

Final Report for the  
Southeast Fisheries Science Center

***Development of Alternative  
Quantitative Tools to Assist in  
Jeopardy Evaluation for Sea Turtles***

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## Author's Note

This report represents my musings over potential methods for evaluating the effects of human activities on protected sea turtle species. It is meant to serve as a starting point for discussion with other modelers and managers who desire quantitative assessment tools for jeopardy evaluations. There are many alternatives not included here, and undoubtedly improvements on options that are discussed. In particular, development of a final framework for assessment should include Bayesian techniques to better evaluate uncertainty and to incorporate data such as size distributions and in-water abundance trends. There are also options that I rejected because of mathematical intractability or insurmountable data gaps. My discussion of the issues associated with the creation of management models for sea turtle assessment should provide modelers with some background on sea turtle biology and data, and for non-modelers who may need a better understanding of the differences between management and heuristic models. I do not consider these ideas proprietary. As a NMFS contractor, I have provided this report for use by the SEFSC to begin discussion of methods and evaluation. I thank the agency, Chris Sasso and Sheryan Epperly for their support, and Barbara Taylor, Paul Wade, Melissa Snover, Tomo Eguchi, Paul Richards and Heather Haas for stimulating discussions of alternative methodologies. I look forward to working with these individuals and other modelers in the future.

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## NOTICE

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The findings and conclusions in this report are those of the author and do not necessarily represent the views of the funding agency.

## Statement of Purpose

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This report was contracted by the Southeast Fisheries Science Center to outline issues associated with quantitative evaluation of the impacts of takes on sea turtle populations. It serves as a framework for development of quantitative risk analysis tools that can be used to evaluate how human activities that result in sea turtle mortality may affect the recovery of stocks, populations or species. Five possible evaluation tools are discussed, with examples of how the models could be used as part of a Section 7 consultation. This report is intended to present issues, options, and recommendations for consideration by NMFS and quantitative ecologists on the pros and cons of various quantitative evaluation alternatives and the associated data/research needs to support them.

## Introduction

Biological Opinions (BiOps) are required by the Endangered Species Act to assess the population-level impacts of “takes” of listed or threatened species, where a “take” may be direct or indirect killing, injuring, or harassment of individuals. BiOps are also required when human activities affect the habitat of a listed species. Activities may need to be reduced or restricted if they are likely to impede recovery of a listed species or stock. For sea turtles, which are under the joint jurisdiction of NOAA Fisheries and the U.S. Fish and Wildlife Service, BiOps are most often written in response to sea turtle interactions with commercial fisheries or for coastal development activities. Under ESA guidelines, BiOps must include an assessment of “jeopardy” to the affected population or species as a whole. BiOps and jeopardy rulings are critical documents in litigation and are regularly challenged by environmental and industry groups. Standardized, quantitative tools are desirable to determine when a “take” is sufficient to cause “jeopardy”, warranting a curtailment of the fishing or development activity.

Because sea turtles are long-lived, wide ranging and difficult to observe throughout most of their life history, data are limited and assessments are difficult. All sea turtles inhabit international waters at some point in their life history and individuals may frequent the waters of multiple nations during their lifetimes. In most cases, only a small fraction of a population is monitored (nests or adult nesting females). Data quantity and quality is highly variable among populations that occur in U.S. waters.

Mathematical models are powerful tools for species assessment and evaluation. The reliability and utility of models depends on data availability and assumptions conferred by model structure. Population models for sea turtles have been reviewed by Chaloupka and Musick (1997), Heppell et al. (2003) and others. Published models have ranged from simple, deterministic life cycle analyses to complex simulation models, all with varying data requirements and assumptions. Most BiOps include a quantitative model based on statistical evaluation to determine the total take invoked by an activity, such as an extrapolation to a total number of turtles affected by a fishery when only a small proportion of takes are actually observed. A second model is needed to quantitatively evaluate how the total take from an activity, and the cumulative take of many human activities, may affect a species or population.

Appropriate model complexity is strongly dependent upon the question asked; a simple model may be robust to uncertainty in life cycle parameters but may be incapable of making precise estimates of population size or the effects of removals. On the other hand, detailed simulation models may

require a large amount of biological information to produce reliable estimates of allowable take. There are tradeoffs in model construction among precision, realism and generality; Levins (1966) argued that a particular model can achieve at most 2 of these 3 qualities. Taylor et al. (2000) proposed a distinction between models for population management and models for heuristic evaluation of population dynamics. Assessments of human impacts that are required for management require precision, but must operate in the absence of demographic data. Tools that rely on simple relationships among measurable parameters can be generally applied to a wide range of populations and can be understood by a broad audience of stakeholders. The sacrifice, then, is in biological realism, and management models cannot be used to predict population dynamics.

An example of the “management model” concept is in the Potential Biological Removal (PBR) developed for use by the International Whaling Commission to set limits on intentional and unintentional takes of marine mammals. This model is routinely applied for assessment of US stocks of marine mammals. The model provides conservative take limits that are appropriate for the “insignificant mortality” goals specified by the Marine Mammal Protection Act and the recovering status of most marine mammal populations. The equation is simple algebra and relies on measureable parameters:

$$PBR = N_{\min} \times R_{\max} \times 0.5 \times F \quad , \quad (1)$$

where  $N_{\min}$  is a minimum estimate of total population size (e.g., the lower 60% confidence interval from a lognormally distributed population estimate),  $R_{\max}$  is the maximum growth rate of the population expected at low density, and  $F$  is a “recovery factor” based on simulation analyses (Wade 1998) that is generally set at 0.1 (most conservative) for endangered populations, 0.5 for threatened populations, and 1.0 for recovered or non-threatened populations. This model is useful for management because it provides a clearly calculated target value of allowable take that incorporates uncertainty and can be applied to any marine mammal population, but is also easy to understand. Poor data on population size results in a lower PBR (wider confidence intervals) and may serve as encouragement for better assessment. Most importantly, PBR is conservative in the face of uncertainty, and thus is precautionary in nature. It is a measure of total allowable removal and must be allocated to various fisheries. Fisheries that exceed their PBR quota are required to participate in a take reduction team to devise strategies for reducing their impact on populations.

PBR may not be appropriate “as is” for sea turtles because there is no estimate of total population size available and many populations are declining. However, it is highly desirable to develop a management model that achieves the objectives of PBR, namely, a robust estimate of total allowable removals that considers uncertainty and can be applied objectively to a wide range of species.

## ***What is jeopardy?***

Section 7 of the Endangered Species Act requires federal agencies to ensure that any action they authorize is not likely to jeopardize the continued existence of listed species or result in the destruction or adverse modification of their critical habitat. In cases where listed marine species might be affected, federal agencies must consult with NMFS regarding the effect of their actions. NMFS defines “Jeopardize the continued existence of” as “to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species” (50 CFR §402.02). Biological Opinions that review the impact of negative actions on listed species provide a ruling on whether the activity constitutes “jeopardy” for the listed species or subpopulation. An analytical framework for evaluating jeopardy is generally described in the ESA, but is not standardized – nor are quantitative tools for unbiased assessment required or specified.

From a population dynamics standpoint, “jeopardy” can be defined as a significant increase in the probability of extinction or a significant decrease in the probability of recovery. Of course, “significance” requires a subjective threshold and a statistical assessment; Section 7 specifies an “appreciable”, rather than “significant”, change in status. This implies that precision should be based on detectability, rather than a p-value, so uncertainty should play an integral role in any management model. By putting jeopardy in these terms we are able to create a framework for evaluation. For example, a decrease in probability of recovery can be indexed as a “detectable” change in expected population growth rate (short-term or asymptotic), given our uncertainty in the data. This permits the establishment of well-defined, quantitative thresholds for evaluation.

Management models that provide an evaluation tool for jeopardy determination should include thresholds or precautionary rules based on population status. Clearly, removal of individuals from a declining population has potential for greater negative impact than removals from an increasing population. By one argument, if the asymptotic population growth rate is  $<1$  then there is no surplus production and the probability of recovery = 0, so all take could be classified as jeopardy! However, this may place unreasonable restrictions on fishery operations or other activities, and the population growth rate observed on nesting beaches may not be that experienced by the entire population. In a similar vein, it can be argued that a recovering population is “sustaining” current removal levels and no jeopardy ruling is warranted for any existing fisheries (but see comments on time lags, below). The goal of standardizing tools for jeopardy determination should be to provide a reasonable and objective model for evaluation under a wide range of species status conditions and uncertainty.

One important consideration for management models and recovery planning is that there are factors beyond population size that contribute to extinction risk and the overall “health” of a population. A decrease in the estimated population growth rate, decrease in population size to a point below its “depensation” point (leading to decreased per capita reproductive success) and increased susceptibility to environmental variance all contribute to the probability of extinction. Resilience of a population or species to catastrophe and other forms of environmental disturbance is dependent on several factors, including genetic and life history diversity, spatial distribution of populations, and quantity and quality of critical habitat for all life stages (McElhany et al. 2000).

## ***Management models for sea turtles***

With the previous comments in mind, and the nominal success of PBR as a management model for marine mammals, we should develop management models for sea turtles that:

1. are based on available data, or at least obtainable data,
2. are precautionary (less information = more conservative ruling) but “reasonable”,
3. consider takes from all anthropogenic mortality sources,
4. are based on estimated mortality (dead turtles), both direct and delayed,
5. are more conservative for declining populations, and
6. are simple enough to clearly explain to stakeholders, plaintiffs, etc.

The Options outlined in this report meet some or all of these criteria.

## ***What we know and don't know about sea turtle stocks***

Sea turtle population parameters vary in quantity and quality by species. A major hindrance for model development and status evaluation is a lack of age-specific information; this deficiency, along with limited sampling opportunities, makes the application of many fishery-based assessment tools unfeasible. Table 1 *subjectively* summarizes the current information on stocks that are under NMFS jurisdiction, and serves as a simple diagnostic of where major uncertainties lie.

Table 1. Qualitative summary of available data on sea turtle populations that occur in U.S. waters.

KEY: **Green** = published estimates for most of population    **Yellow** = published estimates for portions of population  
**Orange** = limited published information or estimates restricted to small subset of population    **Red** = no published estimates  
**Blue** = simulation or life cycle model estimates available

	Loggerhead		Leatherback		Green		Hawksbill		Kemp's Ridley	Olive Ridley	
	Atlantic	Pacific*	Atlantic	Pacific	Atlantic	Pacific*	Atlantic	Pacific*	Atlantic	Atlantic	Pacific
Size of nesting population	Green	Yellow	Green	Yellow	Green	Yellow	Yellow	Yellow	Green	Orange	Yellow
Size of adult population	Yellow	Orange	Yellow	Orange	Green	Yellow	Yellow	Red	Green	Red	Yellow
Size of juvenile population	Red	Red	Red	Red	Orange	Red	Red	Red	Blue	Red	Red
Nesting population trends	Green	Yellow	Green	Yellow	Green	Yellow	Yellow	Yellow	Green	Orange	Yellow
Stock structure	Green	Orange	Green	Yellow	Orange	Yellow	Yellow	Orange	Green	Orange	Orange
Sex ratio (adult or subadult)	Yellow	Orange	Red	Red	Yellow	Yellow	Orange	Red	Yellow	Orange	Orange
Sex ratio (primary)	Yellow	Orange	Yellow	Yellow	Yellow	Yellow	Yellow	Orange	Green	Orange	Yellow
Nest survival rates	Green	Orange	Green	Yellow	Yellow	Yellow	Yellow	Red	Green	Orange	Yellow
Remigration interval	Yellow	Orange	Green	Yellow	Yellow	Yellow	Yellow	Red	Green	Orange	Orange
Nests per female, eggs per nest	Green	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Yellow	Green	Yellow	Yellow
Growth rates, age at maturity	Yellow	Orange	Orange	Red	Orange	Yellow	Yellow	Red	Yellow	Orange	Orange
Annual survival rates	Blue	Red	Yellow	Orange	Orange	Yellow	Yellow	Yellow	Blue	Red	Red
Distribution, migration	Yellow	Orange	Yellow	Orange	Orange	Orange	Orange	Red	Yellow	Red	Orange
Critical habitat	Yellow	Orange	Yellow	Orange	Yellow	Orange	Yellow	Orange	Yellow	Red	Red
Take estimates – direct	Green	Orange	Yellow	Orange	Yellow	Orange	Orange	Yellow	Green	Orange	Yellow
Take estimates - bycatch	Orange	Orange	Orange	Orange	Orange	Orange	Orange	Orange	Yellow	Orange	Orange
Size distributions of bycatch	Green	Orange	Yellow	Orange	Orange	Orange	Orange	Orange	Yellow	Orange	Orange
Reproductive values	Blue	Orange	Orange	Orange	Blue	Orange	Blue	Red	Blue	Orange	Orange
Elasticity analysis	Blue	Red	Red	Blue	Blue	Blue	Blue	Red	Blue	Red	Red

\* most of the variables listed here have been estimated or calculated for loggerheads, hawksbills and green turtles originating in Australia. It may be possible to use these estimates for N. Pacific turtles, although differences in somatic growth rates, habitat and mortality sources may make such extrapolations tenuous.

## Overview of Approaches – General Issues

Sea turtles have unique biological characteristics and dispersal behaviors that restrict the quality and type of data that can be used in assessments. Although some of the data deficiencies shown in Table 1 can be addressed through an expanded research program, there are general issues in sea turtle status assessment that need to be considered before a management model or assessment framework is selected.

### **Stock definition**

Stock assessment requires definition of the stock. This is required in all recovery plans, and is currently in revision for many sea turtle stocks as new genetic data are incorporated. There is a great need for standardized assessment methods that allow for pooling of data from multiple nesting beaches. In most cases, there are a few well-studied nesting beaches and a large number of poorly or only recently monitored ones. Because total population size or population trend (preferably both) are required for any assessment tool, proper pooling and weighting of data should be carefully considered. Ideally, all sea turtle populations affected by U.S. law that will be subjected to a standardized jeopardy decision-making framework will also be assessed and monitored similarly.

There are drawbacks to defining stocks too finely or too broadly. If subpopulations are identified, evaluation of takes will require knowledge of the genetic composition of the removals by each fishery or other anthropogenic stressor. This is actually difficult, because the assignment of an individual to a genetic subgroup relies on a probabilistic evaluation that requires reasonably large sample sizes, unless distinct haplotypes can be identified for each subgroup (T. Eguchi, *pers. comm.*). Assuming that issues of sample size and identification can be overcome, a likely scenario will then be that take evaluation will be based on “weak stock management”, where jeopardy is determined by the potential effects of the fishery on the smallest or fastest declining stock. While such an approach may be desirable from a conservation standpoint, there is a risk of delay in jeopardy evaluation and take reduction caused by the need for genetic evaluation. If stocks are defined too broadly, there is a danger of losing important genetic variation in the population or species if impacts are regional and there is unequal mixing of the population in space and time.

### **Time lags and evaluation of population status**

A critical issue in population assessment for sea turtles is the long time lag between hatching and age at first reproduction. Recent estimates from growth rate models predict ages at maturity of 25-30+ years for loggerhead and green turtles and 10-15 years for ridley turtles (Snover 2002). Leatherbacks probably reproduce for the first time at a relatively young age (12-15 years) but aging techniques are poor for this species and there seems to be little relationship between size and age, as estimated by growth rings in the eye sclera (L. Avens, *pers. comm.*, Zug and Parham 1996). Difficulties with stock assessment arise because only nesting females are regularly censused, and techniques for evaluating the proportion of nesters that are reproducing for the first time are either unavailable or subjective (possible exceptions are Kemp’s ridley females, which can be assessed by size and physical characteristics of the carapace (R. Marquez, *pers. comm.*), and Australian loggerhead and green turtles, which have been widely assessed by laparoscopic examination of the gonads (C. Limpus, *pers. comm.*)). Trends in the number of nests or nesting females only reflects the total population trend after a population has approached

a stable age distribution, which requires constant mean vital rates over 2-3 generations (Caswell 2001). In addition to making assessment methods based on deterministic life cycle models inaccurate, population trend or status based on nesting females alone may or may not reflect current mortality, particularly removals of juveniles. Some models, such as PBR, get around the time lag problem by basing allowable removals on maximum potential growth rate rather than current observed growth rate, but this may result in allowable takes that are too high for populations in decline unless conservative recovery factors are applied. If population size is decreasing, a constant number of removals becomes a larger proportion of N each year.

There are two potential solutions to this problem. The first is to do more in-water evaluation of population size and size structure, to detect possible shifts in mortality rates that affect juveniles. Such research would also enable evaluation of predictive population models. The second is to base precautionary thresholds for jeopardy determination on the age class of turtles affected, with greater precaution applied when the removals consist of juveniles because their removal is not likely to be detected in the nesting population for some time.

### ***Estimation of total allowable take (cumulative risk)***

The cumulative effects of removals must be considered in any evaluation of individual threats. Current allowable takes range from dozens to thousands for various fisheries and other human activities that affect sea turtles. While allocation of take to various mortality sources will be contentious, the risk of exceeding a sustainable level of exempted takes should not be underestimated. Management models should be based on thresholds for total allowable removals that account for the status (trend) and size of a population. A precautionary approach will assume that mortalities from various threats are additive, and total annual removals could be based on the upper 95% CI of the summed estimate.

An assessment that considers total removals may ultimately require a two step process: evaluation and allocation. Once an allowable removal level has been determined, those removals can be allocated to various fisheries. While undoubtedly onerous, all fisheries management and, more recently, marine mammal assessment through PBR, includes this allocation step. Assessment of the population-level risk from individual anthropogenic threats can be conducted by comparing each removal to the total allowable take, or takes can be qualitatively compared to other assessed removals. The jeopardy question could then be recast as “is threat x likely to tip total removals over the maximum allowed threshold?”

### ***Methods based on population size***

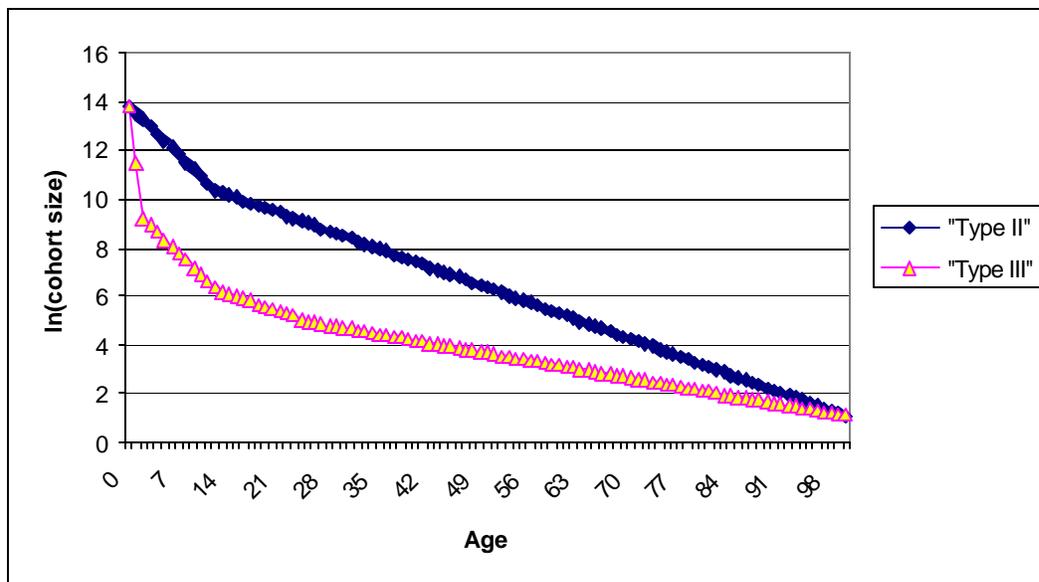
Most assessment methods require some estimate of population size to determine if takes represent a significant threat. Even methods that rely on changes in vital rates, such as elasticity analysis, require an estimate of population size to put quantitative boundaries on the potential survival rate changes.

Potential Biological Removal and similar options utilize  $N_{\min}$ , an estimate of population size based on a lower confidence interval. Population size may be subpopulation, life stage, or sex specific, but the resulting removal maxima will also be restricted to the animals described in  $N_{\min}$ . For marine mammals,  $N_{\min}$  can be a relatively straightforward calculation based on extrapolations of survey data, once the stock structure of the population or species has been determined. For sea turtles, with multiple life stages in multiple habitats,  $N_{\min}$  is a more problematic calculation. We generally only have an idea of adult female population size,

although cohort size could also be calculated with information on hatchling production. Most importantly, we know almost nothing about early life history of sea turtles. The performance of PBR as a management model for evaluation of take was based on an analysis of simulations that utilized a simple logistic growth model. The approach ignores demographics altogether as a factor regulating population growth rates and truly does rely upon an estimate of total population size. Thus the back-calculation and extrapolations required to estimate total population size for sea turtles makes PBR in its original form impractical for sea turtles (TEWG 2000).

The shape of the survivorship curve has an enormous impact on population size, even orders of magnitude (Figure 1). If most sea turtle in a cohort die within their first year or so, the population size of juveniles and adults will be much smaller than if hatchlings experience a relatively high survival rate to age 1.

A.



B.

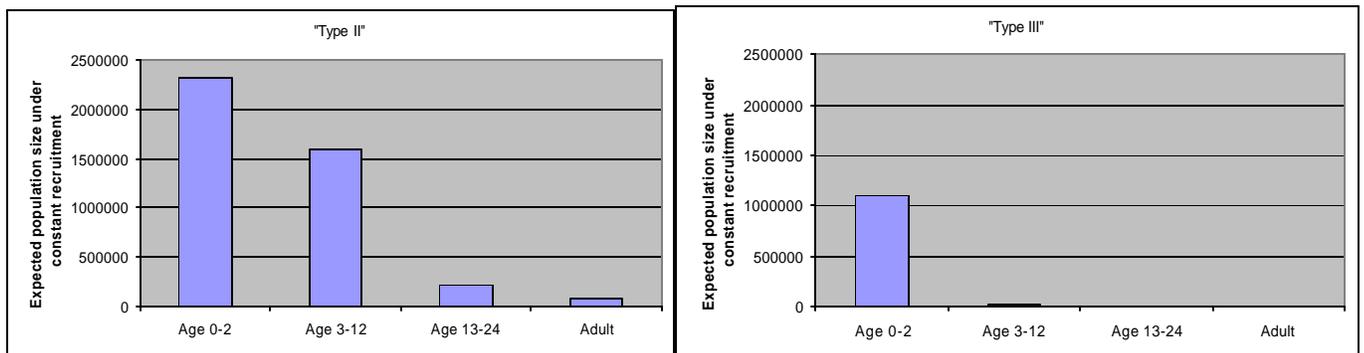


Figure 1. Expected population size from two plausible survivorship curves for sea turtles. “Type II” represents a more constant annual survival rate throughout life. “Type III” represents a curve where most mortality occurs in the first 2 years of life. A. Natural log transformed curves for Type II and Type III cohorts that start with 1,000,000 hatchlings. B. Summed numbers at age for populations with each type of survivorship curve, assuming a stable population with constant recruitment.

Extrapolation of population size from age-structured models is possible with back calculation based on adult female population size, proportion neophytes (first time nesters, assumed to all be age x), adult sex ratio and an assumption that the population is at or near a stable age distribution. This back calculation, while crude, may be better than extrapolation of in-water survey estimates and can include bootstrapped confidence intervals if appropriate.

As an example, Kemp’s ridleys are thought to mature at about 12 years of age (Snover 2002) and researchers at Rancho Nuevo have estimated that around 30% of nesters are there for the first time (TEWG 2000). The population has been increasing at 12-15% per year (Heppell et al. 2005). We can calculate the number of 11 year-old females (assuming “knife-edge” maturation at age 12) as:

$$N_{11 \text{ year-olds}} = (N_{\text{adults}} * \text{proportion neophyte}) / (S_{\text{subadult}} * \lambda_{\text{hat}}), \quad (2)$$

where  $N_{\text{adults}}$  = (observed nests/nests per female)\*remigration interval,  $S_{\text{subadult}}$  is the annual survival rate of subadult turtles, and  $\lambda_{\text{hat}}$  is the observed mean growth rate of the population, preferably based on hatchling production. This last variable accounts for increases or decreases in average cohort size over time. In 2004, there were 6446 nests counted at the three primary nesting areas in Tamaulipas. Thus, the estimated number of neophyte nesters is 1,547 and the number of female 11 year-olds is 2,204. If sex ratio of subadults mimics the primary sex ratio of 0.7 (female bias), the total estimated number of 11 year-olds in the population in 2004 would be 3,150. This calculation can be repeated to estimate the expected number of 10, 9, 8, 7 and 6 year-olds to get a population size estimate for all subadult (large benthic juvenile) age classes (Figure 2). In the case of Kemp’s ridleys or other stocks with good population-wide counts, the actual expected differences in cohort strength could be used to refine the estimate of population size.

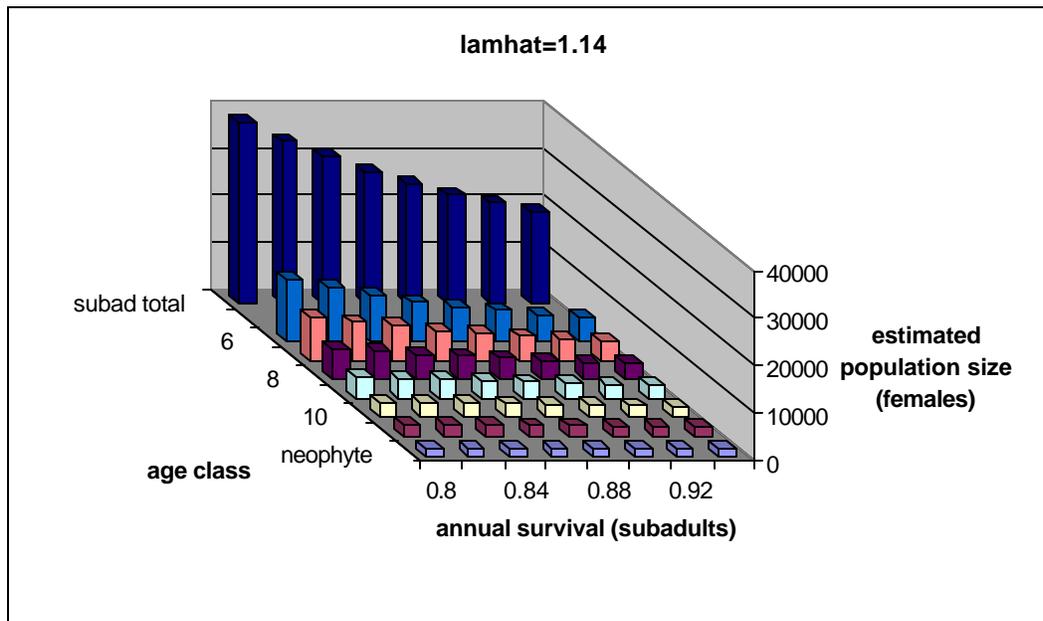


Figure 2. Back-calculated population size for subadult Kemp’s ridleys assuming 30% neophyte nesters, a 14% per year increase in cohort size ( $\lambda_{\text{hat}}$ ), and constant annual survival rate for ages 6-11.

Population size can also be estimated from in-water surveys, but extrapolation may be difficult and highly uncertain. Aerial and shipboard censuses may be more uncertain for turtles than for marine mammals, given the large and variable amount of time that turtles spend underwater (Lutcavage and Lutz 1997). Any assessment measure that relies on population size should specify the use of a lower confidence interval estimate of abundance and be population and/or life-stage specific, when possible. Trends in abundance and size distributions will provide critical information for verifying model-based population abundance estimates.

### **Methods based on the estimated value of individuals**

The long time to maturity for sea turtles means that an individual juvenile turtle is likely to die before it reaches adulthood and contributes to population growth. Clearly, removal of 100 hatchling turtles is less of a worry than removal of 100 subadult turtles that have survived many years of natural and anthropogenic mortality risk. Reproductive value (Fisher 1930) is a measure of an individual's current and expected contribution through reproduction, based on its current age, the probability of reaching maturity, and expected life span. Recent threat assessments for loggerhead and Kemp's ridley turtles have evaluated stressors according to the rough number of removals and the mean reproductive value of the age classes thought to be affected. This allows an evaluation of relative risk of various stressors and to an extent accounts for natural mortality. The evaluations rely on a calculation of "adult equivalency" based on the reproductive values of each age class as given by a deterministic, age-structured matrix model, and effectively "discount" the risk posed by stressors that affect juveniles.

Although calculation of age-specific reproductive values requires a complete life cycle model, thus requiring estimation of many unknown age-specific survival rates, there are some useful shortcuts that can reduce uncertainty in the reproductive value calculations. For age-structured models that group adults into a single life stage, with indeterminate lifespan based on a constant annual survival rate, adult reproductive value (*AdRV*) can be calculated as:

$$AdRV = \frac{F * s_{Ad}}{1 - s_{Ad}}, \quad (3)$$

where *F* is annual fertility (female offspring per female per year, which must account for remigration interval), *s<sub>Ad</sub>* is the annual survival rate of adults, and *I* is the asymptotic population growth rate of the population, as expected once the population has reached a stable age distribution (Heppell et al. 2000). This relationship generates a response surface of predicted adult reproductive values, illustrating the effects of uncertainty on *AdRV* (Figure 3). Unfortunately, uncertainty in remigration interval or other reproductive parameters can have a substantial effect on *AdRV*, particularly in declining populations ( $\lambda < 1$ ). Thus, methods based on reproductive value and adult equivalents are best for relative comparisons within species.

One potential drawback of utilizing adult equivalents to discount the impact of juvenile removals is that the inverse relationship between  $\lambda$  and *AdRV* leads to lower adult equivalency for *juveniles* in declining populations. While removal of adult individuals is more critical when  $\lambda < 1$ , due to an increase in adult reproductive value (Figure 3), individual juveniles will then be "worth" comparatively less and higher takes will be allowed unless additional recovery factors are included in the equation for maximum allowable removals.

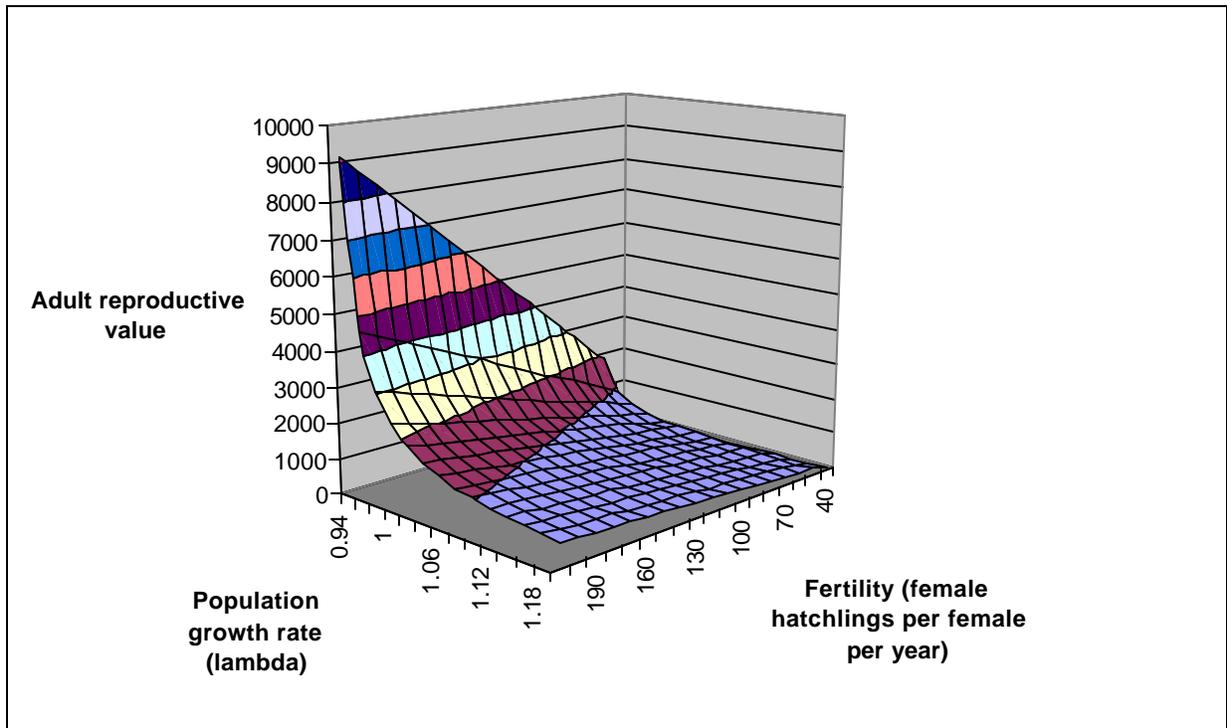


Figure 3. Response surface of adult reproductive value based on a deterministic model for a species with an adult survival rate of 0.92 (Eqn. 3).

Because jeopardy rulings are based on an assessment of the removals of individuals, rather than a change in vital rates, it may be useful to explore approaches that treat individuals as potential vital contributors to population recovery. Instead of evaluating removals based on mean reproductive value, which does not change relative to population size, removals could be evaluated as a “sampling without replacement” exercise. The potential for an individual to be a critical contributor to population growth is determined by age and a series of binomial probabilities that vary according to demographic stochasticity. The importance of an individual increases with a decrease in population size according to a hypergeometric distribution in this case.

Assessment of the risk of removing individuals in a probabilistic framework is a radical departure from matrix-based approaches that treat groups of individuals in aggregate. Aggregate models based on average survival and reproductive rates, however, may not capture the importance of individual removals. Individual-based models have been compared with aggregate structured models (Huston et al. 1988, Scheffer et al. 1995) and often lead to very different distributions of size and age than expected from average vital rates applied to groups of individuals.

Exploration of an individual-based approach to jeopardy assessment, including appropriate thresholds, requires further study.

## Methods based on population trends and extinction risk

Dennis et al. (1991) provided an analytical tool for evaluation of extinction risk from stochastic processes. The model relies only on a time series of abundance and calculates the probability of extinction over a set time period based on the size, exponential trend, and year-to-year variance of a population. As a population is projected stochastically, where

$$N_{t+1} = N_t e^{m+s^2}, \quad (4)$$

the resulting trajectories will spread in a predictable way according to a normal distribution.  $\mu$  and  $\sigma^2$  are the mean rate and variance of annual change in the population, calculated as

$$m = \text{mean} \left( \ln \left( \frac{N_{t+1}}{N_t} \right) \right), \sigma^2 = \text{var} \left( \ln \left( \frac{N_{t+1}}{N_t} \right) \right). \quad (5)$$

The model describes a diffusion process with drift, and is commonly referred to as a “diffusion approximation” of extinction risk (Dennis et al. 1991). The extinction risk for a population can be estimated as the probability that the population declines from some initial size  $N_0$  to some lower size  $N_q$  within a specified amount of time ( $t_e$ ). Absolute extinction probability is assessed when  $N_q = 0$ , but often risk is evaluated according to a “quasi-extinction threshold”, or a population size from which the population is unable to recover. Quasi-extinction probability can be analytically calculated as:

$$\Pr(N_0 \rightarrow N_q \text{ within } t_e) = p' \Phi \left( \frac{-\ln \left( \frac{N_0}{N_q} \right) + |m| t_e}{\sigma \sqrt{t_e}} \right) + e^{\left( \frac{2 \ln \left( \frac{N_0}{N_q} \right) |m|}{\sigma^2} \right)} \Phi \left( \frac{-\ln \left( \frac{N_0}{N_q} \right) - |m| t_e}{\sigma \sqrt{t_e}} \right), \quad (6)$$

where  $\Phi$  is the cumulative normal distribution function, and

$$p' = \begin{cases} 1 & m \leq 0 \\ e^{-2m \ln(N_0 / N_q) / \sigma^2} & m > 0 \end{cases}.$$

The general results of this model are that the probability of quasi-extinction increases when  $N_0$  is small, when  $t_e$  is long, when  $\mu$  is low and/or when  $\sigma^2$  is high. The approximation assumes that all parameters remain constant through time; thus, it is used as a comparative index of population health or to set general recovery goals (McClure et al. 2003).

In the original model proposed by Dennis et al., significant upward biases in  $\sigma^2$  occur for many time series, particularly when there is a high level of sampling error and when only a portion of the total population is sampled. This bias is partly due to a cumulative effect of sampling error on  $N_{t+1}/N_t$ . Holmes (2001, 2004) has developed smoothing techniques that reduce the error bias, thereby providing a more accurate estimate of  $\sigma^2$ . Holmes also determined that  $\sigma^2$  is biased by

autocorrelation in the time series, and suggests using a running sum based on generation time instead of raw population numbers.

For sea turtles, population time series are for nests or nesting females, a small fraction of total population size. Also, generation time for most species is much longer than the available time series, so data smoothing cannot be based on generation time. Snover and Heppell (in prep) recommend quasi-extinction risk evaluation based on diffusion approximation to use a running sum of  $x$  years determined by the mean remigration interval of the species or population. This provides a population size estimate that is closer to the number of adult females and reduces some of the variance in nest numbers that is caused by variable remigration intervals. An expected reduction in population size, such as the probability of a 90% reduction in the number of nests or nesting females, may be a better metric than “extinction risk” for evaluation of nesting beach trends. One disadvantage of the 90% reduction threshold is that population size is not considered – evaluation of risk to 2 populations with the same  $\sigma^2$  and  $\mu$  but  $N_a=1000$  and  $N_b=10,000$  would be the same.

The analytical framework provided by the diffusion approximation equation has great potential for providing “rules of thumb” for populations of a general size, year-to-year variance, and status. Response surfaces of “viable” vs. “unviable” take levels across a range of population sizes can be generated, allowing a visual assessment of maximum allowable removals.

### ***Methods based on recruitment to nesting population***

Recruitment to the nesting population holds great promise for evaluation methods if reliable techniques to identify neophytes can be found. Proper identification of neophytes requires saturation tagging (along with an assumption of strong nesting site fidelity) or, preferably, anatomical, physiological, or morphometric markers. An estimate of recruitment to the adult population allows separation of changes in population size due to survivorship to maturity and cohort strength from changes due to adult female survival or remigration interval. However, some caution is required because the proportion of a nesting population that consists of females reproducing for the first time can change for a variety of reasons. A higher proportion of neophytes is expected in a growing population, at least until the adults reach a new stable age distribution, but can also be due to an increase in adult mortality.

### ***Methods based on elasticity analysis of the life cycle***

Elasticity analysis determines the proportional change in population growth rate following a proportional change in a vital rate (survival, growth or reproduction) or a transition matrix parameter (de Kroon et al. 1986; Caswell 2001). Elasticity analysis of deterministic matrix models has been used for sea turtles to compare the relative effectiveness of management options that alter survival rates in terms of their impacts on the asymptotic population growth rate,  $\lambda$  (review in Heppell et al. 2003). If a complete life table or transition matrix is available, the elasticity of  $\lambda$  to a proportional change in a matrix entry  $A_{i,j}$  is dependent of the reproductive value vector  $v$  and stable age distribution vector  $w$ :

$$E_{i,j} = \frac{A_{i,j} v_i w_j}{\mathbf{1} \langle v, w \rangle}. \quad (7)$$

When the analysis is based on changes in survival rates of one or more age-classes and adults are grouped into a single stage, the elasticity values can be estimated without a complete life table, so long as an estimate of  $\lambda$  is available (Heppell et al. 2000). These estimates are relatively robust to uncertainty and permit evaluation of the life cycle when juvenile annual survival rates are unknown.

To apply elasticity analysis in jeopardy evaluation, the general premise would be that removals of turtles of a life stage with average elasticity  $x$  should not exceed a rate that reduces the asymptotic population growth rate  $\lambda$  by a fixed percentage. Removals can be estimated as a change in annual survival ( $S$ ) if natural and anthropogenic annual deaths can be divided by an estimate of the population size for the affected life stage. Alternatively, the total instantaneous mortality of a cohort ( $Z$ ) is calculated with or without the removals, where

$$S = \exp(-Z). \quad (8)$$

and  $S_2/S_1$  gives the proportional change in  $S$  expected from  $Z$ s calculated with and without additive removals (deaths).  $Z$  can be based on the slope of a line fit to ln-transformed numbers at age, and can include removals additively. The predicted proportional change in annual survival of a life stage could then be multiplied by the elasticity value for that life stage to get an estimated proportional change in  $\lambda$  (Heppell 1998, Caswell 2001).

There are 2 main problems with this approach. First, removals must be translated into a proportional change in annual survival. With limited data on population size and removals themselves, such a calculation could be very tenuous. To estimate the change in  $S$  using a catch curve approach ( $Z$  calculation), annual removals must be allotted to each age class. The second problem is that, like reproductive values, elasticities are based on predicted changes in population growth when the population is at a stable age distribution. Maximum removal calculations could severely under- or over-estimate allowable take for populations that are undergoing recent population recovery or decline, respectively. Elasticity analysis is unlikely to be a useful tool in the direct evaluation of removals, but may be useful for comparative analysis of relative impacts of stressors.

### ***Methods based on population productivity***

In fisheries stock assessment, allowable removals are based on population productivity and a pre-set population size target based on some proportion of “virgin stock biomass”, or the biomass of the fish stock prior to exploitation. A critical management level designated as “overfished” occurs when the population size estimate is less than 25% of virgin stock biomass. Unlike the exponential growth models commonly applied to sea turtles, the population is assumed to be capable of recovery at this low level so long as fishing mortality can be reduced; in other words, 25% virgin stock biomass is still well above the “depensation” point at which recruitment would not be able to exceed mortality. Fisheries models are generally based on a curved relationship between adult female stock and recruitment, either with reduced recruitment at very high female stock density (“Ricker curve”) or an asymptotic relationship between stock and recruits (“Beverton-Holt curve”).

There are two arguments against relying on a stock-recruit curve and productivity analysis for sea turtles. First, most sea turtle populations are thought to be well below their virgin stock biomass, at a point where exponential growth is expected and productivity is maximized (unless

density-dependence is operating in the form of reduced reproduction due to low population density). The exponential recovery of Kemp's ridley and olive ridley nests in Mexico lends support for this argument. Second, data supporting density-dependent productivity in sea turtles is limited to a few nesting beach examples, where the density of females clearly affects survival of the eggs and hatchlings, and two foraging ground studies (juvenile growth rates in Bahama and Australian green turtles). However, empirical data to support stock-recruit relationships in fishes are also quite rare. In fact, the "steepness" parameter of the relationship between relative stock abundance and relative recruit abundance ( $h$ ) is often a major unknown in fisheries stock assessments, along with natural mortality. While it may be natural to assume that density-dependent productivity occurs in sea turtles, the functional relationship can have enormous impacts on our estimates of population recovery rates, population size, and allowable take. As in fish stock assessments, detailed sensitivity analyses and construction of appropriate confidence intervals that express our uncertainty in density-dependence should be required.

Virgin stock abundance or biomass has not been estimated for most sea turtles, with the exception of simulation models by Chaloupka (2002). The models used in fisheries stock assessments to back-calculate this number generally require age distribution information of catches to estimate fishing mortality rates and selectivity curves that determine the probability of catching individuals of various ages/sizes. Fishery dependent and fishery independent data are needed to accurately assess mortality rates and population distribution. These data are generally quite limited for sea turtles, although improvements in age-length keys through skeletochronology may make this type of analysis possible in the future.

Regardless of data improvements, production- and virgin stock biomass approaches for sea turtles must carefully consider the form of the stock-recruit relationship and the potential for density-dependence. Clearly there are populations at low density that have continued to decline, even when removals are drastically reduced. With the exception of stocks that are enhanced through egg protection, there is little evidence for rapid population response following a reduction in sea turtle takes. This may be due to time lags, particularly in the late age at maturity species, or a failure of management to substantially reduce total removals.

## Option 1: PBR-like approach

*Premise:* Mortalities (removals) should not exceed a minimum estimate of population size multiplied by ½ of the maximum annual population growth rate of a population, multiplied by a “recovery factor” based on population status.

*Meets management model criteria:* 1, 2\*, 3, 4, and 6.

\* see “Cons”, below

Potential Biological Removal (PBR), was developed for marine mammal stocks (Barlow et al. 1995), which share many life history traits and data restrictions with sea turtles. PBR is based on the concept that human-caused mortality of a protected mammal stock cannot exceed one-half the potential net productivity rate of the population, adjusted by a recovery factor (F) that varies from 0.1 to 1. It is a simple equation that requires a minimum population size estimate ( $N_{\min}$ ), the maximum rate of increase predicted (or measured) for a population ( $R_{\max}$ ), and pre-determined risk criteria (low risk to minimal risk) for the recovery factor:

$$\text{PBR} = N_{\min} \times (0.5 \times R_{\max}) \times F$$

Wade (1998) ran stochastic simulations for a number of marine mammal life histories to determine appropriate recovery factors for endangered and threatened populations. The default F values are: widespread populations with little risk of depletion,  $F = 1.0$ , threatened species  $F = 0.5$ , and endangered species  $F = 0.1$ . It has been determined that a stock experiencing a level of human-caused mortality exceeding PBR could be depleted and would likely fail to reach recovery goals (Wade 1998; Taylor et al. 2000). Discussions with the developers of the marine mammal PBR methodology have suggested that with modification, such as calculating separate PBRs for each life stage, PBR is potentially applicable to sea turtles.

PBR represents total removal from a population (above “natural” mortality, which is reflected in the maximum growth rate) and thus requires a separate step for allocation of takes to various fisheries. The population or stock also must be defined for all parameters. The model does not include age structure, so  $R_{\max}$  is assumed to be constant across age classes as it would be for a population at stable age distribution.

### *PBR calculations for sea turtles*

The PBR estimate varies depending on each of the three required parameters. As an example, Figure 4 shows changes in PBR with various recovery factors when  $R_{\max}$  is set at 0.12 or 0.05. For a population of 2500 turtles, total allowable human-induced take would range from 10 to 150 individuals. The lowest PBRs occur for very slow-growing populations, where additions to the population (recruitment) each year are only a small proportion of the total population size. While PBR numbers can be very low if conservative recovery factors are used, it is important to realize that PBR represents a conservative estimate of allowable take to achieve an Optimal Sustainable Population size (Wade 1998). For sea turtles, PBR could be based on a desired population size or recovery rate, or a turtle-specific adaptation of the OSP.

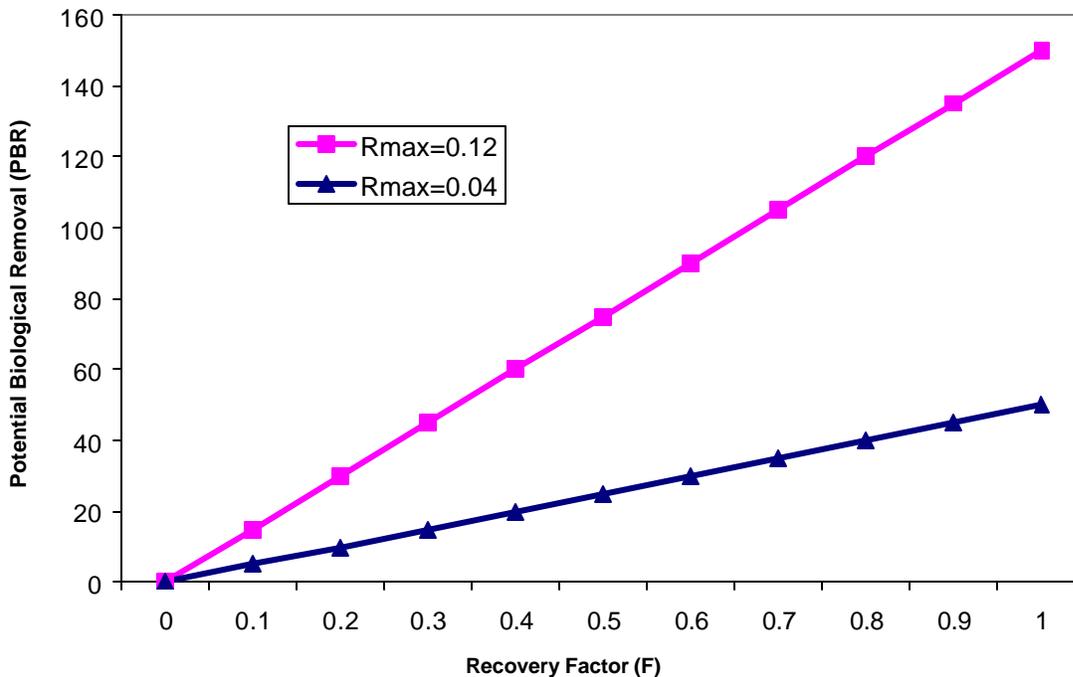


Figure 4. Effect of the recovery factor (F) on PBR for a population of 2500.

To meet evaluation method criteria, PBR for sea turtles should be based on a minimum population size estimate for adult females until a reasonable estimate of total population size can be determined. Past application of PBR to recommendations for strandings limits relied exclusively on adult removals and did not provide a limit on takes of subadult or juvenile turtles (TEWG 2000). Application of adult equivalents using reproductive value allows evaluation of juvenile turtle takes relative to the adult PBR.

#### *Parameters*

$N_{\min}$  = lower confidence interval on total adult population size (60% is generally used for marine mammals). Requires extrapolation of nesting beach numbers to adult females and an estimate of adult sex ratio.

$R_{\max}$  = maximum population growth rate, can be based on observed trends on nesting beaches (which include enhancement). The latter should be based on pooled nesting beach totals and standardized by index beach trend estimation whenever possible.

F = recovery factor. Endangered or declining populations = 0.1, Threatened populations = 0.5. Others = 1.0

#### *Jeopardy Evaluation Suggestions*

Cumulative annual removals (kills) must be evaluated relative to the PBR estimate. Removals of juveniles can be evaluated separately based on adult equivalents, because  $N_{\min}$  is based exclusively on the adult population. To quantitatively address juvenile removals,  $N_{\min}$  should be estimated for the life stage(s) affected. Evaluation of individual threats could be based on the proportion of PBR affected as a relative measure of risk/impact (see example, below).

*Pros:*

- relatively simple, can be generalized to all species
- “proven” in marine mammal management

*Cons:*

- Many simplifying assumptions, little biology.
- Requires population size estimate (including extrapolation for both sexes unless sex ratio of takes is known).
- Requires assessment of all potential human-related mortalities and allocation of those takes, and some infrastructure to support take evaluation or reduction teams.
- Stock structure should be taken into account, but this would require knowledge of the proportion of each stock in the total take, in individual takes, and the status of each stock.
- Declining populations are common for sea turtles, so relying on a maximum potential growth rate may not be conservative for some species.
- Likewise, populations that are recovering rapidly due to enhancement on nesting beaches, PBR may be too low.
- Due to the low population growth rate expected for many species, the PBR estimate is very low. It is quite possible that current takes already exceed PBR for many species, possibly by a substantial margin. How, then, can new (and old) fisheries be evaluated?

*Alternatives*

1. PBR could be life stage specific (requiring extrapolations of  $N_{\min}$ ) or based on hatchling equivalents rather than adult equivalents.
2. No RV adjustment (more conservative PBR) for declining or endangered stocks.
3. Multiply by a factor based on the ratio of current population growth to the growth rate required to achieve a desired recovery goal. This factor could be greater than 1.0 for populations that are rapidly increasing due to enhancement.
4. Do PBR assessments on a regional basis. This would allow managers to consider takes relative to the abundance of turtles in their jurisdictional waters, rather than over the entire population. One disadvantage to this approach is in defining  $R_{\max}$ , and the immigration and emigration rates for the region would need to be assessed.

EXAMPLE:

500 subadult and juvenile loggerheads killed

$$500 \times 0.8 = 400 \text{ adult equivalents}$$

$N_{\min}$  adult loggerhead population (SE US) = (60,000 nests/3.5 nests per female x 3 year remigration)/0.75 adult sex ratio = around 75,000 adults

$R_{\max} = 0.04$  with nest survival enhancement (4% per year increase S Florida observed 1988-2000)

$F = 0.5$  threatened status

$$\text{PBR} = 750 \text{ adults } (75,000 \times (0.04 \times 0.5) \times 0.5)$$

Result:  $400/750 = 53\%$

In this example, a single stressor is evaluated that does not exceed the PBR. However, *cumulative impacts* of multiple takes (mortalities) must be considered. For PBR to work effectively, an assessment of all human-related mortalities, at minimum, should be undertaken, followed by an allocation of take to various fisheries. Human-related impacts that are not measured or managed will be problematic.

One way to evaluate single stressors could be to set impact thresholds based on the percent of PBR that is taken. For example, if estimated annual take for fishery x is:

<10% of adult PBR = no jeopardy

10-25% of adult PBR = strict monitoring and take reduction efforts required

>25% of adult PBR = jeopardy evaluation, restrictions and take reduction required

This approach would not negate the need for cumulative impacts assessment, but could serve as a starting point for evaluation.

## Option 2: Recruitment to nesting female population approach

*Premise:* total removals (based on adult equivalents) should not exceed observed or expected recruitment to nesting population (neophytes), or a desired recruitment level based on population recovery goals.

*Meets management model criteria:* 1\*, 2, 3, 4, 5 and 6.

\* requires estimate of recruitment to adult population, which is currently unknown for most species

This is really a variation on the PBR approach that was originally proposed by Colin Limpus.

MR = “maximum removals”, which in this option are “neophyte-equivalent removals”, or maximum total adult removals allowed, based on observed recruitment of females to the nesting population. Use adult equivalents for take of juveniles, and divide by proportion female (by life stage if available, otherwise use primary sex ratio).

$$MR = \text{neophyte/total nesters} \times N_{\min}/(\text{prop female}) \times \text{maxreduction}$$

### *Parameters*

$N_{\min}$  = lower 95% confidence interval on total number of nesting females. Requires extrapolation of nesting beach numbers to adult females and an estimate of adult sex ratio.

*Maxreduction* = maximum reduction in recruitment acceptable, similar to the recovery factor. For populations that are in the recovery process, this would need to be quite low.

### *Jeopardy Evaluation Suggestions*

Same issues as PBR (Option 1) for extrapolation to whole population and total removals (cumulative impacts). Removals of juveniles could be based on adult equivalents.

### *Pros:*

- Simple, logical
- Possibly more palatable than PBR
- Can be modified to account for population status, desired recovery goals

### *Cons:*

- requires recruitment estimate, currently unavailable for most populations
- proportion of neophytes in a population may be high if adult survival is too low to sustain the population, so this would not be a conservative estimate of allowable take for a population that is declining due to reduced adult survival.
- time lags = disconnect between takes of juveniles and recruitment to nesting population

### *Alternatives:*

1. Minimum recovery rate approach (see PBR Alternative 3)
2. Determine maximum removal of adults or adult equivalents given recruitment levels

Example (single impact assessment):

If estimated annual take for fishery x is:

<10% of adult MR = no jeopardy

10-25% of adult MR = imperiled - strict monitoring and take reduction efforts required

>25% of adult MR = jeopardy evaluation, restrictions and take reduction required

*Note:* this is a possible way to evaluate single stressors, but could still lead to a cumulative impact that exceeds MR.

500 subadult and juvenile loggerheads killed

$$500 \times 0.8 = 400 \text{ adult equivalents}$$

$N_{\min}$  nesting female loggerhead population (SE US) = (60,000 nests/3.5 nests per females) = around 17,000 nesting females

Prop neophyte = 0.2 with nest survival enhancement (guess)

Sex ratio = 0.7

Maxreduction = 0.5

MR = about 2500 adults or adult equivalents

Result:  $400/2500 = 16\%$ , no jeopardy for this fishery but monitoring required

### Option 3: Other recruitment-based approaches

*Premise:* numbers of animals caught + numbers that die of natural causes should not equal or exceed the number of births, which may be density-dependent. To meet recovery goals, deaths should be less than births to sustain a growing population.

*Meets management model criteria:* 2, 3, 4, and 6.

\* see “Cons”, below

For each life stage,

$$N_R \leq N_r - N_M - N_g, \quad (9)$$

where  $R$  = removals,  $r$  = recruitment,  $M$  = natural mortality and  $g$  = growth into the next life stage. Eqn. 9 can be expressed as an ordinary differential equation or as a discrete difference equation, where the latter is probably more conducive to assessment based on annual removals of individuals. The recruitment of individuals into a life stage may be time dependent, based on known changes in population growth or cohort strength, and would not necessarily require the population to be at a stable age distribution.

#### *Parameters*

All parameters should be based on precautionary confidence intervals. The annual number of removals must be cumulative within a life stage.

#### *Pros:*

- Simple
- More familiar to fisheries scientists?
- Could be applied to a regional aggregation of turtles, rather than the entire population ( $N_r$  becomes recruitment into the region)

#### *Cons:*

- Required parameters are difficult to estimate and may require assumption of stable age distribution, although analytical relationships based on life cycle characteristics may be possible.
- $N_r$  and  $N_g$  will be difficult to estimate, but could be derived from mark-recapture studies

#### *Alternatives:*

1. Could produce an aggregate estimate instead of life stage-specific, akin to Total Allowable Catch in fisheries assessment. Fewer parameters but time lags are an issue.
2. Could use a life table approach without a stock-recruit function with the premise that to achieve population stability,  $\ln(\lambda)=0$ , each female must replace herself, and removals are equivalent to reducing that probability. The equations would be based on net reproductive rate,  $R_0$ , where  $R_0$ =survivorship to maturity x lifetime expected offspring production. Allowable removals would be based on expected lost productivity (with reproductive value incorporated), thereby affecting the population growth rate. The latter could be based on a desired recovery rate instead of population stability.

## Option 4: Diffusion approximation (PVA) approach

*Premise:* removals scaled as adult equivalents should not increase risk of extinction or decrease probability of recovery by more than a fixed amount, according to a diffusion model based on time series of the population.

*Meets management model criteria:* 1, 2, 3, 4, and 5.

MR = number of adult female or nest equivalents removed to increase probability of extinction over a pre-set time period by x%

This should be solved analytically by determining the change in  $N_0$  required to increase extinction risk by a set percentage, or to increase time to population recovery by a set percentage.

### *Parameters*

$t_e$  = time horizon for evaluation. Although this is often set at 100 years for extinction risk evaluation, this is arbitrary.

$\sigma^2$  = variance of same time series

$\mu$  = lower 95% confidence interval of the mean of  $\ln(N_{t+1}/N_t)$  for a time series of nest abundance for the species or stock. Requires extrapolation of index nesting beach numbers to total population size unless the “extinction risk” is based on the probability of a 90% reduction in abundance.

### *Jeopardy Evaluation Suggestions*

Evaluate takes according to maximum number of potential removals from weakest stock affected, as the extinction probability of increasing or very large stocks will be so small that calculation of change is meaningless.

### *Pros*

- Suitable for increasing and decreasing populations
- Analytical result may lead to general “rules of thumb” for recovering, stable or declining stocks
- Desired rate of recovery can be specified by Recovery Plan recommendations.

### *Cons*

- Extrapolation as shown here is somewhat convoluted, particularly due to the need to use a running sum of nests to reduce variance bias
- Diffusion approximation = “Black box” to stakeholders?
- Time lags – do juvenile removals really translate into adult population loss?
- Only time series available for sea turtles are nest or nesting female counts. The former is particularly problematic (see Considerations, above).

### *Alternatives:*

- base on quasi-extinction threshold instead of 90% reduction
- alternative extrapolation/equivalency calculations
- equate removals to change in  $\mu$  rather than N

Example (Northern Reproductive Unit): 500 subadult and juvenile loggerheads killed

$500 \times 0.8 = 400$  adult equivalents,

$400 \times 0.7$  sex ratio = 280 adult female equivalents

Proportion that are from Northern Reproductive Unit = 0.6, NRU adult equivalent take = 170

$170/2.5$  year remigration interval)  $\times 3.5$  nests/female = about 240 “nest equivalents” = removal per year,  $240 \times 3 = 720$  total nest equivalent removals

Probability of a 90% reduction in the number of Northern Reproductive Unit nests in a 100 year time frame (3 year running sum of nests,  $\mu = -0.008$ ,  $\sigma^2 = 0.02$ ) = 0.231

Reduction in  $N_0$  required to increase that probability by 5% = 720 nests over 3 years = MR

Result: jeopardy if only NRU considered

## Option 5: Trend analysis

*Premise:* Take trend/mortality trends should not exceed population trend(s). In particular, take trends (slopes) should be low enough to meet recovery goals.

Comparison of trends in removals should be part of required monitoring of any identified take, so this option is not exclusive of Options 1-4.

*Meets management model criteria:* 1, 2, 5 and 6.

MR = number of adult female or nest equivalents removed to achieve a removal trend that is less than observed population trend

This should be solved analytically by determining the change in  $N_0$  required to increase extinction risk by a set percentage.

Set Type II error level (probability of failing to detect a negative trend) according to population status; endangered populations should have high power ( $1-\beta$ ).

MR could be a target reduction in take that would result in a reduction in the removals slope ( $m_2$ ) to be equal to or less than the population index slope ( $m_1$ ).

### *Parameters*

$m_1$  = slope of population index

$m_2$  = slope of removals

Power ( $1-\beta$ ) to detect slope

### *Jeopardy Evaluation Suggestions*

Trend analysis is probably not suitable by itself as a method for jeopardy evaluation. However, the Turtle Expert Working Group (2000) proposed a possible method to set maximum takes (in this case, using strandings as a proxy for takes): determine the number of indexed removals next year or over a series of future years that would be required to achieve a take index slope that is equivalent to the population index slope. The primary problem with this method, and potentially any that involve trend estimation, lies in the significance of the slopes and power to detect differences in slopes. Proxies for take, in this example, strandings, are notoriously variable, and an exponential regression line often does not provide a good fit to the data. This suggested method for setting strandings limits was not agreed upon by all Group members, and a minority opinion was issued.

### *Pros*

- Suitable for increasing and decreasing populations
- Based on trends in indices of abundance rather than extrapolated  $N$
- Requires standardized monitoring of removals
- Is not based on total removals – can evaluate stressors individually

### *Cons*

- All the general problems with trend assessment, particularly significance of differences in slopes

- Time lags, which are particularly important if take = juveniles but population index = nests

*Alternatives:*

- Base decision on statistical analysis of differences in slopes, but with careful attention to power (Type II error)
- Use  $\mu$  instead of regression slope
- Base trend comparisons on index counts of same life stages (juvenile, subadult, adult) where possible to obtain such trends
- Adjust population index time series according to the time lag to age of removals (example, t-10 years for loggerheads if take = benthic immatures) – this will be problematic for late-maturing species

Example: Kemp's ridley Interim Strandings Limit (ISL), Gulf of Mexico (from TEWG 2000):

For Kemp's ridleys, it appeared that in many years the rate of increase of strandings exceeded the observed rate of increase of nests and hatchlings at Rancho Nuevo. A new method was introduced that worked on the key assumption that to maintain and enhance population growth that rate of increase of strandings should be lower than the population rate of increase. This method relied on the estimated slopes of hatchling production and strandings and set the ISL for the following year to obtain a desired slope. It was used on a trial basis to calculate an ISL for Kemp's in 1998, as a full analysis of potential changes in population size and strandings on the ISLs had not been attempted.

As a measure of population growth rate, the group agreed to use hatchling production 2 years prior to the strandings year. The strandings years used in the analysis were 1986-1997 to incorporate all available strandings data. The strandings did not include headstarted, TED-tested, or cold-stunned turtles and also excluded post-hatchlings.

*Slope calculation*

Hatchlings released 2 years prior and strandings were ln-transformed to calculate and compare slopes. Slopes and regression statistics were calculated with Excel 5.0. Over the 12 year time period the two slopes were nearly identical ( $m = 0.8$  for hatchlings and  $m = 0.78$  for strandings). Strandings were much more variable than hatchlings, but the regression was still significant at the 0.05 level ( $p = 0.028$ ). The standard error of the hatchling slope was 0.011.

*Setting the ISL*

Strandings in 1998 should not exceed a value that maintains or lowers the slope to a desired level. To set the ISL conservatively, the group agreed that the desired strandings slope should be 1 standard error below the slope observed for hatchling production. Using a minimization program in Excel (Solver), the number of strandings in 1998 was calculated to give a strandings slope of 0.69 over the years 1986 - 1998 ( $0.69 = \text{observed slope of } 0.08 - 0.11, \text{ the standard error}$ ). That point estimate was 334 turtles (Figure 5).

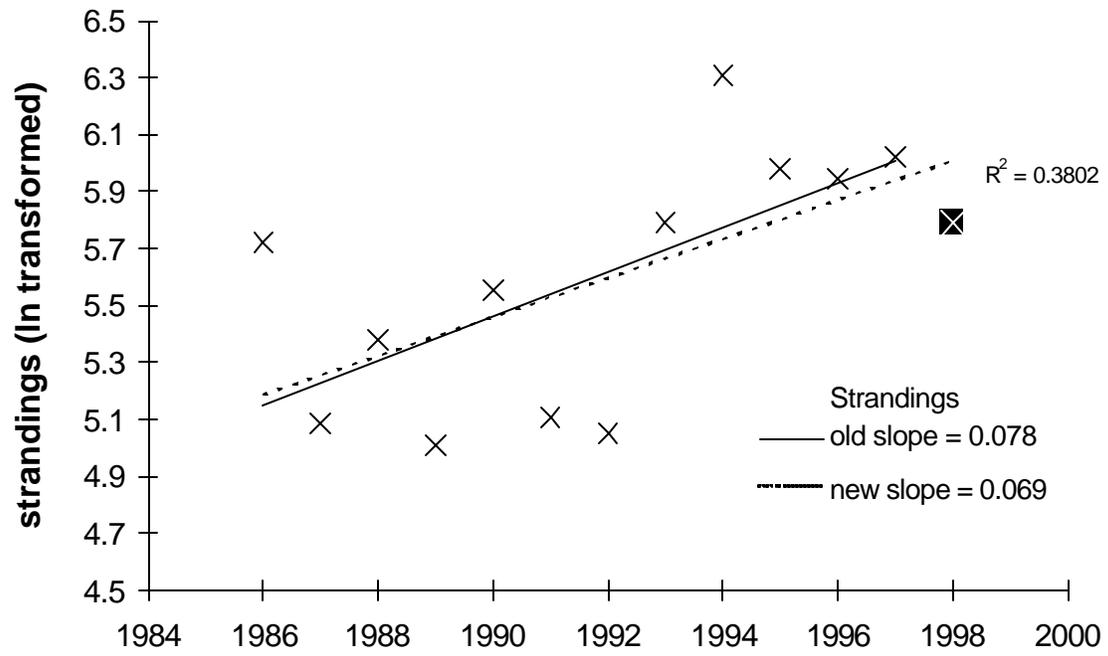


Figure 5. Evaluation of target reduction in strandings (as an index of removals) required to achieve the population index slope based on hatchling production at Rancho Nuevo. From TEWG (2000).

## Plan for analysis and performance evaluation

Some screening of options presented here is required because some may be impractical or less desirable by NMFS. For those options and alternatives that are preferred, the robustness of each method and evaluation of appropriate risk thresholds should be done through simulation analysis (for a nice example, see Holmes 2004). The population-level effect of removals can be simulated with stochastic age-structured models that include measurement error, variable remigration interval and age at maturity, and other bits of “biological realism” that may affect their efficacy. The structured models should include a generic model for “fast” species – Kemp’s ridley, leatherback – and a model for “slow” species – loggerhead, green and hawksbill turtles. Criteria for jeopardy thresholds should include a range and be set *a priori*, e.g., 5% chance of extinction in 3 generations and 20% chance of extinction in 3 generations. The performance of each option as a management tool should be evaluated according to its ability to correctly assess jeopardy, where probability of Type I error = overestimate allowable take, fail to make jeopardy call when it should be, and Type II error = underestimate allowable take, make a jeopardy call when it is not really necessary.

## Recommendations for development of sea turtle stock assessment methods

Sea turtle stock assessment can greatly benefit from establishment of standardized methods and/or models. Recovery planning, jeopardy evaluation, maximum allowable take estimation, and evaluation of the success of management measures all require stock assessment. To develop such tools, a panel of quantitative ecologists with different areas of expertise should meet to discuss alternative approaches, similar to the workshop conducted in 1995 for assessment of longline mortality in Hawaii (Bolten et al. NMFS Tech Memo NMFS-SWFSC-230). In contrast to that meeting, the goal of an assessment development workshop should be to develop new tools that can be applied to a range of populations and that account for data availability and uncertainty. Creative experts from fisheries stock assessment, marine mammal assessment, sea bird and sea turtle assessment who specialize in different modeling approaches should attend. An important outcome of such a meeting would be the identification of monitoring efforts or other data collection that are needed to parameterize assessment models.

One possible direction to take would be to truly evaluate the potential for application of traditional assessment methods, despite the paucity of data. In a Bayesian or other probabilistic framework, uncertainties in age structure, recruitment, and selectivity functions for fisheries and survey gear could be specified as priors.

An underutilized source of information for many sea turtle populations is the size distribution of groups of animals, such as bycatch, strandings, and in-water surveys. The size distribution in a sample of turtles only reflects the distribution for that subsample of the population, and is dependent on size-specific catchability. Thus, size distributions from a single year or location provide little information on population dynamics. However, shifts in size distributions from samples of animals collected in similar times and locations with similar gears can provide clues about changes in cohort strength and recruitment variability, population recovery or decline, and possibly changes in mortality rates (e.g., Seminoff et al. 2003, Hoyle and Maunder 2004). Reliable age-length keys are critical for this evaluation, and likely will need to be regionally

specific due to high variance in growth rates. Further exploration of the utility of size distributions in assessment of population status should be a high research priority.

Alternatively, there may be other ways to utilize existing or obtainable data to evaluate sea turtle status in a rigorous way. By convening a workshop of scientists with a range of quantitative and ecological expertise, modifications of traditional assessment methods and entirely new approaches can be discussed and evaluated.

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