

Assessment of the population-level impacts of potential increases in marine turtle interactions resulting from a Hawaii Longline Association proposal to expand the Hawaii-based shallow-set fishery¹

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Abstract

The Hawaii Longline Association is proposing to expand the Hawaii-based shallow-set longline fishery, which will likely increase the level of sea turtle interactions.

Leatherbacks and loggerheads are the most common turtle species interacting with this fishery and the majority of interacting turtles are released alive (100% since 2004) with varying degrees of injury. The post-interaction mortality rates are estimated at 0.205 for loggerheads and 0.229 for leatherbacks. In this study I estimate the increase in quasi-extinction risk to turtle populations from mortalities associated with this fishery. I use diffusion approximation methods to estimate the mean quasi-extinction risk using a quasi-extinction threshold of 50% of current population size and a time threshold of 63 yr for leatherbacks and 100 yr for loggerheads. As the diffusion approximation uses nest census data, only units of adult females are considered and the turtles interacting with the fishery are converted to adult female ‘equivalents’ by assuming a 65% female sex ratio and mean reproductive values of 0.41 for loggerheads and 0.85 for leatherbacks.

Nesting data from Japan (loggerheads), Jamursba-Medi, Papua, Indonesia (leatherbacks) and Costa Rica (leatherbacks) were used. Results of this study indicated that to minimize increased risks of quasi-extinction, mortalities of adult female (or ‘equivalent’) Japanese loggerheads should be less than 4, from Jamursba-Medi leatherbacks. the mortalities should be less than 3 adult females, and for the Costa Rica leatherback population, no adult females should be killed. The proposed interaction levels of the expanded fishery are 46 loggerheads and 19 leatherbacks. These levels are estimated to result in 2.51 adult female mortalities for loggerheads in Japan, 1.56 adult female mortalities for leatherbacks from Jamursba-Medi, and 0.12 adult female leatherbacks from Costa Rica.

Introduction

Predicting absolute extinction in populations is complicated by the unpredictable behaviors of small populations and it is a common practice in conservation biology to use quasi-extinction thresholds in population viability assessments (Morris and Doak 2002). Snover and Heppell (in review) present a quasi-extinction risk index called susceptibility to quasi-extinction (SQE) that can be used to classify populations based on relative risks. Using population simulations, they show that the method is robust in assessing actual risk (in terms of a binary assessment of at risk or not at risk), assuming that current conditions remain the same over the time period of the projection. As they use long time frames of 3 generations (following IUCN criteria) they clarify that SQE values are primarily useful as an index for comparing populations and assessing the impacts of increased mortalities by comparing SQE values between perturbed and non-perturbed populations. Here I apply this technique to nest census data for Pacific loggerheads and leatherbacks to assess the impacts of increased mortality expected to result from a proposed expansion of the Hawaii-based shallow-set longline fishery. The analyses presented here are designed to be a tool for managers to assess how different levels of fishery interactions may affect the extinction risk of marine turtle populations.

Data and populations considered

Leatherbacks

Leatherback nesting data for Jamursba-Medi, Papua, Indonesia are reported in Hitipeuw et al. (2007) for 1981, 1984-1985, 1993-1997, and 1999-2004. Nesting occurs year-round for leatherbacks in this region, with peaks from April to October. As not all months were surveyed in all years, Hitipeuw et al. (2007) used information on the

proportion of annual nesting that occurs in each month from year-round surveys to estimate the number of nests between April – October for all years. Data for all of 2005 and 2006 through August are in a Report to the Western Pacific Fishery Management Council (WPFMC).² I used the same method as Hitipeuw et al. (2007) to estimate nesting in September and October 2006 resulting in a nesting dataset for the time period of 1993-2006 for this region. The data point for 1998 was estimated as the mean of 1997 and 1999 (Fig. 1; Dennis et al. 1991). I used the value of 5.5 nests per female (Martínez et al. 2007) to estimate the number of nesting females.

For the eastern Pacific, nesting leatherback data for Parque Nacional Las Baulas, Playa Grande, Costa Rica are reported in Tomillo et al. (2007) for the 1988/1989 to 2003/2004 nesting seasons (Fig. 1). As there is a saturation tagging program at this beach, all females are identified and the census data are numbers of females nesting per year.

Loggerheads

Loggerheads found in the North Pacific are predominately from nesting beaches in Japan. Genetic analyses of loggerheads taken in the Hawaii-based longline fisheries indicate that 100% of these turtles are from the Japanese nesting populations (P. Dutton, personal communication). Nesting data for Japanese loggerheads are from the Sea Turtle Association of Japan (STAJ; unpublished data provided to the WPFMC) and Kamezaki and Matsuzawa (2002). The STAJ data are from 1998 to 2007 and these were estimated back to 1990 using data from Kamezaki and Matsuzawa (2002). Thirty-three Japanese nesting beaches have been monitored annually for nest counts since 1990 (Kamazaki and

² Leatherback conservation at Warmon Beach, Papua-Indonesia, Final report for the period of November 2005 – October 2006 (Ref No.: 04-WPC-034)

Matsuzawa 2002)³. The 1998 to 2007 STAJ data represent all Japanese nesting data. For 1998 and 1999, the 33 beaches in Kamazaki and Matsuzawa (2002) represented 51.7 and 52.6% of the total nesting reported by the STAJ. I assumed that the 33 beaches with nesting data reported from 1990 to 1998 (Kamazaki and Matsuzawa 2002) represented 52.1% of total nesting in Japan and used this ratio to extend the STAJ time series back an additional 8 years (Fig. 2).

Post-interaction mortality rates

Since the reopening of the Hawaii-based shallow-set fishery in 2004, all of the loggerhead and leatherback turtles taken have been released alive. NMFS convened a workshop to elicit expert opinion on post-interaction mortality rates based on the severity of the injury to the turtle (Table 1; Ryder et al. 2006). Using the observer data from the shallow-set fishery since 2004, each turtle taken in the fishery was assigned a post-interaction mortality rate to assess a mean post-interaction mortality rate for each species (Memorandum to W.L. Robinson 1 Feb. 2008⁴). The overall mean post-interaction mortality rate for the Hawaii-based shallow set fishery from 2004 to 2007 is 20.5% (95% C.I. 14.7 – 26.2%) for loggerhead turtles and 22.9%⁵ (95% C.I. 12.6 – 33.1%) for leatherback turtles. Many of the injury categories in Table 1 were not found in the loggerhead and leatherback takes in the shallow-set fishery since 2004. Between 2004 and 2007 16 leatherbacks and 45 loggerheads interacted with the fishery and as those numbers grow it is possible that we will see more turtles in different injury

³ Previous to 1990, less than 20 beaches were monitored regularly for nesting. Between 1989 and 1990 the number of beaches monitored nearly doubled, therefore data were estimated back to 1990.

⁴ Memorandum from Chris Yates to William Robinson dated 1 February 2008; subject: Observed captures and estimated mortality of sea turtles in the Hawaii shallow-set longline fishery, 2004-2007.

⁵ This number is slightly different from that presented in the memorandum as a rounding error was found. The number reported in the memo is 22.3% but the correct number, for interactions between 2004 and 2007 is 22.9%.

categories. With these small numbers, even a single event of a serious injury with high a post-interaction mortality rate would alter the mean post-interaction mortality rates reported here, hence these numbers should be monitored as the fishery progresses to ensure they do not change substantially. Large increases in mean post-interaction mortality rates will alter the results presented here.

Nearly half of the leatherbacks were externally hooked and released with the hook and substantial line still attached. The remaining leatherbacks were primarily externally hooked and released with the hook and little line or with no gear. Of the 16 leatherbacks interacting with the shallow-set fishery between 2004 and 2007, only one was mouth-hooked. For loggerheads, the highest interaction category was category III (hooked in soft tissues of the mouth or esophagus above the level of the heart) and most of these were released with all gear removed. The next highest category was externally hooked and again most of these were released with no gear attached.

Population-specific interactions with the fishery

For loggerheads, the current interaction limit for the Hawaii-based longline fishery is 17 and in the proposed expansion of the fishery it is estimated that as many as 46 would interact with the fishery. For leatherbacks, the current interaction limit is 16 and the expected increase of interactions is 19. The break-down of these numbers in terms of expected interactions associated with each of the nesting populations is considered here.

For loggerheads this is trivial as we know from genetics that 100% of these turtles interacting with the shallow-set fishery are from Japan.

For leatherbacks, Table 1 in Dutton et al. (2007) shows the approximate annual number of nests per beach for the Western Pacific metapopulation. From this table, the Jamursba-Medi nesting assemblage represents ~38% of the nesting in this region. Genetics data for leatherback turtles taken in the Hawaii-based longline fisheries suggest that 6% of takes are from the East Pacific and 94% of takes are from the West Pacific (P. Dutton, personal communication). If all West Pacific leatherbacks are equally likely to migrate to the North Pacific, then 35.7% (0.38×0.94) of leatherbacks interacting with the Hawaii-based shallow-set longline fishery are likely to be from Jamursba-Medi. However, based on satellite telemetry studies, it appears that the direction of post-nesting migration is related to the season of the nesting, with winter nesters heading southeast to the high latitudes of the South Pacific Ocean (Benson et al. 2007a). Summer nesters head either northeast towards the eastern North Pacific Ocean or west to the South China Sea (Benson et al. 2007b). Again from Table 1 in Dutton et al. (2007) the vast majority of summer nesting in this region occurs at Jamursba-Medi with very low levels of summer nesting elsewhere. Hence, because of the nesting seasonality, it is possible that the adult female leatherbacks that interact with the Hawaii-based longline fisheries are predominantly from Jamursba-Medi. The satellite telemetry studies are only of adult females and the migration patterns of juveniles and adult males are unknown. To account for the possibility that the Jamursba-Medi nesting assemblage is disproportionately represented in the shallow-set interactions, I consider the midpoint of the range 38-100% = 69% as the proportion of the West Pacific leatherbacks interacting with the Hawaii-based shallow-set fishery sourcing from the Jamursba-Medi nesting assemblage. This

results in 65% (0.69×0.94) of the total leatherbacks interacting with the fishery being attributable to Jamursba-Medi.

For the East Pacific, Martinez et al. (2007) found a total of 346 leatherbacks nesting in Mexico during the 2003-2004 nesting season and Tomillo et al. (2007) found a total of 188 females nesting in Costa Rica. Assuming 5 nests per female and a mean remigration interval of 2.5 yr (Spotila et al. 1996), I estimate 1335 adult female leatherbacks for the Eastern Pacific, with 14% from Costa Rica. Hence, 0.8% (0.14×0.06) of leatherbacks interacting with the Hawaii-based shallow-set longline fishery are likely to be from the Costa Rica population.

Analytical approach

Diffusion Approximation

I used the diffusion approximation approach discussed in Snover and Heppell (in review) to assess the status of the nesting populations considered here. The methods used to estimate parameters for diffusion approximation are reported in Dennis et al. (1991) and Morris and Doak (2002). These methods are based on a model for exponential population growth in a randomly varying environment (Morris and Doak 2002)

$$(1) \quad N_{t+1} = N_t \lambda_t$$

where N is the population size, t is time and λ_t is the population growth rate in year t .

Two key parameters estimated by this method are $\hat{\mu}$, the arithmetic mean of the log population growth rate, and $\hat{\sigma}^2$, the variance of the log population growth rate which accounts for sources of variability, including environmental and demographic stochasticity and observation error (Dennis et al. 1991, Morris and Doak 2002). These

parameters are used to make inferences regarding total population growth rates and quasi-extinction risks.

Selection of quasi-extinction threshold

Merrick and Haas (2008) applied a diffusion approximation analysis to loggerhead turtle bycatch from the Atlantic Sea Scallop Fishery and they used a quasi-extinction threshold (QET) of 250 adult females for a population with a current estimate of 34,881 adult females (~0.7% of current population size). Looking at time thresholds of 25, 50, 75 and 100 yr, a population of that size would have to decline at rates of 20, 10, 7, and 5 % per year respectively to reach the quasi-extinction threshold. Not surprisingly, they found essentially zero risks of reaching the quasi-extinction threshold, and when they considered the impact of removing the mortality of 75 adult females that the fishery is estimated to kill each year, obviously it could not lower the quasi-extinction risk (there cannot be a risk of < 0). Hence, to achieve the resolution necessary to detect changes in risk of quasi-extinction, it is essential to select a reasonable level of QET for which non-zero values are obtained. A QET of 50% is consistent with the IUCN listing criteria, that a species is considered vulnerable if it is likely to decline by 50% of its current size over 3 generations⁶, and it is the value I use in this analysis.

Selection of the time threshold

Similarly, I again follow the IUCN listing criteria which suggests time thresholds of 3 generations or 100 yr, whichever value is smaller⁶. To estimate generation time for leatherbacks, I used the mean value of age to sexual maturity of 14 yr (Zug and Parham

⁶ 2001 IUCN Red List Categories and Criteria version 3.1,
<http://www.iucn.org/themes/ssc/redlists/RLcats2001booklet.html>

1996) and an adult survival rate of 0.90 to estimate a generation time of 21 yr. or a 3 generation time period equal to 63 yr (Snover and Heppell in review). Age to maturity for the Japanese loggerhead population is not understood. This parameter is estimated at >30 yr for Atlantic loggerheads (Snover 2002), however Japanese loggerheads nest at a smaller size (Hatase et al. 2004) and potentially at a younger age. If age to maturity is assumed to be 27 and adult survival rate is 0.90, 3 generations is ~101 yr, hence I used the time period of 100 yr as suggested by the IUCN when 3 generations is >100yr.

Susceptibility to quasi-extinction

Following Snover and Heppell (in review), I used the parametric bootstrap estimation procedure from Morris and Doak (2002) to compute quasi-extinction risks to quasi-extinction thresholds (QET) of 50% of current population size based on the 95% CI of $\hat{\mu}$ and $\hat{\sigma}^2$ for a time horizon of $T = 3$ generations or 100 yr, whichever value is smaller. Snover and Heppell (in review) define susceptibility to quasi-extinction (SQE) as the proportion of the parametric bootstrap replicates that indicate a >90% chance of dropping below a pre-defined quasi-extinction threshold (QET). Using population simulations, Snover and Heppell (in review) demonstrated that SQE values greater than 0.4 indicate that a population is at risk of being reduced to the quasi-extinction threshold (QET) level used. At this critical value (0.40) ‘Type I’ errors (considering a population to not be at risk when it is) occur at a rate of about 10% and reducing the critical value to 0.3 lessens this rate at the expense of increased ‘Type II’ errors (considering a population to be at risk when it is not). The choice of only using replicates that indicate a >90% chance of dropping below the QET was somewhat arbitrary and values other than 90%

could be used, however, new critical values would need to be established for different values.

I have found that this concept of SQE as defined above is not transparent in practical management applications. Hence, I am using the mean value of the parametric bootstrap instead. This has the advantage of being easily interpreted as the mean risk of reaching the quasi-extinction threshold in the specified timeframe. I used the same population simulations as in Snover and Heppell (in review) to determine that the range of critical values for this metric is 0.65-0.75. In other words, populations with a mean risk of quasi-extinction > 0.75 are at risk, populations with a mean risk < 0.65 are not at risk and populations with means between 0.65 and 0.75 are potentially at risk. This definition of SQE classifies populations the same as that of Snover and Heppell (in review) while providing an index for quasi-extinction risk that is more tractable to managers.

Once a baseline SQE was established for each nesting population, I used this mean risk of quasi-extinction in conjunction with an approach similar to Kaplan (2005). Kaplan (2005) estimated that 181 eastern Pacific leatherbacks were killed by the international longline fleet in 1998. Spotilla et al. (2000) estimated a population size of about 1690 adult females in the eastern Pacific. Hence, assuming all mortalities were adults and a 50% sex ratio, Kaplan (2005) calculated that of the total adult female mortality rate, 0.054 per year arises from the international longline fleet. He added this mortality to his estimate of population growth rate, r , to indicate what the population growth rate would be if all mortality from longline interactions were removed. With assumptions regarding age-class and sex ratios of turtles in the bycatch, a similar method

can be applied here. Assuming a constant $\hat{\sigma}^2$, new values of $\hat{\mu}$ can be used in the diffusion approximation to establish a new SQE value to determine if mortalities from fisheries bycatch are likely to affect the persistence of the population.

I considered the SQE values estimated for the datasets at QET=50% of the current population size (reduction of 50 % from current population size) as baseline values, resulting in estimates of $\hat{\mu}_b$, $\hat{\sigma}_b^2$ and population growth rate r_b , where the subscript b denotes baseline. Following recommendations in Snover and Heppell (in review), I used a running-sum of 3 yr and current population size, n_o , was estimated as the sum of the last 3 yr of data. I considered the effect of $m_i = 1, 2, 3 \dots 10$ additional annual adult female mortalities on SQE values. The intrinsic rate of population increase (r) is calculated as

$$(6) \quad r = \mu + \frac{\sigma^2}{2} \text{ (Dennis et al. 1991),}$$

hence for each value of m_i , a new value of $\hat{\mu}_i$ was estimated as

$$(7) \quad \hat{\mu}_i = \left[r_b - \left(\frac{m_i}{n_o} \right) \right] - \frac{\sigma_b^2}{2}.$$

New confidence intervals around $\hat{\mu}_i$ were constructed using the standard error of $\hat{\mu}_b$ and new susceptibility to extinction values were estimated for each m_i using the Dennis et al. (1991) method. The bootstrap results were smoothed by fitting a logistic curve to the results

$$(8) \quad SQE_{new} = \left[1 + \left(\frac{1}{SQE_0} - 1 \right) \exp(-bA) \right]^{-1}$$

where SQE_0 is the base value of SQE, SQE_{new} is the new value of SQE, A is the number of additional adult female mortalities and b is a fitted parameter that describes the rate of increase of the curve.

Reproductive Values

Sizes of loggerhead turtles interacting with the shallow-set fishery range from 40 to >95 cm carapace length, with an approximate mean of 64 cm carapace length. Therefore, most of the loggerhead turtles interacting with this fishery are juveniles. As the above analysis only deals with adult females (because these are the only portion of the population being censused) we need to equate these juveniles to adult females using reproductive values. To truly assess an individual's reproductive value, precise information on survival rates, fecundity rates, age and individual growth rates are needed. As we don't have this information for Japanese loggerheads, I created a range of population models assuming different ages to maturity, size at maturity and survival rates (Table 2). I used age-based Leslie matrix models where the dominant left eigenvector contains the reproductive value for each age class. Each turtle interacting with the shallow-set fishery from 2004-2007 for which size was recorded was assigned an age, based on the growth curve used in each model, and the corresponding reproductive value; a mean of these reproductive values was calculated. For the models analyzed, mean reproductive values ranged from 0.22 to 0.41. In a letter to the Council, the Pacific Islands Regional Office (PIRO) indicated that they would use the value 0.41 in their jeopardy assessment⁷ and so I use that value in this assessment.

⁷ Letter from Lance Smith of PIRO to Eric Kingma of WPRFMC dated 24 April 2008, 'Variables for Estimates of Annual Adult Female Mortalities in Shallow-set Fishery'.

For leatherbacks, the estimated lengths range from 100 to 192 cm carapace length with 11 turtles represented (measurements are not available for all leatherbacks interacting with the fishery). Western Pacific leatherbacks reach maturity as small as 126 cm carapace length (Zug and Parham 1996), hence ~82% of the leatherbacks interacting with the fishery are potentially mature. Given the high proportion of adults represented in the bycatch and the uncertainties associated with estimates of growth, survival rates and fecundity parameters, the PIRO⁷ recommends using a mean reproductive value of 0.85 to account for the fact that most but not all of the leatherbacks interacting with this fishery are likely to be adults.

Sex Ratios

In addition to reproductive values, a sex ratio of the turtles interacting with the fishery needs to be assumed to estimate the proportion of females in the bycatch. There are no sex ratio studies for Japanese loggerheads or West Pacific leatherbacks, however studies of other populations typically find a female bias in sex ratios. Table 3 summarizes sex ratio studies for other loggerhead and leatherback populations. Based on this information, PIRO has decided that it will use 0.65 female as the sex ratio for both loggerheads and leatherbacks.⁷

Potential applications of approach to management decisions

As annual takes of adult females are increased, SQE values increase accordingly. The rate of increase in SQE values is closely linked with current population size, small populations will be more impacted by additional takes than large ones. There are numerous ways to consider the point where the increase in SQE, and the corresponding interaction level, becomes unacceptable. In considering which method to use,

transparency and ease of application are important for management decisions. I will suggest and implement a method here with the understanding that other approaches can be considered.

As a goal in determining take levels for endangered species is to not appreciably reduce their likelihood of their survival and recovery, I argue that we want to use take levels consistent with very small changes in SQE. Consider the value 1-SQE. A cutoff percentage of this value, for example 1 - 10%, can be used whereby fatal takes of adult females that increase SQE by $> 0.01(1-SQE)$ to $0.1(1-SQE)$ is considered an unacceptably large increase. 1% of 1-SQE is likely a very conservative value while 10% of 1-SQE is likely liberal and the exact value (whether in this range or outside its bounds) that results in jeopardy is a management decision that must be made with consideration of other threats to the populations (e.g. threats that may not be apparent from the nesting beach trends). The use of 1-SQE has the advantage of being conservative for populations with high SQE and less so for low SQE values. For example, for $0.05(1-SQE)$, if $SQE = 0.99$, SQE cannot increase by more than 0.0005, whereas if $SQE=0.01$, this value can increase by up to 0.0495. To apply this method, I used the parametric bootstrap procedure described above with 10000 repetitions to determine new SQE values for 1 to 10 additional adult female mortalities (Fig. 3). These values were fitted with logistic curves (Eq. 8) and the resulting values of b were 0.027 for Jamursba-Medi, 0.174 for Costa Rica, and 0.017 for Japan.

Results and Discussion

All three of the Pacific populations considered here appear to be declining with μ values < 0 (Table 4) and the SQE values were all above the critical range of 0.65-0.75 for

QET = 50% (Table 5). For the Costa Rica population, an annual loss of >1 adult female beyond the current level resulted in excessive (as defined in this paper) increases in SQE (Table 5). The results for the larger Jamursba-Medi nesting population indicated that adult female mortalities of less than 4 (and ideally less than 2 to stay under the 0.05(1-SQE) range) would have a minimal impact on SQE. Of the three populations, the Japanese loggerhead population was the largest and the results for this population indicated that adult female mortalities less than 7 (or 3 for the 0.05(1-SQE) range) would have a minimal impact on the populations risk of extinction.

These numbers are small and may seem to suggest that this method is overly conservative, however these populations are all small and declining and the allowable fatal interactions from them should reflect their status. The values above are in terms of adult females, and once these numbers are placed into a context of total interactions, accounting for sex ratio (0.65 female for both species), reproductive value (0.41 for loggerheads and 0.85 for leatherbacks), and the fact that most turtles interacting with this fishery will survive (mean post-interaction mortality rates of 0.205 and 0.229 for loggerheads and leatherbacks respectively), the total interactions that equate to the numbers of adult female interactions (Table 5) fall within the ranges proposed for expansion of the fishery (Table 6). For example, an interaction level of 46 loggerheads results in ~3 adult female mortalities (Table 6) and the range proposed by the methods presented here is ~1 to ~7 adult females (Table 5).

Conclusions

The SQE values calculated for a nesting beach are strongly and negatively correlated with current population size and population trend (in terms of abundances on

nesting beaches; Snover and Heppell, in review) and these parameters obviously change over time. If the populations assessed here continue to decline, detectable changes in SQE may be found with fewer adult female losses, and the reverse of this is true as well. Hence it is advisable to periodically assess the status of the populations interacting with the longline fisheries.

The population growth rates and SQE values considered here apply only to the nesting female segment of the population. For most populations, this is the only portion censused for trends and we cannot assume that what is happening on the nesting beach parallels the rest of the population is not appropriate and caution needs to be applied in interpreting these results. For example, the Japanese loggerhead trends have historically been cyclic with periods of increases alternating with declines. The nesting abundances have been increasing since 1997, but the two most recent years of data for this population are suggestive of a substantial decline in numbers. No real inferences can be made on only two years of data, however the mortalities of juveniles off the Baja peninsula of Mexico are well documented (Peckham et al. 2007) and these mortality levels are relatively recent (increasing to current levels over the last 15-20 years or so; H. Peckham pers comm.). The current declining numbers in the Japanese loggerhead trends may simply be the start of another cycle, however it may also be that the reduction of the juveniles in Baja is just now being manifested in the nesting beach data and the population could be declining at a much more rapid rate than the analyses here represent. Considerations of extenuating circumstances such as these should be accounted for when determining acceptable interaction levels.

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Table 1. Post-interaction mortality rates for hardshell and leatherback turtles caught in longline fisheries. Numbers are the percent of hardshell (leatherback) turtles expected to die as a result of the corresponding injury and release condition (as per Ryder et al. 2006).

Nature of Interaction	Released with hook and with line greater than or equal to half the length of the carapace	Released with hook and with line less half the length of the carapace	Released with hook and entangled (line is not trailing, turtle is entangled)	Released with all gear removed
Category	Hardshell (Leatherback)	Hardshell (Leatherback)	Hardshell (Leatherback)	Hardshell (Leatherback)
I Hooked externally with or without entanglement	20 (30)	10 (15)	55 (65)	5 (10)
II Hooked in upper or lower jaw with or without entanglement. Includes ramphotheca, but not any other jaw/mouth tissue parts (see Category III)	30 (40)	20 (30)	65 (75)	10 (15)
III Hooked in cervical esophagus, glottis, jaw joint, soft palate, tongue, and/or other jaw/mouth tissue parts not categorized elsewhere, with or without entanglement. Includes all events where the insertion point of the hook is visible when viewed through the mouth	45 (55)	35 (45)	75 (85)	25 (35)
IV Hooked in esophagus at or below level of the heart (includes all hooks where the insertion point of the hook is not visible when viewed through the mouth) with or without entanglement	60 (70)	50 (60)	85 (95)	N/A
V Entangled only	50 (60)			1 (2)
I Comatose/resuscitated	N/A	70 (80)	N/A	60 (70)

Table 2. Parameters used in the Leslie matrix models to estimate the reproductive values of juvenile loggerheads interacting with the Hawaii-based shallow-set fishery in relation to adults. Size at maturity is based on lengths of nesting females reported in Hatase et al. 2004b.

Parameter	Values
First year survival rate	0.38
Juvenile survival rate	0.74-0.86
Adult survival rate	0.84-0.95
Remigration interval	2.7 yr
Eggs per nest	112
Nests per year	4
Hatch success rate	0.7
Sex ratio	0.65
Size at maturity	74 – 84 cm SCL
Age to maturity	24 – 29 yr

Table 3. Summary of literature on sex ratios in loggerhead and leatherback populations

Loggerheads	
<u>Source</u>	<u>Results</u>
Godley et al. 2001a	Found high nest incubation temperatures (above 29° C) suggestive of an 'extremely high proportion of females' in Cyprus.
Godley et al. 2001b	Estimated 89-99% females for Cyprus.
NMFS 2001	Juvenile strandings that were necropsied for sex determination between 1995 and 1999 from Texas to Virginia (N=758) were found to be 67.5% female.
Öz et al. 2004	Estimated 67% and 74% of hatchlings were female in Turkey.
Casale et al. 2006	Necropsy results for 310 loggerheads within the Mediterranean Sea showed 54.2% were female.
Kaska et al. 2006	Estimated 60-65% of hatchlings were female in Turkey
Leatherbacks	
<u>Source</u>	<u>Results</u>
Godfrey et al. 1996	Estimated nest sex ratios at 69.4% female in Suriname
Binckley et al. 1996	Estimated nest sex ratios of 74.3 – 100% female in Costa Rica (Pacific coast)
TEWG 2007	Necropsied strandings along the southeast Atlantic coast range from 57-87% female

Table 4. Parameters used in the calculation of the susceptibility to quasi-extinction (SQE) index for each population considered. $\hat{\mu}$ is the arithmetic mean of the log population growth rate and $\hat{\sigma}^2$ is the variance of the log population growth rate. . Calculations were made using a 3-yr running sum and current population size (N) was estimated as the sum of the last three years of data (approximating the total number of adult females). QET is quasi-extinction threshold and T is the time horizon for the quasi-extinction risk (the lesser value of 3 generations or 100 yr).

Population	$\hat{\mu}$	S.E. of $\hat{\mu}$	$\hat{\sigma}^2$	N	QET=50%	T
Leatherbacks, Playa Grande, Costa Rica	-0.185	0.080	0.055	335	168	63
Leatherbacks, Jamursba-Medi, Papua	-0.037	0.052	0.019	1515	758	63
Loggerheads, Japan	-0.032	0.045	0.020	2915	1548	100

Table 5. Susceptibility to quasi-extinction (SQE) values for the three populations considered here and the number of adult female mortalities that will result in an increase of SQE equivalent to 1, 5, and 10% of (1-SQE). For example, for loggerheads, SQE = 0.8311, 10% of (1-SQE) is 0.0169, resulting in a ‘new’ SQE of 0.8480 which would be achieved by an increase of 7.48 adult females per year (Fig. 3; Eq. 8 with $b = 0.017$).

	Leatherbacks Costa Rica	Leatherbacks Jamursba-Medi	Loggerheads Japan
SQE	0.9985	0.8001	0.8311
% Increase in SQE	Equivalent adult female mortalities		
0.01(1-SQE)	0.06	0.47	0.72
0.05(1-SQE)	0.30	2.38	3.66
0.10(1-SQE)	0.61	4.85	7.48

Table 6. Expected adult female mortalities and increases in the susceptibility to quasi-extinction (SQE) index based on different interaction levels for the Hawaii-based shallow-set longline fishery. Mean adult female mortalities were estimated using a 65% female sex ratio, 0.205 and 0.229 post-interaction mortality rates for loggerheads and leatherbacks, and 0.41 and 0.85 mean reproductive values for loggerheads and leatherbacks. Increases in SQE are based on the fitted logistic curves in Fig. 3. Numbers in brackets use the 95% CI on the post-interaction mortality rates to estimate adult female mortalities and the percent increase in 1-SQE.

Proposed Interactions		Expected adult female mortalities			Increase in SQE: $X(1-SQE)$	
Loggerheads						
Current	17	0.93 [0.67, 1.19]			0.013 [0.001, 0.016]	
Proposed	46	2.51 [1.81, 3.21]			0.035 [0.025, 0.044]	
Leatherbacks						
		Total	Jamursba-Medi	Costa Rica	Jamursba-Medi	Costa Rica
Current	16	2.02 [1.11, 2.93]	1.31 [0.34, 1.90]	0.12 [0.03, 0.18]	0.027 [0.007, 0.040]	0.022 [0.006, 0.031]
Proposed	19	2.40 [1.32, 3.48]	1.56 [0.42, 2.26]	0.14 [0.04, 0.21]	0.033 [0.009, 0.048]	0.043 [0.011, 0.062]

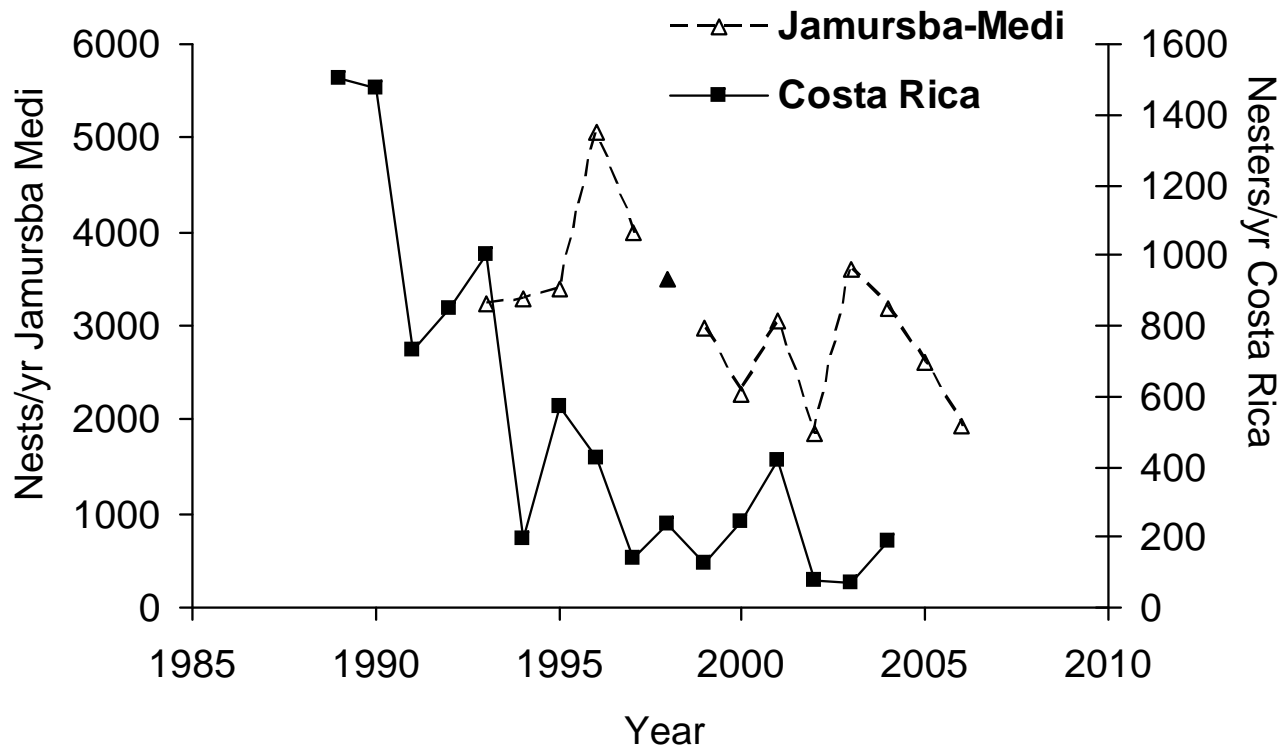


Figure 1. Nest or nester abundance trends for Jamursba-Medi, Papua, Indonesia (Hitipeuw et al. 2007) and for Parque Nacional Las Baulas, Playa Grande, Costa Rica (Tomillo et al. 2007). The 1998 datapoint for Jamursba Medi is missing and it was estimated as the mean of the nest numbers for 1997 and 1999 (filled triangle).

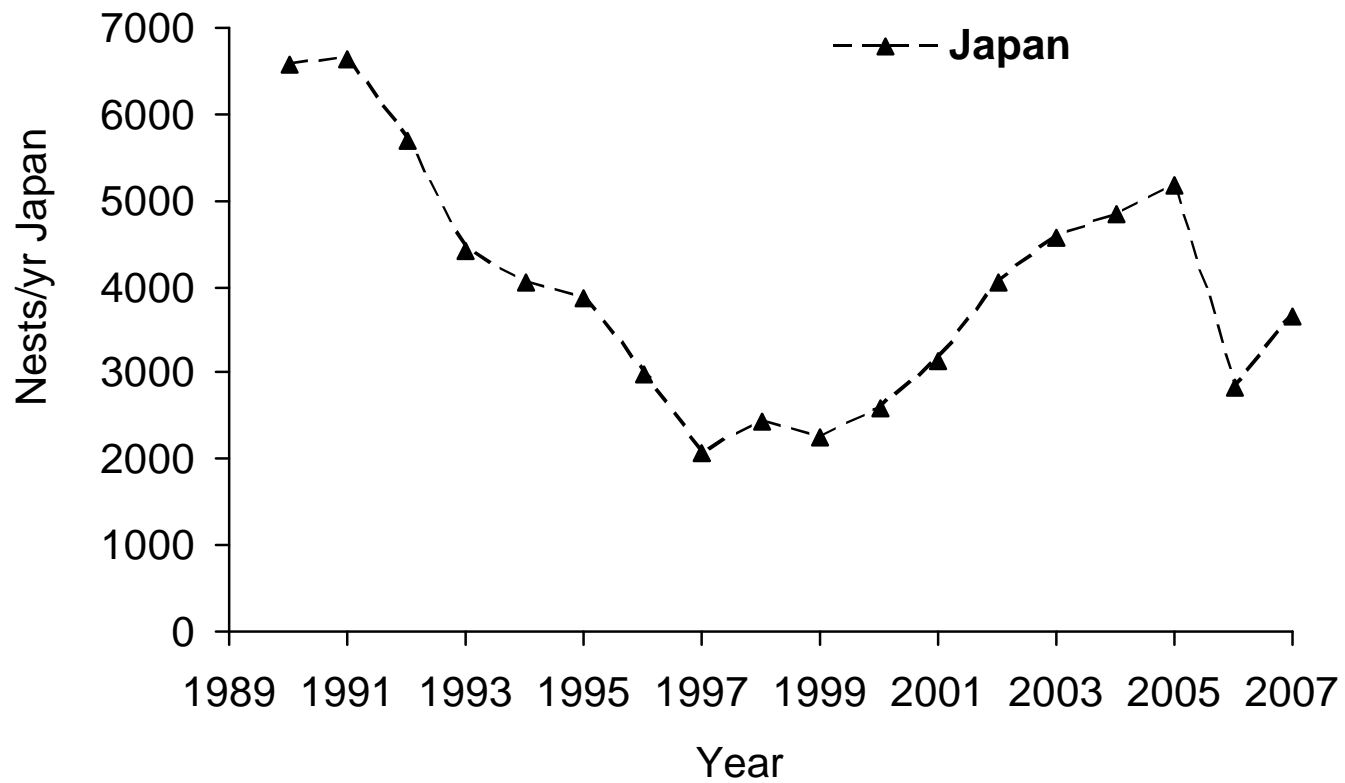


Figure 2. Nest abundance trends for loggerheads in Japan (Sea Turtle Association of Japan, unpublished data and Kamezaki and Matsuzawa 2002).

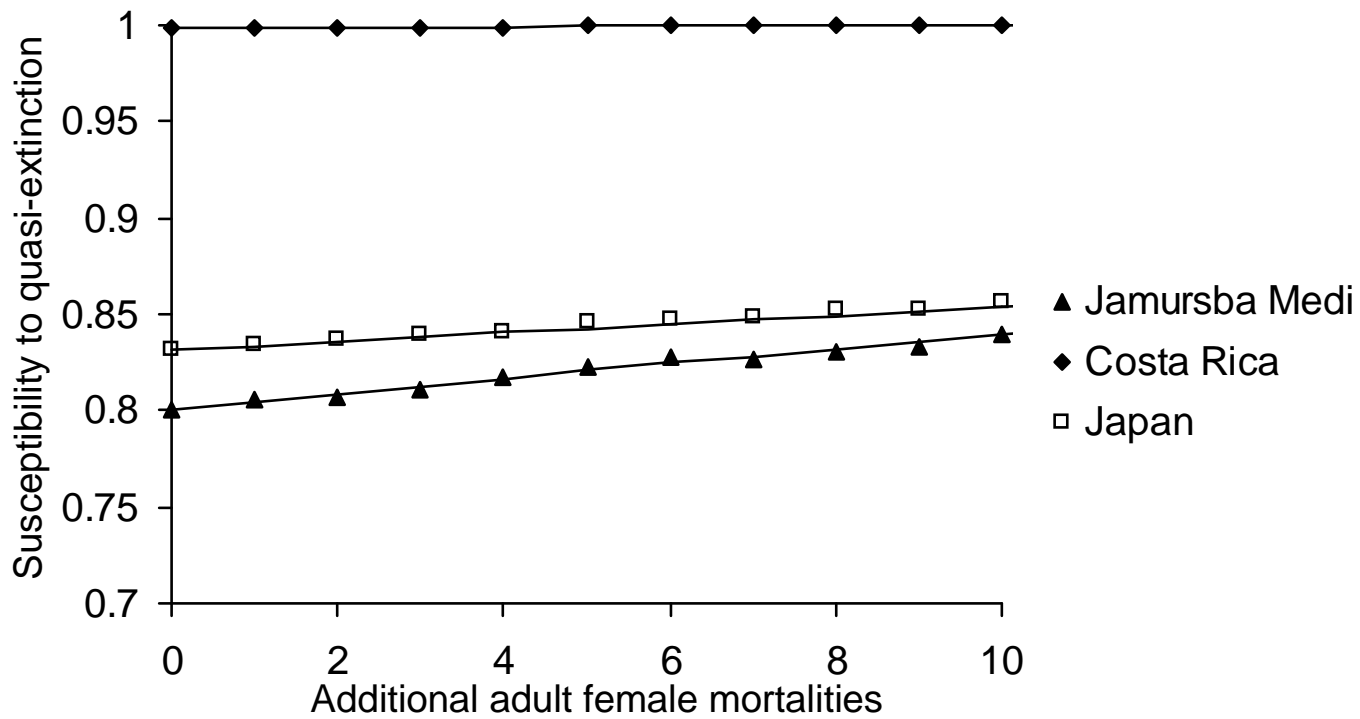


Figure 3. Changes in the susceptibility to quasi-extinction index as mortalities of adult females increases.