current fishing pressure." They based this claim on their comparison of average exploitation rates against the average r from the 62 species considered. Average comparisons are misleading; assessments must be population specific, where the productivity of a population is compared against the exploitation rate exerted on that population. Furthermore, if both total annual removals due to fishing and total biomass of all species of sharks were accurately known, an estimate of the proportion taken by fishing derived by forming the quotient of these two variables would represent a biomass-weighted population estimate of the average annual exploitation rate, not an average annual exploitation rate that gives equal weight to each population.

Furthermore, fishing gear selectivity was not considered in the exploitation rate calculations. Selectivity is a key management tool that allows shark populations to be exploited sustainably (Walker 1998). In Western Australia, for example, the dome-shape selectivity of gillnets plays a key role for the sustainable exploitation of Dusky Shark, a species with very low productivity (Braccini et al. 2013). More generally, it is now well established that the same population (i.e., same productivity) exploited under different selectivity schedules can sustain different exploitation rates (e.g., Hilborn and Walters 1992; Haddon 2001). Worm et al. (2013) derived r estimates based on the Smith et al. (1998) approach, which assumes an equal rate of F (i.e., equal selectivity) on all age classes above the age at maturity of females (i.e., "knife-edge selectivity"). As acknowledged by Smith et al. (1998), this is problematic for populations exploited by highly size-selective gear. Hence, incorporating gear selectivity is crucial for accurately modeling how exploitation rate affects a population. Furthermore, Worm et al. (2013) did not justify the use of the Smith et al.'s (1998) approach over other approaches (e.g., the modified Euler-Lotka equation method of Myers et al. 1997, which is suitable for determining potential extinction risks). Smith et al.'s (1998) method tends to produce estimates of productivity that are lower than those derived from other methods. Independent of how productivity was estimated, the value of natural mortality (M) used in the calculations can affect the derived productivity estimates. If M values are not representative of a heavily exploited population (i.e., low M values), productivity will be underestimated (Cortés 2007; Gedamke et al. 2007). In addition, the productivity estimates calculated by Worm et al. (2013) and

Smith et al. (1998) and the exploitation rates derived by Worm et al. (2013) and those summarized in Table 5 of Worm et al. (2013) are not directly comparable unless the same method for calculating *M* was used. Unfortunately, Worm et al. (2013) did not mention what method was used to calculate *M*.

Finally, *r* is not just species specific but population specific, and it depends on the accuracy and precision of the parameter estimates used in its calculation. Many of the parameter values used by Smith et al. (1998), however, vary considerably among populations or are biased. For example, the population doubling time for Blue Shark in the North Atlantic (Aires-da-Silva and Gallucci 2007) is less than half that estimated by Smith et al. (1998), who used estimates from different ocean basins and did not consider parameter uncertainty. Hence, a global assessment of shark populations must consider these effects.

Uncertainty and Sensitivity of Assumptions

The importance of acknowledging and modeling uncertainty in fisheries science has been widely recognized for many decades (Hilborn and Walters 1992). Accounting for uncertainty is particularly important for global assessments, where a multitude of data sources are combined. Worm et al. (2013) acknowledged the limited availability of data and the need for making numerous assumptions. However, the authors used average estimates for all of the quantities in their assessment and made just a vague reference to uncertainty, only reporting lower and upper limits for the number of shark individuals killed per year. Uncertainty was not accounted for in their assessment of population status. The assessments were done using point estimates. As reported above, there was no consideration of the many sources of uncertainty introduced in the calculation of catches (reported, IUU, and discards), in the estimation of PRM, in the conversion of weight to numbers and vice versa, in the estimation of global shark biomass, and, finally, in the calculation of global exploitation rates and population productivity.

The assessment also neglected to use sensitivity tests, which is an important step in testing the assumptions applied when modeling natural systems. When uncertainty and different assumptions were considered in the present study, exploitation rates and the status of global shark populations varied substantially (Table 2). Exploitation rates ranged between 0.012 and 0.169. These values could not be compared directly to the study by Worm et al. (2013) because these authors did not identify the 62 species for which *r* was estimated. However, because Worm et al. (2013) included the 26 species analyzed by Smith et al. (1998) within those 62 species, I compared the alternative exploitation rates to the *r* values reported by Smith et al. (1998). Worm et al. (2013) reported that for the majority of shark species the global exploitation rate exceeds their *r* values. In contrast, the present study shows that, depending on the assumptions made for calculating global catch and biomass, the percentage of shark populations for which exploitation rate exceeds *r* ranges from 0% to 100% (Table 2).

DISCUSSION

Global studies provide an overall perspective on relevant topics, so they generally attract considerable attention. By their nature, however, the generalizations made carry substantial uncertainty. Given the considerable public and political interest in the state of fisheries and marine ecosystems in general, uncertainties arising from models and data shortcomings must be presented fully and transparently (Brander et al. 2013).

Uncertainty is pervasive in quantitative assessments (Maunder and Piner 2015), even for species for which a wealth of information is available. For a global quantitative assessment of all shark species, as attempted by Worm et al. (2013), the level of uncertainty is expected to be even larger. Further, quantitative assessments rely on a range of life history, abundance, and exploitation information. Hence, a global assessment of all shark populations based on a single estimate of exploitation rate derived from global catch and biomass estimates is overly simplistic and misleading. Worm et al. (2013) did not test their assumptions, a standard practice in quantitative assessments, and did not provide enough information to fully reproduce their calculations, in particular the exploitation rates. As shown in the present study, the status of shark populations is very dependent on the assumptions made when deriving global shark catch and biomass. It must be noted, however, that the present study is not an alternative to the approach of Worm et al. (2013); rather, it is an attempt to better inform the public debate by showing the high level of uncertainty when attempting a global quantitative assessment of all shark populations.

Shark finning is arguably the most imminent threat for some sharks (mostly pelagic species). Discarding can be highly detrimental in some fisheries (Mandelman et al. 2008), and several reef shark populations have dramatically declined, particularly those occurring in unprotected reefs near populated islands (Nadon et al. 2012 and references therein). However, placing all sharks in the same basket is counterproductive; it can divert resources from those at genuine risk. In the United States, for example, the exaggerated declines in elasmobranch species reported by Baum et al. (2003) and Baum and Myers (2004) resulted in a myriad of petitions from nongovernmental organizations for listing species as "endangered." The conclusions from these studies were overstated (Burgess et al. 2005), so listing of most species was not warranted. However, responding to these petitions and conducting these reviews pulled time and resources away from work on actual endangered species such as Smalltooth Sawfish *Pristis pectinata*. Hence, rather than drawing overall generalizations with unquantifiable levels of accuracy and precision, research effort and scientific advice should focus on identifying populations at most risk and developing population-specific measures to reduce exploitation and increase abundance.

ACKNOWLEDGMENTS

I gratefully acknowledge Norman Hall, Stephen Taylor, Enric Cortés, and three anonymous reviewers for their insightful comments on earlier versions of the article.

REFERENCES

Agnew, D. J., J. Pearce, G. Pramod, T. Peatman, J. R. Beddington, and T. J. Pitcher. 2009. Estimating the worldwide extent of illegal fishing. PloS One 4:e4570.

Aires-da-Silva, A. M., and V. F. Gallucci. 2007. Demographic and risk analyses applied to management and conservation of the Blue Shark (*Prionace glauca*) in the North Atlantic Ocean. Marine and Freshwater Research 58:570–580.

Alverson, D. L., M. H. Freeberg, S. A. Murawski, and J. G. Pope. 1994. A global assessment of fisheries bycatch and discards. Food and Agriculture Organization, Rome: FAO Fisheries Technical Paper 339.

Baeta, F., M. Batista, A. Maia, M. J. Costa, and H. Cabral. 2010. Elasmobranch bycatch in a trammel net fishery in the Portuguese west coast. Fisheries Research 102:123–129.

Baum, J. K., and R. A. Myers. 2004. Shifting baselines and the decline of pelagic sharks in the Gulf of Mexico. Ecology Letters 7:135–145.

Baum, J. K., R. A. Myers, D. G. Kehler, B. Worm, S. J. Harley, and P. A. Doherty. 2003. Collapse and conservation of shark populations in the Northwest Atlantic. Science 299:389–392.

Bonfil, R. S. 1994. Overview of world elasmobranch fisheries. Food and Agriculture Organization, Rome: FAO Fisheries Technical Paper 119.

Braccini, M., R. McAuley, and F. Rowland. 2013. Temperate demersal gillnet and demersal longline fisheries status report. Pages 271–284 *in* W. J. Fletcher and K. Santoro, editors. Status reports of the fisheries and aquatic resources of Western Australia 2012/13: the state of fisheries. Department of Fisheries, Perth, Western Australia.

Braccini, M., J. Van Rijn, and L. Frick. 2012. High post-capture survival for sharks, rays and chimaeras discarded in the main shark fishery of Australia? PloS One 7:e32547.

Brander, K., A. Neuheimer, K. Andersen, and M. Hartvig. 2013. Overconfidence in model projections. ICES Journal of Marine Science 70:1065–1068.

Burgess, G. H., L. R. Beerkircher, G. M. Cailliet, J. K. Carlson, E. Cortés, K. J. Goldman, R. D. Grubbs, J. A. Musick, M. K. Musyl, and C. A. Simpfendorfer. 2005. Is the collapse of shark populations in the Northwest Atlantic Ocean and Gulf of Mexico real? Fisheries 30(10):19-26.

Clarke, M., L. Borges, and R. Officer. 2005. Comparisons of trawl and longline catches of deepwater elasmobranchs west and north of Ireland. Journal of Northwest Atlantic Fisheries Science 37:429–442.

Clarke, S. C., M. K. McAllister, E. J. Milner-Gulland, G. P. Kirkwood, C. G. J. Michielsens, D. J. Agnew, E. K. Pikitch, H. Nakano, and M. S. Shivji. 2006. Global estimates of shark catches using trade records from commercial markets. Ecology Letters 9:1115–1126.

Cortés, E. 2007. Chondrichthyan demographic modelling: an essay on its use, abuse and future. Marine and Freshwater Research 58:4–6.

Dulvy, N. K., J. K. Baum, S. Clarke, L. J. V. Compagno, E. Cortés, A. Domingo, S. Fordham, S. Fowler, M. P. Francis, C. Gibson, J. Martínez, J. A. Musick, A. Soldo, J. D. Stevens, and S. Valenti. 2008. You can swim but you can't hide: the global status and conservation of oceanic pelagic sharks and rays. Aquatic Conservation: Marine and Freshwater Ecosystems 18:459–482.

Dulvy, N. K., and 17 co-authors. 2014. Extinction risk and conservation of the world's sharks and rays. eLife 3: e00590.

Eschmeyer, W. N., editor. 2014. Catalog of fishes: genera, species, references. Available: research.calacademy.org/research/ichthyology/catalog/fishcatmain.asp. (February 2014).

FAO (Food and Agriculture Organization). 2012. FishstatJ-software for fishery statistical time series. Food and Agriculture Organization, Rome.

Francis, M. P., L. H. Griggs, and S. J. Baird. 2001. Pelagic shark bycatch in the New Zealand tuna longline fishery. Marine and Freshwater Research 52:165–178.

Frick, L. H., T. I. Walker, and R. D. Reina. 2010. Trawl capture of Port Jackson Sharks, *Heterodontus portusjacksoni*, and Gummy Sharks, *Mustelus antarcticus*, in a controlled setting: effects of tow duration, air exposure and crowding. Fisheries Research 106:344–350.

Gedamke, T., J. M. Hoenig, W. D. DuPaul, J. A. Musick, and S. H. Gruber. 2007. Using demographic models to determine intrinsic rate and sustainable fishing for elasmobranchs: pitfalls, advances and applications. North American Journal of Fisheries Management 27:605–618.

Gilman, E., K. Passfield, and K. Nakamura. 2012. Performance assessment of bycatch and discards governance by regional fisheries management organizations. International Union for Conservation of Nature, Gland, Switzerland.

Haddon, M. 2001. Modelling and quantitative methods in fisheries. Chapman and Hall/CRC, Boca Raton, Florida.

Heymans, J., and K. Howell. 2011 Do we have enough information to apply the ecosystem approach to management of deep-sea fisheries? An example from the west of Scotland. ICES Journal of Marine Science 68:265–280.

Hilborn, R., and C. J. Walters. 1992. Quantitative fisheries stock assessment: choice, dynamics and uncertainty. Chapman and Hall, New York.

Jennings, S., F. Mélin, J. L. Blanchard, R. M. Forster, N. K. Dulvy, and R. W. Wilson. 2008. Global-scale predictions of community and ecosystem properties from simple ecological theory. Proceedings of the Royal Society B: Biological Sciences 275:1375–1383.

Kulka, D., D. Rivard, and I. Scott. 2012. MSC certification The United States fishery for Spiny Dogfish (*Squalus acanthias*). Moody Marine Ltd, Moody International Certification, Dartmouth, Canada.

Last, P. R., and J. D. Stevens. 2009. Sharks and rays of Australia, 2nd edition. CSIRO Publishing, Collingwood, Ontario, Canada.

Mandelman, J. W., P. W. Cooper, T. B. Werner, and K. M. Lagueux. 2008. Shark bycatch and depredation in the U.S. Atlantic pelagic longline fishery. Reviews in Fish Biology and Fisheries 18:427–442.

Mandelman, J. W., and M. A. Farrington. 2007. The estimated short-term discard mortality of a trawled elasmobranch, the Spiny Dogfish (*Squalus acanthias*). Fisheries Research 83:238–245.

Maunder, M. N., and K. R. Piner. 2015. Contemporary fisheries stock assessment: many issues still remain. ICES Journal of Marine Science. 72: 7-18.

Maunder, M., and A. Punt. 2004. Standardizing catch and effort data: a review of recent approaches. Fisheries Research 70:141–159.

McAuley, R., and C. Simpfendorfer. 2003. Catch composition of the Western Australian temperate demersal gillnet and demersal longline fisheries, 1994 to 1999. Perth, Western Australia. Department of Fisheries Research Report 146.

McAuley, R., C. Simpfendorfer, and N. Hall. 2007. A method for evaluating the impacts of fishing mortality and stochastic influences on the demography of two long-lived shark stocks. ICES Journal of Marine Science 64:1710–1722.

Musick, J. A., G. Burgess, G. Cailliet, M. Camhi, and S. Fordham. 2000. Management of sharks and their relatives (Elasmobranchii). Fisheries 25(3):9-13.

Myers, R. A., G. Mertz, and P. S. Fowlow. 1997. Maximum population growth rates and recovery times for Atlantic Cod, *Gadus morhua*. Fishery Bulletin 95:762–772.

Nadon, M. O., J. K. Baum, I. D. Williams, J. M. McPherson, B. J. Zgliczynski, B. L. Richards, R. E. Schroeder, and R. E. Brainard. 2012. Re-creating missing population baselines for Pacific reef sharks. Conservation Biology 26:493–503.

O'Driscoll, R. L., D. MacGibbon, D. Fu, W. Lyon, and D. W. Stevens. 2011. A review of hoki and middle depth trawl surveys of the Chatham Rise, January 1992–2010. Ministry of Fisheries, Wellington, New Zealand. New Zealand Fisheries Assessment Report 2011/47.

OECD (Organisation for Economic Co-operation and Development). 2005. Why fish piracy persists: the economics of illegal, unreported and unregulated fishing. Organisation for Economic Co-operation and Development, Paris.

Smith, S. E., D. W. Au, and C. Show. 1998. Intrinsic rebound potentials of 26 species of Pacific sharks. Marine and Freshwater Research 49:663–678.

Stevens, J. D., R. Bonfil, N. K. Dulvy, and P. A. Walker. 2000. The effects of fishing on sharks, rays, and chimaeras (chondrichthyans), and the implications for marine ecosystems. ICES Journal of Marine Science 57:476–494.

Stobutzki, I., M. Miller, P. Jones, and J. Salini. 2001. Bycatch diversity and variation in a tropical Australian penaeid fishery; the implications for monitoring. Fisheries Research 53:283–301.

SEDAR (System for Electronic Document Analysis and Retrieval). 2007. Stock assessment report small coastal shark complex, Atlantic Sharpnose, Blacknose, Bonnethead, and Finetooth shark. National Marine Fisheries Services, Silver Spring, MD. NOAA/NMFS 13.

Walker, T. I. 1998. Can shark resources be harvested sustainably? A question revisited with a review of shark fisheries. Marine and Freshwater Research 49:553–572.

Walters, C. J., and S. Martell. 2004. Fisheries ecology and management. Princeton University Press, Princeton, New Jersey.

Watson, R., A. Kitchingman, A. Gelchu, and D. Pauly. 2004. Mapping global fisheries: sharpening our focus. Fish and Fisheries 5:168–177.

Worm, B., and T. A. Branch. 2012. The future of fish. Trends in Ecology and Evolution 27:594–599.

Worm, B., B. Davis, L. Kettemer, C. A. Ward-Paige, D. Chapman, M. R. Heithaus, S. T. Kessel, and S. H. Gruber. 2013. Global catches, exploitation rates, and rebuilding options for sharks. Marine Policy 40:194–204. AFS