Robust estimates of decline for pelagic shark populations in the northwest Atlantic and Gulf of Mexico

invited **response**

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Burgess et al. (2005, this issue) present a critique of two articles describing shark declines in the northwest Atlantic (Baum et al. 2003) and Gulf of Mexico (Baum and Myers 2004), and contend that we have overstated the results of our research. In these two papers, we examined trends in relative abundance for multiple large pelagic shark species. Pelagic sharks include oceanic and coastal (denoted by *) species, and our research focused on 9 of the 17 species we modeled: those we analyzed at the species level (blue *Prionace glauca,* dusky* *Carcharhinus obscurus*, oceanic whitetip *C. longimanus,* silky* *C. falciformis,* tiger* *Galeocerdo cuvier,* white* *Carcharodon carcharias*), and those that dominated the species groups we analyzed (scalloped hammerhead* *Sphyrna lewini,* bigeye thresher *Alopias superciliosus,* and shortfin mako *Isurus oxyrinchus*). We took utmost care to estimate reliable trends in abundance, including the analysis of multiple data sets, the development and implementation of new statistical methods, and the application of extensive auxiliary analyses. Here, we first describe the data sets we considered and detail those we closely scrutinized, since four of Burgess et al's six criticisms concern this aspect of our research. We then address their two remaining criticisms: that we did not take into account all factors that might affect shark catchability nor consider explanations other than overexploitation for the declines. In reviewing their concerns, we still find that our results are robust and our conclusions balanced.

Although the authors imply that our results are flawed because we only utilized data from pelagic longlines, of all gear types this one covers the largest proportion of the northwest Atlantic ranges for each of the focal shark species we analyzed. Pelagic longlines are also the primary method of exploitation for the wide-ranging oceanic sharks in our studies.

Burgess et al. also contend that no one single data set should be used to predict the status of a population. The critical point, however, is that no other data set comes close to containing a similar temporal span for the geographical area covered by the U.S. pelagic longline logbook data we analyzed (Baum et al. 2003). It contains the largest sample size of any data set for each of our focal species in the northwest Atlantic. We emphasize that in general very few data sources exist which are sufficient to determine reliable trends for pelagic shark species. For example, the Japanese pelagic longline logbook data from the Atlantic appears to contain a time series for sharks beginning in 1971, but sharks have actually only been recorded at the species level from 1995 onwards. Attempts to infer trends in abundance for individual species back to 1971, using an extrapolation of current species catch composition, are highly questionable (Nakano and Clarke 2004). In our two studies, we analyzed 5 data sets in addition to the U.S. logbook data: 1 from U.S. Bureau of Commercial Fisheries surveys, and 4 from different scientific observer programs (Baum et al. 2003, Baum and Myers 2004). Burgess et al. mention 3 of these data sets, and suggest that 25 others could have been used to estimate trends in relative abundance: of these, 5 have been (Shepherd & Myers 2005), or are being, analyzed by Myers and colleagues and show declines consistent with our results (although over limited geographical areas), we have been unable to access 7 for proprietary reasons, and 13 contained insufficient temporal span (only 2 to 6 years of data) for this purpose at the time of our study. Hence, the supposed wealth of data sources listed by Burgess et al. is not real.

In our northwest Atlantic shark research, we initially examined four Canadian and U.S. observer data sets (see Baum et al. 2003:391 Note 10), using generalized linear models to standardize the catch rates. However, none of these sources alone contained enough data to assess population trends because of high variability in the fleet, high autocorrelation within trips, and limited and/or nonrandom fleet coverage (<5%); nor could they be combined for analysis because of limited temporal and spatial overlap and differences between the fisheries (e.g., target species) (Baum et al. 2002). We hope that these sources will prove suitable for such analyses in the future as more data become available or as new methods allow us to correct for these inconsistencies.

Burgess et al. also challenge our use of relative abundance indices as indicators of population status. The reality, however, is that data sufficient for sophisticated stock assessment models are available for few species, and of these, only a handful are sharks. Estimating trends in relative abundance is often critical to determining the status of the remaining, data-poor species (e.g., Lande et al. 2003; Myers and Worm 2005). The large pelagic sharks we studied exemplify such cases—they experience high exploitation rates, yet because they are usually taken incidentally and are of low commercial value, their catches have been little monitored and few long-term data sets include them. Indeed our analyses focused on species whose status in the northwest Atlantic was poorly known. We recognize that Cortés et al. (2002) have produced a stock assessment of a large coastal shark complex (>10 species were combined) in which they estimated a substantially lower rate of decline over the same time period we examined. However, we have strong reservations about the way that data sets with contradictory information—which implausibly imply that populations were increasing and decreasing simultaneously—were analyzed and averaged out to produce their estimates. This practice is inappropriate for estimating population trends and warnings against it have been issued repeatedly in the literature (e.g., Hilborn and Walters 1992; Schnute and Hilborn 1993). The other stock assessments Burgess et al. propose we should have referred to did not include any of the same species as those in our research.

We understand that there are technical difficulties associated with using logbook data, and proceeded to do so only after conducting the aforementioned observer data analyses. It is important to distinguish, however, between factors that increase year-to-year variability in catch rates, and hence the uncertainty of estimated trends, and factors that can potentially bias trend estimates. Of primary concern to us in analyzing these data was a trend in whether or not fishers record their shark catches. We therefore developed a method to model only the non-zero catches for each species (or species group; Baum et al. 2003; Kehler and Myers unpublished), and we confirmed the robustness of our results using seven alternative model types and data combinations that represented a wide range of hypotheses about the data (Baum 2002; Baum et al. 2003 supplementary material).

We also considered that when sharks are recorded in logbooks, they may be under- or over-reported, and we previously detailed our sensitivity analyses and the robustness of our results to this potential bias (Baum 2002; Baum et al. 2003 supplementary material). Burgess et al. suggest we should have cross-checked logbook catches from individual sets with those from the National Marine Fisheries Service (NMFS) Pelagic Observer Program. This would have necessitated the use of trip identification codes, which we could not obtain for privacy reasons. As a proxy, we compared mean non-zero catch rates for each of the nine areas in our analysis, for each species, and for sharks overall between the logbook and observer data. Using data from all years (not two years as Burgess et al. stated), we demonstrated that these were comparable for all species recorded from 1986 (see Baum 2002 Fig. 2.28; Baum et al. 2003 supplementary material) except white shark, for which there were no observer records in the 1990s.

The authors posit that the logbook data do not provide information on population trends for the white shark. This species had the least precisely estimated trend in abundance (95% CI: 59–89%) because it was the least frequently recorded among those we analyzed, but we believe ours to be an unbiased estimate (Baum et al. 2003). The fact that confidence intervals overlapped among years is not relevant in interpreting the rate of change and its associated error. Burgess et al. caution that since observers have recorded no white sharks recently, records in the logbook data are misidentifications. The lack of white shark observations is, in fact, probabilistically reasonable: white sharks were recorded in only 3% of sets in the logbook data, and only 3–4% of the fleet had observer coverage. Moreover, a decade earlier U.S. observers had reported 300 from Japanese pelagic longline trips, and according to the observers, the Japanese fleet had their highest white shark catch rates in the same areas (Areas 2–4; they did not fish in Area 1) as fishers reporting in the logbooks (Areas 1–4; Baum et al. 2003). Finally, the authors believe that certain fishers, particularly in the Gulf of Mexico, misidentify white shark. We note that were all data from the Gulf of Mexico removed, the estimated declines for white shark would be greater.

We recognized that misidentification of any shark species could influence our analyses. Therefore, we grouped species that have similar-looking congeners at the genus level, rather than estimating species-specific trends. A telling result was the 70% decline for all unidentified sharks (Baum et al. 2003 supplementary material), indicating that the population declines we reported cannot be attributed to changes in the proportion of sharks that were unidentified.

Burgess et al. observe that in 1993 reporting requirements changed and logbook data no longer contained reports from fishers directly targeting sharks. We agree that this change would have significantly affected overall shark catch rates—making it appear as though sharks declined after 1993—and hence we excluded all the shark targeted sets over the entire time period from our analyses (see Baum 2002:32–33; Baum et al. 2003).

We concur with Burgess et al. that the pelagic longline logbook data did not adequately sample some large coastal shark species, like sandbars (*Carcharhinus plumbeus*) and blacktips (*C. limbatus*), and we recognize that within this group these are the two species for which there are stock assessments. It was because sandbar was poorly sampled and not recorded to species until 1994 that we did not model trends or present any results for this species. Blacktip sharks were modeled only within the group of six large coastal species (Baum et al. 2003), and we drew no inferences about the trends in abundance for individual species within this group. Indeed, their trends may be very different from one another given the different life histories (ages at maturity range from ~7 years from blacktip sharks up to 21 for dusky sharks) of these species.

With respect to our second publication (Baum and Myers 2004), Burgess and colleagues charge that our conclusions were biased because we reported population declines, but did not suggest population increases for species whose catch rates had increased. In fact, we modeled each of the shark species that was caught in both the 1950s and 1990s and found that we did not have statistical power to detect changes in abundance for species with fewer than 25 observations. Species with more observations were all declining. We speculated that the appearance of sandbar sharks in the 1990s could have been the result of several factors including increases in offshore areas, while the disappearance of blacktips could have been due to a decline. However, we reported no results for any of the species that were caught in only one of the two time periods, regardless of whether the species had potentially increased or decreased.

We agree with Burgess et al.'s criticism that changes in pelagic longline fishing methods between the 1950s and 1990s could have affected species' catchability, and we consider this an essential area of investigation (e.g., Ward et al. 2004). Accordingly, Ward and Myers (2005) developed new methods to estimate the effects of depth on catchability. Within the Gulf of Mexico, the greater depths fished in the 1990s (~82–138 m compared to ~53–91 m in the 1950s) would have decreased the catchability of the sharks. Thus, we included species-specific depth correction factors in our models (Baum and Myers 2004). Although habitat models have also been used to correct estimates of fish abundance (by combining information on hook depth with the species' preferences for environmental conditions), Ward and Myers (in press) found that their depth corrections provided better fits to the observed depth distribution. Habitat models fit worse even than models that assumed no effect of depth on catches.

A paucity of data precluded modeling the effect of the other gear changes, which Burgess et al. suggest have altered shark catchability. Any effect of hook type on catch rates would be lessened by the fact that three-quarters of sets in our 1990s data used the same type of hook (J-hook) as those in the 1950s (Baum and Myers 2004). In addition, the impact of changes in hook size and the introduction of circle hooks into the fishery apparently has been either negligible, or to slightly increase shark catch rates (Bacheler and Buckel 2004; Cooke and Suski 2004; Watson et al. 2005), implying that shark declines would be greater if we included these factors in our analysis. A change in leader material from wire to monofilament has substantially increased shark bite-offs, but its effect on shark catch rates (those that are retained) remains ambiguous. The effect appears species-specific: in one of the two known experiments addressing this issue, of the species that were included in our analyses, catches of dusky, scalloped hammerhead, and tiger sharks were each higher on monofilament leaders, whereas catches of shortfin mako were higher on steel (Branstetter and Musick 1993). Sample sizes in both studies were very small ($n \leq 13$ for any of the species we analyzed; Berkeley and Campos 1988; Branstetter and Musick 1993), and we consider this unresolved issue an important area for future research.

Contrary to Burgess and colleagues' speculation that our conclusions were "overly pessimistic" because of the data sets we examined, other relative abundance indices (typically from data sampling small geographic areas) support our findings, or suggest that our original analysis (Baum et al. 2003) may have underestimated the extent of the declines because of the relatively short time series available in the logbook data. For example, the Virginia Institute of Marine Science (VIMS) shark survey has been used to infer declines of 75% in tiger sharks and 80% in dusky sharks (Musick et al. 1993; Musick and Conrath 2002). Campana et al. (2004) found very similar decline rates as we did for blue sharks in Canadian Atlantic waters, but suggested there was a contradiction between our results for Area 7 (offshore from the Grand Banks) and our overall trend for this species (Baum et al. 2003). In fact, it is apparent from Baum et al. 2003 (Figure 3F) how the minimal decline in Area 7 and much larger declines in other areas lead to an overall estimated instantaneous rate of change of –0.066, which equates to the 60% decline between 1986 and 2000 that we reported. Results from the International Commission for the Conservation of Atlantic Tuna's preliminary blue shark assessment are also similar (Anon. 2004). Baum and Myers (2004), which was based on two independent data sets, indicated substantially greater declines for oceanic whitetip sharks (>99% since the 1950s) in the Gulf of Mexico. Scientists there once considered this species a nuisance because of its prevalence around vessels (Bullis and Captiva 1955; Backus et al. 1956), whereas nowadays it is rarely seen (e.g., Russell 1993). Additionally, the longest, standardized, fishery-independent shark survey, conducted off the North Carolina coast since 1972, suggests larger declines for dusky, scalloped hammerhead, and tiger sharks than those we estimated, and the NMFS Northeast inshore trawl survey shows a decline in dusky sharks of 96% since 1974 (Shepherd and Myers, unpublished).

We join others in highlighting the substantial increase in directed and incidental fishing pressure as the single greatest threat to sharks. For example, based on the VIMS survey it was concluded that the dusky shark population had "collapsed because of the western North Atlantic shark fisheries" (Musick et al. 2000). Musick et al. (2000) also noted that the Convention on International Trade in Endangered Species (CITES) Animals Committee "became concerned about the conservation status of some sharks because of their inherent vulnerability, and rapidly expanding shark fisheries around the world." Likewise, it was concern about directed exploitation, and that "bycatch mortality may still high enough to harm shark populations" (Musick et al. 2000) that led to an urgent call for population assessments of elasmobranch species that often appear as bycatch in pelagic commercial fishing operations (NMFS 2000 cited in Beerkircher et al. 2002). The life histories of sharks augment their vulnerability to this overexploitation and others have noted that like some whales and some sea turtles, several sharks may be endangered with extinction (Musick 1999; Musick et al. 2000). Still, despite any direct evidence, Burgess et al. suggest that regime shifts and changes in fishing behavior may have been partly to blame for reported shark declines. Until we see analysis linking these factors to the observed declines, it seems sensible to adopt a precautionary approach and infer that the observed declines are caused by intense exploitation.

We encourage critical evaluation of our research, and welcome the opportunity to refine our estimates, particularly as more detailed studies on technical aspects of the fisheries become available. We also recognize that not all shark species are declining—some small coastal elasmobranchs may in fact be increasing (Shepherd and Myers 2005). In examining Burgess et al.'s criticisms, however, we find that the declines we reported were appropriate. Thus, we believe that several pelagic shark species, including bigeye thresher, dusky, oceanic whitetip, scalloped hammerhead, and silky sharks, should be of high conservation priority in the northwest Atlantic, and until there is reliable information to the contrary, these vulnerable fishes should be managed accordingly.

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