IMPACTS OF INCIDENTAL BYCATCH FROM THE AMERICAN SAMOA-BASED LONGLINE FISHERY TO MARINE TURTLE POPULATIONS  

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SUMMARY

The Protected Resources Division of the Pacific Islands Regional Office (PIRO) of the National Marine Fisheries Service (NMFS) requested a modeling exercise from the Pacific Islands Fisheries Science Center (PIFSC) to assess the population impacts to sea turtle populations from a proposed action in the American Samoa-based longline (ASLL) fishery. This fishery has documented 29 observed green turtle (Chelonia mydas) interactions, 8 leatherback (Dermochelys coriacea) interactions, and 5 olive ridley (Lepidochelys olivacea) sea turtle interactions from 2006 through the second quarter of 2015. These interactions are documented from NOAA on-board observers, who are only present for approximately 20% of all vessel trips. According to the PIRO Sustainable Fisheries Division, the proposed action is for the continued operation of the ASLL fishery in the same geographic location with an estimated annual effort of 29 vessel trips, 5,920 sets, and 17,554,000 hooks. Given these observer data, proposed action, and a retrospective assessment of bycatch abundance (1) the PIRO staff estimated the fishery to result in an expected 23.07 leatherback, 19.95 green, and 10.81 olive ridley interactions annually. Although no turtles of these species have been observed in the catch to date, in addition, the PIRO further estimates 4 hawksbill (Eretmochelys imbricata) and 2 loggerhead (Caretta caretta) interactions to occur annually. Under the U.S. Endangered Species Act, leatherbacks and hawksbills are “endangered” throughout their range, the loggerhead South Pacific Ocean distinct population segment (DPS) is “endangered”, and green and olive ridley turtles are composed of populations with both “threatened” and “endangered” status. Here, I model the effects of the proposed action to determine the mortality to adult nester equivalents (ANE) for each species. My approach combines bycatch and natural mortality, demographics, population sex ratios, and size-to-age models. I then compare these ANE estimates to the entire estimated cohort of adult females in each population and conclude the proposed action itself has an insignificant effect which does not merit any further quantitative modeling.
METHODS

Determining Adult Nester Equivalents

From fishery observer data, PIRO estimated the associated mortality rates for sea turtle bycatch is 90% for greens and hawksbills, 72% for loggerheads, 70.6% for leatherbacks and 29% for olive ridley sea turtles. As sea turtle population monitoring abundance estimates are primarily at nesting beaches, we must first convert anticipated fishery mortalities into ANE (2, 3) to understand population impact. This is calculated from the formula:

$$\sum_{i}^{j} \left( \frac{M_{i}}{n} \right) \psi_{1}^{a} \psi_{2}^{b}$$

where $M_{i}$ is the estimated annual female mortality imposed by the fishery, and $\psi_{1}$ and $\psi_{2}$ represent the natural annual survival of juvenile and subadult turtles. The variable $M_{i} = \mu_{i} \Phi_{i}$, where $\mu_{i}$ is the post-fishery-interaction mortality (i.e. mortality coefficient), $\Phi_{i}$ is the proportion of the population that is female, and $l$ is the estimated interactions. Instead of assigning all bycatch to an average demographic by size (and therefore age) $M_{i}$ is split among $n$ individuals ($i$ through $j$) observed for each species in the bycatch records from 2006 to July 1, 2015. The exponents $a$ and $b$ are the number of years that each turtle remains within each age class before reaching breeding age. This procedure accomplishes an exact demographic match to fishery interaction records, and avoids the statistical issues of non-normality, small sample stochasticity, and poor goodness-of-fit – all of which could bias resulting ANE estimates (2, 4).

When only information on juvenile survival is known, $\psi_{1} \psi_{2}$ simply collapses to $\psi_{1}$ and then the exponent $a$ comprises the entire time remaining to maturity. The sum of $a$ and $b$ can never exceed the age at first reproduction (AFR, 5). However, as the ANE calculates the impact to the adult breeding female demographic – which has been demonstrated to be comprised primarily of neophyte breeders (6-8) in other populations – the formula does not further discount for the natural mortality of adults.

Species Demographic Data

For Pacific leatherback turtles, size-to-age relationships are established through an empirically-fit von Bertalanffy growth model (VBGF, 9), natural juvenile survival is 0.805 (10), AFR = 22 (4), and the female population proportion is 0.65 (11). All of these demographic metrics were conserved from previous fishery assessments for leatherbacks (2, 4). Using the running-sum method (12), data provided by the SWFSC Marine Turtle Ecology and Assessment Program suggests the total breeding population of Western Pacific leatherbacks currently stands at roughly 2,739 females. These numbers include all annual nesting activity, assume a four year remigration interval and a clutch frequency of 5.5 (13), and consider the SWFSC counts (Jamursba Medi and Wermon populations) to be 75% of the leatherback nesting activity in the Western Pacific (2). Unlike previous assessments, we combined boreal summer and winter nesting activity, as satellite tracking shows females breeding in both seasons migrate into the ASLL fishing area (14), and as we do not have any tracking information on juveniles (the most interacted demographic).

For green turtles in the South Pacific, there are no published size-to-age relationships or AFR estimates. However we can estimate such numbers, using recently synthesized information from the recent NMFS-USFWS global green turtle assessment (15), and newly available demographic data from the Hawaii population (5, 16). The global AFR average for green turtles
ranges from 18–33 years (median 25.2), which is similar to the 17-28 years for the Hawaii population. Having described the VBGF parameters \( t_0 \) (-0.181 years) and \( L_{\infty} \) (89.7 cm) for the Hawaii population, and lower and upper estimates for AFR, we then solve for the remaining VBGF parameter, \( k \), that minimizes the error to the modeled sizes at maturity that are the 95%, 97.5%, and 99% of \( L_{\infty} \) (9). Having the result, \( k = 0.173 \), we now can calculate age, given size, for green turtles based on expected global and regional values. For juvenile natural survival, we estimate \( \psi_1 = 0.812 \), which is the average for neighboring populations in Hawaii (5), Australia (17), and the eastern Pacific (18, 19). Based on the recent global assessment, we calculate the total green turtle breeding population at 180,019 females. This number is the total of the Central West Pacific (6,551), East-Indian West Pacific (67,796), Southwest Pacific (82,810), Central South Pacific (3,118), and East Pacific (19,744) populations (15).

For the remaining three sea turtle species, we have little demographic information available. For olive ridleys, we know that one nesting population in the East Pacific alone has greater than 1,000,000 females nest annually (20). South Pacific populations of loggerhead and hawksbill populations are substantially smaller, however. The PIRO estimates that annual hawksbill numbers amount to 5,797, which with a 4-year running sum would equal 23,190 nesting females in the population (D. Golden pers. comm.). The most recent ESA review for loggerheads (21) showed the last four years of nesting at four Australian index rookeries totaled roughly 1,400 nesting females. When demographic information is lacking, we default to a 50:50 male to female population ratio, juvenile annual survival of 0.85, and the time between interaction and first breeding (remaining juvenile stage duration) of 10 years. The interaction rate, and fishery mortality for each species remains set by the PIRO, and the demographic defaults are considered conservatively.

**Fishery Observer Data**

The above ANE formula uses the measured straight carapace lengths from observed fishery interactions to establish bycatch demographics. Measured length data are available for all species for which bycatch is observed (i.e. not for loggerheads or hawksbills), though at varying rates. For green turtles, 29 of 29 turtles entangled or hooked by ASLL fishing vessels were brought aboard and measured before being released or obtained (when dead). For olive ridleys, this number is 5 of the 5 total turtles observed taken in the fishery. However, for leatherbacks only 5 of 8 (63%) of turtles observed taken in the fishery were boarded and measured. As leatherbacks are substantially larger than other sea turtles, boarding the turtles for accurate measurements is often logistically impractical. However, if we only used the measured lengths of boarded leatherbacks, we could be biasing the data to smaller turtles (which are theoretically easier to board), and then perhaps incorporating this artifact into our analyses. If such artifacts were to occur, the effect would be to underestimate the ultimate population impacts, as larger turtles have a greater probability of reaching breeding maturity.

To account for this, we deal with the leatherback observer data in three ways. First, we calculate the ANE only considering the measured lengths from the 5 turtles that were boarded and physically measured with calipers. Second, when measured lengths are not available we examine the observer notes narrative for information on length estimates of turtles that remained in the water and were not boarded. If the observer notes reference the “carapace length” or “SCL” of the turtle, we treat that estimate as a true measurement. If the observer however estimates the “total turtle length” or “approximate turtle length”, we reduce the observer’s length estimate by 20% to discount for the head and tail lengths based on empirical
measurements (2). Third, we treat all observer notes on length estimates as true measurements; the same as if the turtle was boarded and measured with calipers. While I acknowledge that any estimate of a turtle in the water is an educated guess, other things being equal, turtles that are not boarded for measurement are likely larger. This method therefore attempts to account for potential artifacts by using all information available, imperfect as it may be. As a reminder, this is only an issue for leatherback turtles. All other species taken in the ASLL fishery are boarded and actually measured 100% of the time.

RESULTS and DISCUSSION

Figure 1 plots the discounting process from raw fishery interactions to ANE for green and leatherback turtles, i.e. the species for which we have detailed demographic variables, in the ASLL. By comparison to other fisheries (3, 4), fishery effort and interaction rates for the ASLL are relatively low. Discounting for bycatch mortality and female sex ratios, to assess population impacts, further lowers these numbers. As the fishery interacts with predominately juveniles of both species, annual discounting for natural survival drops the anticipated ANE significantly below 1.0, or a single nesting female. For leatherbacks, the median calculated ANE value is 0.55 (Figure 1a), analogous to the fishery incurring a single adult female mortality every 1.8 years. This is the median for leatherbacks, or the second scenario for observer data, which appreciates the observer notes but adjusts the numbers depending on their listed description and stated quality. In the first scenario, which only used validated measurements, the leatherback ANE is 0.17. However, in the third scenario, which treats all observer estimates of non-boarded turtle lengths as accurate, the ANE is 1.7. Considering this range of estimates is analogous to the fishery incurring a single adult nester mortality every 0.6-5.8 years. For green turtles, all animals were boarded and measured and the ANE is simply 0.10 (Figure 1b). This is analogous to the ASLL incurring a nester mortality every 10.1 years.

With considerably less demographic variables available for the remaining sea turtle species, we estimate the ANE at 0.31 for olive ridleys, 0.35 for hawksbills, and 0.14 for loggerheads. As stated in the Methods above, these numbers are conservative (i.e. overestimating fishery impact) with a relatively high estimate of natural juvenile survival, and a relatively short time to maturity, for turtles taken. In reality the turtles likely have a lower natural survival, and a longer time to maturity – both of which would lower the ANE calculation. For hawksbills and loggerheads, since the ASLL fishery has never interacted with a single turtle, so any impact is hypothetical. However, as with the North Pacific longline fisheries (22), it is a reasonable assumption that the ASLL, if it were to interact with loggerheads and hawksbills, would interact with predominantly young juveniles.

To put these numbers into perspective, I then compare the ANE for each species to the total breeding female abundance for the representative reference nesting population to which these turtles belong. For green turtles alone, DNA studies provided by the PIRO (P. Dutton pers. comm.) show the 29 juvenile turtles taken in the fishery are from 5 of the 6 proposed DPSs in the Pacific. These are namely the Central West Pacific, East-Indian West Pacific, Southwest Pacific, Central South Pacific, and the East Pacific (15). (Unsurprisingly, DNA evidence suggests Hawaiian green turtles were not represented in this sample.) As such, as a reference point, the estimated ANE for green turtles is < 0.00001 a proportion of the reference nesting population. For leatherbacks, the ANE constitutes < 0.0002 (1 in 5,019) of the reference nesting population. For olive ridleys, the ANE constitutes < 0.000001 of the females breeding annually in one single
nesting area. And for hawksbills and loggerheads the ANE is < 0.000001 and equal to 0.0001, respectively, of the reference nesting populations.

The analytical approaches I present here yield precise and straightforward conclusions for all sea turtle species considered. The proposed action by PIRO for the ASLL fishery is expected to result in annual interactions for each sea turtle species ranging from 2-23 turtles. However, after discounted for fishery-induced mortality, sex ratios, and natural survival to breeding age, the ANE for all species is well below a single individual (Figure 1). Quantitative population viability analysis (PVA) modeling typically works with discrete whole numbers and not partial individuals. However, rounding the calculated ANE estimates up to a single individual would overestimate the fishery impacts for all species on average by a factor of 5. Besides being trivial in an absolute sense, the calculated ANE is also relatively trivial when compared to each species’ breeding population. Mortality from the ASLL fishery would impact each population by an average of -0.007% (analogous to reducing the intrinsic per capita rate of increase, denoted as r, by 0.00007) of the nesters in each population. I therefore recommend no further quantitative modeling at this time and only suggest qualitative considerations for the fishery actions to these sea turtle populations. There are many uncertainties in such calculations, however, which should be taken into consideration in preparing similar assessments in the future.

First, the ASLL fishery observers have recorded 29 green, 8 leatherback, and 5 olive ridley interactions in total from 2006 through the second quarter of 2015 (Figure 2). These numbers are the observed interactions given partial fishery coverage by vessel-based NMFS observers, and should be considered accordingly. Complete examination of the numbers reveals a relative leveling in the cumulative sea turtle interactions since 2011, reflected in the running average over the time series. However, this overall trend masks two important features: (i) a dramatic decline in green turtle interactions and (ii) an apparent increase in leatherback interactions (Figure 2). Both of these trends should be explored in greater detail as this may reveal more ecological information that is relevant for an assessment of bycatch impacts. Leatherbacks had both the highest ANE (Figure 1a) and the highest impact proportional to the estimated breeding female population. Any increase in leatherback bycatch should be closely monitored.

Second, a significant aspect of the impact assessment is the ANE which relies on demographic information at the population- and individual-level. For all species, greater attention to juvenile vital rates in the wild, age to maturity, and VBGF growth models, in particular will be of great benefit. However, especially for leatherbacks, more attention should be focused on the measured lengths of turtle interactions. For only 5 of 8 interactions were turtles brought on board fishing vessels and measured with calipers. The remaining 3 of 8 turtles were set free while they remained in the water, and their lengths were estimated by the observer from the ship. As discussed above in the Methods, these estimates are likely subject to significant inaccuracies. The PIRO as a result might consider additional technologies to provide fishery observers (wearable cameras, laser measure devices) to aid accurate measurements of turtles in the water. Due to their large size, it is to be expected that entangled or hooked leatherbacks will frequently not be boarded. However, in these scenarios it is important to obtain as accurate data as is possible to improve the accuracy of information associated with interactions and our assessment of the population level impacts.

Third, also with respect to demographic issues, post-fishery-interaction survival probabilities are likely positively correlated with turtle length. As discussed for leatherbacks, 5 of
8 turtles were measured by fishery observers on board the fishing vessel. These turtles had an average length of 73.6 cm (range 57.5-87 cm), and all of them died, thus achieving a mortality coefficient of 100%. The 3 remaining that were not boarded and not measured were by various estimations larger than these 5 (see above). These 3 turtles all were released alive, and were given a much greater probability of survival. The mortality coefficient for these 3 turtles was 21.7%, which is an average of individual mortality coefficients derived from the location of the gear interaction and the amount of gear remaining when the turtle was released (23). As it is currently applied, each species is given a single mortality rate, and this metric is then fed into the ANE formula (as $\mu_i$) and applied to all demographics. Since the current ANE formula already addresses the importance of demographics of bycatch in terms of natural survival, incorporating the demographic differences in fishery mortality is sensible. The impact of this revision is easily assessed, now, by applying the mortality coefficients assigned to each interaction event by the PIRO in the ANE formula. This procedure would replace using a species average for $\mu_i$. For leatherbacks, the result of evaluating mortality at the individual level changes the ANE estimate to 0.25-0.33-0.58, across the three length estimation scenarios. Previously the ANE ranged from 0.17-1.71 for all length estimation scenarios. Future revisions of fishery-related mortality may take this aspect of demographics and its influence to post-interaction survival into account.

Fourth, it is important to frame the ANE for each species within the context of the reference population(s) of breeding females. For olive ridleys, more than 1,000,000 females nest at one index site in the Eastern Pacific and perhaps further information is not required. For green turtles, the recent global assessment calculated detailed population-level abundance numbers for each DPS. These kinds of data are not available for other species, but are incredibly valuable when doing such analyses such as the current assessment. For leatherbacks, hawksbills, and loggerhead populations that forage in the ASLL region of the South Pacific, much less is known. Future fishery assessments will benefit from robust efforts to accumulate data and estimate nesting abundances across the region at the population level.

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**Figure 1.** Annual nester equivalents (ANE) calculation for expected numbers of leatherback and green turtles taken in the American Samoa-based Longline (ASLL) fishery under the proposed action. The ANE procedure employed here incorporates an exact demographic match to the observed interactions, and relies on accurate length measurements by fishery observers of bycaught turtles and conversion of these recorded lengths to ages, through models. (A) After discounting for fishery survival, male population proportions, and natural survival, 23.07 expected leatherback interactions become 0.55 adult breeding females. I consider this ANE estimate the median, as the number is affected by the treatment of leatherbacks that were not boarded, but to which observer notes were applied variously to estimate turtle lengths (see Methods). Discounting these notes lowers the ANE to 0.17 but accepting the notes as 100% accurate raised the ANE to 1.7. At present, I recommend the median scenario, with the caveat that the PIRO implement measures to increase the utility of estimated lengths from all turtle interactions. (B) Considering 19.95 expected annual interactions, the ANE estimate for green turtles is 0.10. Without such detailed demographic data, additional ANEs were conservatively calculated for olive ridley, hawksbill, and loggerhead sea turtles assuming annual juvenile survival of 0.85, and 10 years between fishery interaction and first breeding. The maximum ANE for this group did not exceed 0.35.
**Figure 2.** Observed, annually cumulative ASLL sea turtle bycatch from 2006 through second quarter of 2015. Cumulative bycatch numbers do not reflect any adjustment from partial observer coverage in the fleet, and are merely the observed interactions since observer coverage began in 2006. As such, the numbers of green turtle interactions (green line) peaked in 2011 and have since declined. Leatherback numbers have increased since 2010, a trend that deserves greater attention. The running average reflects that the total sea turtle bycatch seems to have stabilized since 2011, but this does not account for differences between species.
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