Center for Independent Experts (CIE) Review on sea turtle impacts from Hawaiian longline fisheries

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Executive summary

1. A 'Type I' error is when a population is not defined to be at risk when in fact it is, and a 'Type II' error is the converse: a population is considered to be at risk when in fact it is not. These two errors should not be given equal weighting in decision-making under the Endangered Species Act. The precautionary principle suggests that the cost to the species of a 'Type I' error is greater than a 'Type II' error. Hence the authors' strategy of minimizing the sum of the two error types is not consistent with the precautionary principle.

2. The performance of the SQE as judged by population simulation is potentially misleading. Overall, 95% of actual extinction risks for these 500 populations had values >0.90 or <0.10. Given this degree of separation in outcomes, it is unsurprising that classification accuracy of the SQE was 82%-92%. In the discussion, the authors state that "the classification is accurate over 80% of the time for sea turtle nest census data", but this claim is critically dependent on the extinction risks selected for the simulated populations. Discussion of whether this degree of dichotomous risk is appropriate is lacking.

3. Many sources of uncertainty in the analyses have been ignored or inadequately quantified. The motivation of the analyses appears in part to have been to deliver precise conclusions for use by managers, whether the information available supported that precision or not. This is contrary to the precautionary principle.

Terms of Reference

The terms of reference for this review are as follows.

(1) Background information on the diffusion approximation methods used to estimate quasi-extinction risks has been peer reviewed and is found in the Snover and Heppell (in press) document. Comments and criticisms on this document are welcome but should not be the focus of the reviews.

(2) An application of the methods presented in the Snover and Heppell (in press) document is contained in the Snover (2008) report which is being used by the Pacific Islands Regional Office in a section 7 consultation to determine conservative interaction levels for marine turtles in the Hawaii-based shallow-set fishery. The peer review reports should contain an assessment of this report, including input on the following questions:

- Is the overall approach appropriately conservative for species listed on the Endangered Species list?
- Do the methods used to determine 'adult equivalents' appear adequate?
- Are the methods used to determine population specific takes appropriate?
- Is the range of allowed adult female anticipated mortalities appropriately conservative for the status of the populations?

(3) Overall, the reports should contain a critical review of the methods used to determine conservative interaction levels for marine turtle populations as presented in the Snover 2008 internal report, including recommendations for improvements.

Before considering the specific questions listed under (2), we consider aspects of the simulation study conducted by Snover and Heppell (in press), as it is critical to the assessment of the adequacy of the analyses conducted by Snover (2008).

As defined on p. 12, a 'Type I' error is when a population is not defined to be at risk when in fact it is, and a 'Type II' error is the converse: a population is considered to be at risk when in fact it is not. These two errors should not be given equal weighting in decision-making under the Endangered Species Act. The precautionary principle suggests that the cost to the species of a 'Type I' error is greater than a 'Type II' error. Hence the selection of a critical value that attempts to minimize the sum of the two error types is not consistent with the precautionary principle.

The performance of the SQE as judged by population simulation (as described in Appendix B; missing Table B1) is potentially misleading. Perhaps because of the nature of the parameterization of this model, 95% of actual extinction risks for

these 500 populations had values >0.90 or <0.10. Given this degree of separation in outcomes, it is unsurprising that classification accuracy of the SQE was 82%-92%. In the discussion, the authors state that "the classification is accurate over 80% of the time for sea turtle nest census data", but this claim is critically dependent on the extinction risks selected for the simulated populations. Some discussion of whether this degree of dichotomous risk is appropriate for species of concern would have helped readers judge the validity of SQE as a general management tool.

Is the overall approach appropriately conservative for species listed on the Endangered Species list?

The SQE approach, as discussed in the above comments on Snover and Heppell (in press), does not conform to the precautionary principle. The construct of the metric, specifically the assignment of the threshold metric value, is predicated on the equivalence of Type I and Type II errors. A conservative approach for a listed species would be to weight Type I errors more heavily than Type II errors, and assign the threshold value accordingly.

Do the methods used to determine 'adult equivalents' appear adequate?

There are several gaps in the technical report that make determination of the "adult female equivalents" unclear. First, on p. 13 there is mention of growth curves to assign animals of given sizes to ages, but there is no further description of the model used, nor are the parameters associated.

The conversion from all individuals interacting with the fisheries to adult females dying is rather opaque. An equation is not presented, but appears to be

$$N_{adf} = N_{total} \cdot SR \cdot \overline{V}_x \cdot Mortality$$

where N_{adf} is number of adult females dying, N_{total} is total interactions, SR is sex ratio, V_x is an estimate of the proportion of iteractions involving adult animals, but is taken to be a mean reproductive value (p. 13), and *Mortality* is mortality rate from fishing interaction.

Perhaps details are clarified in Heppell *et al.* (2004); but the rationale for using reproductive value as a conversion factor from juveniles to adults needs to be supported and justified in a peer-reviewed arena. We would rather see justification for the values of 0.41 for loggerheads and 0.85 for leatherbacks receive some discussion and justification in this document rather than a footnote indicating they were taken from correspondence between agency personnel. What is the sensitivity of the overall outcome of this assessment to the use of these values? A more precautionary approach would be to assume that all individuals interacting with the fishery are potentially adults.

Are the methods used to determine population-specific takes appropriate?

Table 6 states the magnitude of proposed take for the populations of interest. The total numbers of 46 loggerheads and 19 leatherbacks are stated on p.6 without clarification of how those values were derived, or whether uncertainty in these values was allowed for, in deriving the confidence intervals on the estimates of expected adult female mortalities.

On p.8, a population estimate of 1335 adult female leatherbacks in the Eastern Pacific was derived, although we could not reproduce the estimate from the information given. While 14% is clearly 188/1335, we are unclear of the relevance of 346 leatherbacks nesting in Mexico to this calculation. There is presumably considerable uncertainty associated with the estimates on pp. 7-8, yet it is unclear whether any of this uncertainty was acknowledged in the analysis.

The apportionment of leatherback take ascribed to the Costa Rica component seems incorrect (or at least inconsistent with the calculation for the Jamursba-Medi component) because the 14% allocation of Eastern Pacific adult females to the Costa Rica component was not taken into account (see notes associated with Table 6, p. 27).

Clearly there is uncertainty associated with a) proportion of leatherbacks taken in the longline fisheries, b) proportion of Eastern Pacific leatherbacks coming from respective nesting assemblages as identified by personal communications of a genetics study. If nothing else, a binomial variance could be ascribed to these events.

In summary, there was not sufficient information on which to assess the methodology used to determine population-specific take from the proposed fishery.

Is the range of allowed adult female anticipated mortalities appropriately conservative for the status of the populations?

A memorandum from February of 2008 is cited as containing information on perhaps 19 leatherbacks and 46 loggerheads taken in the fishery. Presumably in conjunction with information presented in Table 1, a mortality rate was assigned to each of those individuals, and presumably those predicted mortality rates were averaged across individuals to produce the estimates presented on p. 5. Note however, the first mention of these post-interaction mortality rates also present confidence intervals for these estimates (presumably based on variability among

individuals, not on uncertainty in the values in the cells of Table 1 that were likely derived by a Delphi-method expert opinion poll).

The manner in which this uncertainty is propagated in the computations of Table 6 is unusual. We will focus on the column labeled "expected adult female mortalities." Bracketed values in Table 6 simply replace point estimate of post-interaction mortality with the upper and lower confidence bounds. This does not make these bracketed values confidence intervals, so they are unnamed and undefined intervals. Furthermore, the expected adult female mortalities are estimated as the product of 4 estimated quantities (see equation above); however the only acknowledged source of uncertainty is in the post-interaction mortality; although the reproductive values used are said to be means; so some form of uncertainty may be available.

The entire concept of using susceptibility to quasi-extinction as a metric to integrate PVA information about management actions is layered atop all of the calculations that have thus far been discussed. It seems the philosophical underpinning of SQE is to boil away uncertainty that is ubiquitous in PVA work, and leave a value with no measure of uncertainty.

The *Ecological Applications* manuscript abstract states that "A drawback to PVA is the high degree of uncertainty in these forecasts due to both population stochasticity and parameter estimation uncertainty. With sparse or noisy data, extinction probabilities frequently have 95% confidence intervals ranging from 0 to 1." From this philosophical perspective, it is easy to see that acknowledging uncertainty in predicting quantities such as adult female anticipated mortalities is considered optional rather than obligatory in this assessment.

Conclusion

The manuscript is inconsistent in handling complexity. There is no indication of how the level of proposed interactions was estimated, and no acknowledgement of uncertainty in the estimates, yet that number drives the rest of the decision process. However, a matter of small consequence (how to interpolate SQE between integer values of additional adult female mortalities) receives logistic curve fitting attention (even though over the range of the independent variables, the relationships are linear). Not only is the relationship between adult female equivalent mortalities and SQE linear, it is nearly horizontal, indicating insensitivity of this metric to increased adult female mortality.

Throughout, there is a cavalier attitude towards quantifying uncertainty. The aim appears to be to develop a procedure that has the appearance of high precision, when in reality, many sources of uncertainty have been ignored. This seems contrary to the precautionary principle.

References

Heppell, S., Crowder, L.B. and Snover, M. 2004. Comparing mortality risks with "adult equivalent" based on reproductive value. 24th Symposium on Sea Turtle Biology and Conservation, February 2004. Vol: I. International Sea Turtle Society. San Jose, Costa Rica.

Snover, M.L. 2008. 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 fishery.

Snover, M.L. and Heppell, S.S. in press. Application of diffusion approximation for risk assessments of sea turtle populations. *Ecological Applications*.

Appendix 1. Statement of work

External Independent Peer Review by the Center for Independent Experts

Impacts of Potential Increases in Hawaii Shallow-set Swordfish Longline Effort on Sea Turtle Populations

Project Background:

The Hawaii Longline Association has proposed to expand the Hawaii-based shallow-set longline fishery, which will likely increase the level of sea turtle interactions. There is very little demographic information for these populations except for time series of nesting beach census data so a relatively simple population viability assessment approach was taken using diffusion approximation on these time series.

Snover and Heppell (in press) present a quasi-extinction risk index based on diffusion approximation 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) for a population based on nesting beach census data, 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.

This technique was applied to nest census data for Pacific loggerheads and leatherbacks to assess the population-level impacts of increased mortality resulting from the Hawaii Longline Association's (HLA) proposed expansion of the Hawaii-based shallow-set longline fishery. Anticipated increases in SQE for turtle populations from mortalities associated with this fishery were estimated. As the SQE index is based on 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 the mortalities should be less than 3 adult females, and for the Costa Rica 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.

The manuscript presenting the SQE index has received full peer review and is now in press with *Ecological Applications*. The specific application of SQE to determining appropriate take levels for protected marine turtle population has not received a full peer review; and as these analyses are designed to be a general tool for managers to assess how different levels of fishery interactions may impact the extinction risk of marine turtle populations, a CIE review is warranted.

Overview of CIE Peer Review Process:

The National Marine Fisheries Service's (NMFS) Office of Science and Technology coordinates and manages a contract for obtaining external expertise through the Center for Independent Experts (CIE) to conduct independent peer reviews of stock assessments and various scientific research projects. The primary objective of the CIE peer review is to provide an impartial review, evaluation, and recommendations in accordance to the Statement of Work (SoW), including the Terms of Reference (ToR), to ensure the best available science is utilized for the National Marine Fisheries Service management decisions.

The NMFS Office of Science and Technology serves as the liaison with the NMFS Project Contact to establish the SoW which includes the expertise requirements, ToR, statement of tasks for the CIE reviewers, and description of deliverable milestones with dates. The CIE, comprised of a Coordination Team and Steering Committee, reviews the SoW to ensure it meets the CIE standards and selects the most qualified CIE reviewers according to the expertise requirements in the SoW. The CIE selection process also requires that CIE reviewers can conduct an impartial and unbiased peer review without the influence from government managers, the fishing industry, or any other interest group resulting in conflict of interest concerns. Each CIE reviewer is required by the CIE selection process to complete a Lack of Conflict of Interest Statement ensuring no advocacy or funding concerns exist that may adversely affect the perception of impartiality of the CIE peer review. The CIE reviewers conduct the peer review, often participating as a member in a panel review or as a desk review, in accordance with the ToR producing a CIE independent peer review report as a deliverable. At times, the ToR may require a CIE reviewer to produce a CIE summary report. The Office of Science and Technology serves as the Contract Officer's Technical Representative (COTR) for the CIE contract with the responsibilities to review and approve the deliverables for compliance with the SoW and ToR. When the deliverables are approved by the COTR, the Office of

Science and Technology has the responsibility for the distribution of the CIE reports to the Project Contact.

Requirements for CIE Reviewers:

Three CIE reviewers shall conduct an independent peer review in accordance with the Statement of Work (SoW). The CIE reviewers shall have strong quantitative expertise in the population dynamics including critical analysis in stock assessment and alternative methods for improving stock assessment methods using limited datasets such as are typically available for protected species. It is desirable to have one CIE reviewer with experience in population dynamics or ecology of sea turtles. Each CIE reviewer shall have the ability to conduct the necessary pre-review preparations, desk review (no travel is necessary), and completion of the peer review report in accordance to the ToR and schedule of milestone and deliverables specified herein, and the number of days for each CIE reviewer shall not exceed 14 days.

The CIE reviewers shall have the requested expertise necessary to complete an impartial peer review and produce the deliverables in accordance with the SoW and ToR as stated herein (refer to the ToR in Annex 1).

(1). Strong quantitative expertise in the population dynamics including population viability assessment methods using limited datasets such as are typically available for protected species. Knowledge of diffusion approximation methods as they apply to quantifying quasi-extinction risks would be helpful.

(2). An understanding of the difficulties inherent in marine turtle research, nesting beach monitoring and the interpretation of these data

(3). As this analysis was driven by immediate needs of management in preparing an EA/EIS and in an ESA section 7 consultation, an understanding of endangered species regulations that drive these management needs would also be helpful

Statement of Tasks for CIE Reviewers:

The CIE reviewers shall conduct necessary preparations prior to the peer review, conduct the peer review, and complete the deliverables in accordance with the ToR and milestone dates as specified in the Schedule section.

Prior to the Peer Review: The CIE shall provide the CIE reviewers' contact information (name, affiliation, address, email, and phone) to the Office of Science and Technology COTR no later than the date as specified in the SoW, and this information will be forwarded to the Project Contact.

<u>Pre-review Documents</u>: Approximately two weeks before the peer review, the Project Contact will send the CIE reviewers the necessary documents for the peer review, including supplementary documents for background information. The CIE reviewers shall read the pre-review documents in preparation for the peer review (see tentative list below).

1) Snover, M.L. and S.S. Heppell. In press. Application of diffusion approximation for risk assessment of sea turtle populations. Ecological Applications.
2) Snover, M.L. 2008. Assessment of the population-level impacts of potential increases in marine turtle interactions from a Hawaii Longline Association proposal to expand the Hawaii-based shallow-set fishery. NOAA/NMFS/Internal Report IR-08-010.

This tentative list of pre-review documents may be updated up to two weeks before the peer review. Any delays in submission of pre-review documents for the CIE peer review will result in delays with the CIE peer review process. Furthermore, the CIE reviewers are responsible for only the pre-review documents that are delivered to them in accordance to the SoW scheduled deadlines specified herein.

<u>Desk Peer Review</u>:

The primary role of the CIE reviewer is to conduct an impartial peer review in accordance to the Terms of Reference (ToR) herein, to ensure the best available science is utilized for the National Marine Fisheries Service (NMFS) management decisions.

Terms of Reference:

CIE reviewers shall conduct an independent peer review addressing each of the following Term of Reference:

(1). Background information on the diffusion approximation methods used to estimate quasi-extinction risks has been peer reviewed and is found in the Snover and Heppell (in press) document. Comments and criticisms on this document are welcome but should not be the focus of the reviews.

(2). An application of the methods presented in the Snover and Heppell (in press) document is contained in the Snover (2008) report which is being used by the Pacific Islands Regional Office in a section 7 consultation to determine conservative interaction levels for marine turtles in the Hawaii-based shallow-set fishery. The peer review reports should contain an assessment of this report, including input on the following questions:

- Is the overall approach appropriately conservative for species listed on the Endangered Species list?
- Do the methods used to determine 'adult equivalents' appear adequate?
- Are the methods used to determine population specific takes appropriate?
- Is the range of allowed adult female anticipated mortalities appropriately conservative for the status of the populations?

(3) Overall, the reports should contain a critical review of the methods used to determine conservative interaction levels for marine turtle populations as presented in the Snover 2008 internal report, including recommendations for improvements.

Independent CIE Peer Review Reports:

The primary deliverable of the SoW is for each CIE reviewer shall be to complete and submit an independent CIE peer review report in accordance with the ToR, and this report shall be formatted as specified in the attached Annex 1. The report will be sent to Mr. Manoj Shivlani, CIE lead coordinator, via email to shivlanim@bellsouth.net and Dr. David Die, CIE regional coordinator, via email to ddie@rsmas.miami.edu.

Schedule of Milestones and Deliverables:

Each CIE Reviewer shall complete the independent peer review in accordance with the following schedule of milestones and deliverables.

October 16, 2008	CIE shall provide the COTR with the CIE reviewer contact information, which will then be sent to the Project Contact
October 20, 2008	The Project Contact will send the background documents and report to the CIE Lead Coordinator and CIE reviewers
October 21- November 7, 2008	Each reviewer shall conduct an independent peer review
November 7, 2008	Each reviewer shall submit draft CIE independent peer review reports to the CIE
November 21, 2008	CIE will submit CIE independent peer review reports to the COTR
December 21, 2008	The COTR will distribute the final CIE reports to the Project Contact

Acceptance of Deliverables:

Each CIE reviewer shall complete and submit an independent CIE peer review report in accordance with the ToR, which shall be formatted as specified in Annex 1. Upon review and acceptance of the CIE reports by the CIE Coordination and Steering Committees, CIE shall send via e-mail the CIE reports to the COTR (William Michaels via William.Michaels@noaa.gov at the NMFS Office of Science and Technology by the date in the Schedule of Milestones and Deliverables. The COTRs will review the CIE reports to ensure compliance with the SoW and ToR herein, and have the responsibility of approval and acceptance of the deliverables. Upon notification of acceptance, CIE shall send via e-mail the final CIE report in *.PDF format to the COTRs. The COTRs at the Office of Science and Technology have the responsibility for the distribution of the final CIE reports to the Project Contacts.

Key Personnel:

Contracting Officer's Technical Representative (COTR):

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Request for Changes:

Requests for changes shall be submitted to the Contracting Officer at least 15 working days prior to making any permanent substitutions. The Contracting Officer will notify the Contractor within 10 working days after receipt of all required information of the decision on substitutions. The contract will be modified to reflect any approved changes. The Terms of Reference (ToR) and list of pre-review documents herein may be updated without contract modification as long as the role and ability of the CIE reviewers to complete the SoW deliverable in accordance with the ToR are not adversely impacted.

ANNEX 1

Format and Contents of CIE Independent Reports

1. The report should be prefaced with an Executive Summary with concise summary of goals for the peer review, findings, conclusions, and recommendations.

2. The main body of the report should consist of an Introduction with
     a. Background
     b. Terms of Reference
     c. Panel Membership
     d. Description of Review Activities

3. Summary of Findings in accordance to the Term of Reference

4. Conclusions and Recommendations in accordance to the Term of Reference

5. Appendix for the Bibliography of Materials used prior and during the peer review.

6. Appendix for the Statement of Work

7. Appendix for the final panel review meeting agenda.

8. Appendix for other pertinent information for the CIE peer review.

Appendix 2

PDF document of Snover (2008), with comments appended for information.

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 fishery[1]

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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

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[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. 2008[4]). 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

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

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 generations[6], 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 smaller[6]. To estimate generation time for leatherbacks, I used the mean value of age to sexual maturity of 14 yr (Zug and Parham

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[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 $\hat{\mu}_b$, $\hat{\sigma}^2{}_b$ 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_0, was estimated as the sum of the last 3 yr of data. I considered the effect of $m_i = 1, 2, 3 \ldots 10$ additional annual adult female mortalities on SQE values. The intrinsic rate of population increase (r) is calculated as

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

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

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

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

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

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 assessment[7] and so I use that value in this assessment.

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[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 PIRO[7] 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 of 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	
Source	**Results**
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	
Source	**Results**
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.

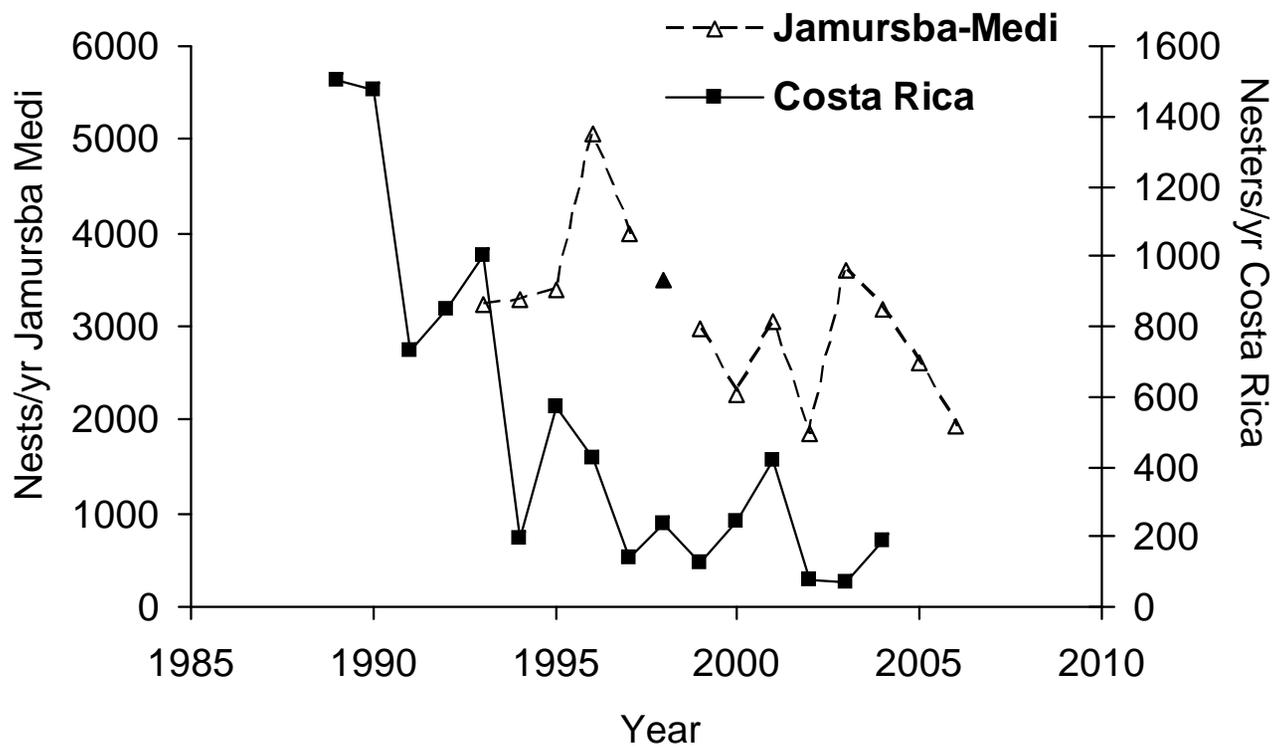|	Proposed Interactions	Expected adult female mortalities			Increase in SQE: $\mathbf{X}$(1-SQE)	
Loggerheads						
Current	17	0.93			0.013	
		[0.67, 1.19]			[0.001, 0.016]	
Proposed	46	2.51			0.035	
		[1.81, 3.21]			[0.025, 0.044]	
Leatherbacks		Total	Jamursba-Medi	Costa Rica	Jamursba-Medi	Costa Rica
Current	16	2.02	1.31	0.12	0.027	0.022
		[1.11, 2.93]	[0.34, 1.90]	[0.03, 0.18]	[0.007, 0.040]	[0.006, 0.031]
Proposed	19	2.40	1.56	0.14	0.033	0.043
		[1.32, 3.48]	[0.42, 2.26]	[0.04, 0.21]	[0.009, 0.048]	[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).
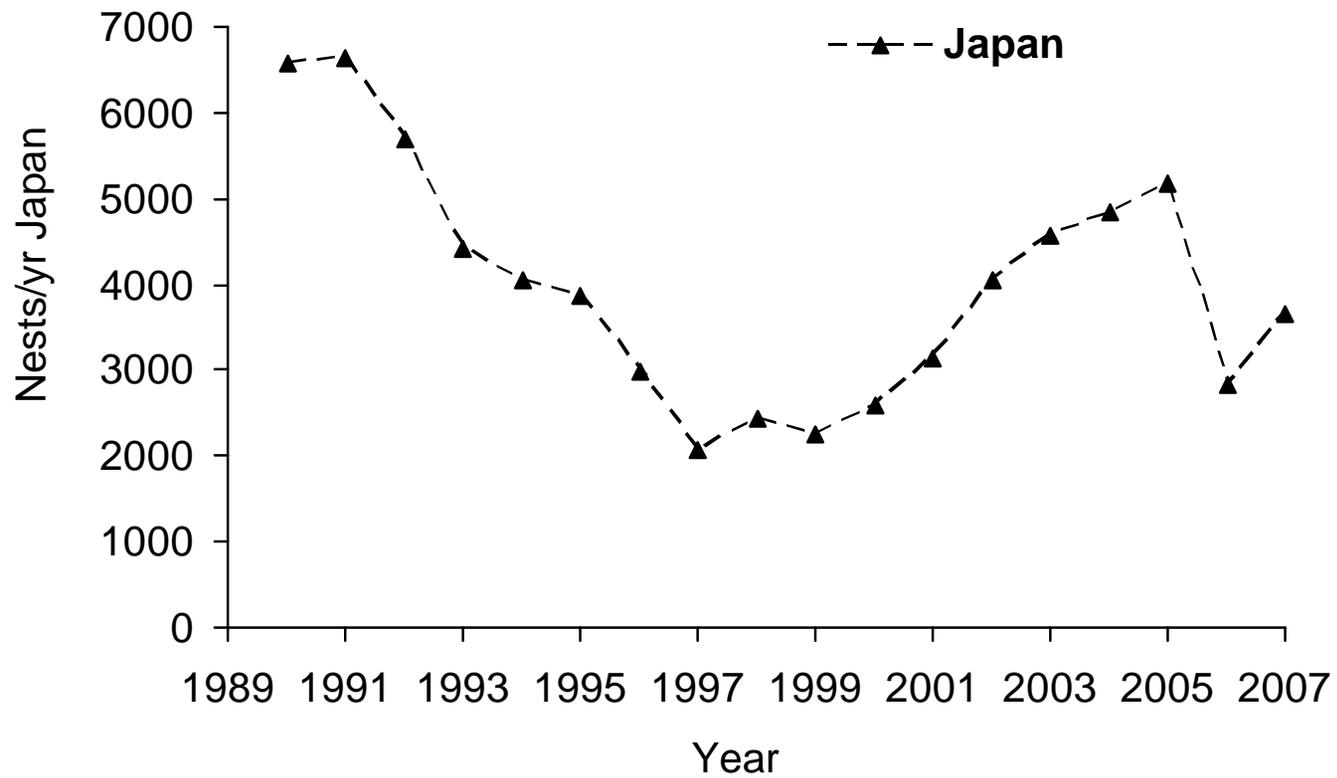
Figure 2. Nest abundance trends for loggerheads in Japan (Sea Turtle Association of Japan, unpublished data and Kamezaki and Matsuzawa 2002).
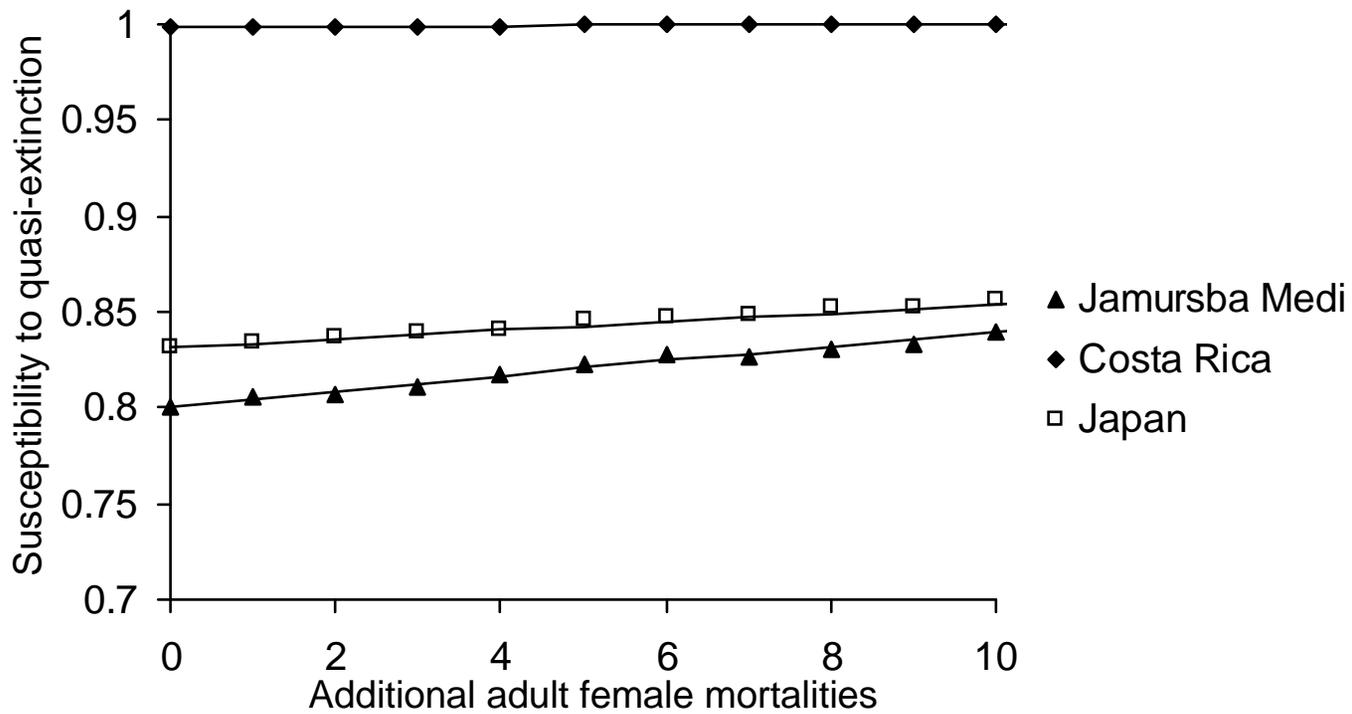
Figure 3. Changes in the susceptibility to quasi-extinction index as mortalities of adult females increases.