

SECTION 1—INTRODUCTION AND BACKGROUND 1

None

SECTION 2—SPECIES OVERVIEW 5

Diet of oceanic and neritic juveniles has been documented for the Mediterranean – see papers by Lazar, Casale. However, these authors have also suggested that Mediterranean juveniles are difficult to classify as oceanic or neritic, as they utilize both habitat types.

SECTION 3—DETERMINATION OF DPS 13

Not reviewed.

SECTION 4—ASSESSMENT OF EXTINCTION RISK

4.1. Description of Extinction Risk Assessment Approaches

To assess extinction risks of loggerhead DPSs, we conducted two independent analyses. The first analysis used the diffusion approximation approach based on time series of counts of nesting females or nests (Lande and Orzack 1988, Dennis *et al.* 1991, Holmes 2001, Snover and Heppell 2009). This analysis provided a metric (susceptibility to quasi-extinction or SQE) to determine if the probability of a population's risk of quasi-extinction is high enough to warrant a particular status listing (Snover and Heppell 2009). This approach is based on stochastic projections of observed trends and variability in the numbers of mature females at various nesting beaches. The second approach focused on determining the effects of known anthropogenic mortalities on each DPS with respect to the vital rates of the species. Anthropogenic mortalities were added to natural mortalities and possible ranges of population growth rates were computed as another metric of population health. This approach focused on how additional mortalities may affect the future growth and recovery of a loggerhead turtle DPS.

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4.2. Computation of Susceptibility to Quasi-extinction (SQE)

4.2.1. Methods

Estimates of quasi-extinction risk are known to have high degrees of uncertainty due to the stochastic nature of populations and their environments and the error involved in data collection and subsequent parameter estimation (Holmes *et al.* 2007). However, there are ongoing needs for management to classify populations in terms of their status when only limited data, often with high observation error rates, are

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available. To address this need, Snover and Heppell (2009) presented a quasi-extinction risk index called susceptibility to quasi-extinction (SQE) that can be used to classify sea turtle nesting populations based on relative risks using only nesting beach census data. This index integrates parameter uncertainty and stochasticity in extinction risk forecasting while allowing managers to balance the risk of making Type I (considering a population to be not at risk when it is) and Type II (considering a population to be at risk when it is not) errors when making decisions. While they showed that the method is robust in assessing actual risk (in terms of a binary metric of “at risk” or “not at risk”) using population simulations, they clarify that SQE values are not indicative of a true probability of quasi-extinction because they assume constant distributions of trend and variance over long time frames (three generations) and ignore density dependence. Rather, the index serves as a tool for classifying populations by relative status.

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The technique involves standard methods of diffusion approximation (Lande and Orzack 1988, Dennis *et al.* 1991). These methods are based on a density-independent exponential model in a randomly varying environment:

$$N_{t+1} = N_t \exp(\mu + \varepsilon), \quad (1)$$

where $\varepsilon \sim N(0, \sigma^2)$, N is the population size, and t is time (Dennis *et al.* 1991, Holmes 2001).

Assuming that the lognormal distribution can be used to compute the probability that the population will be of a certain size, two parameters estimated by this method are μ , the arithmetic mean of the log population growth rate, and σ^2 , the variance of the log population growth rate, which accounts for sources of variability, including environmental and demographic stochasticity, and observation error (Holmes 2001, Morris and Doak 2002 - Chapter 5). These parameters were estimated using the regression analysis, whereas the confidence intervals were estimated using the method of Dennis *et al.* (1991). These estimated parameters were used to make inferences on total population growth rates and quasi-extinction probability.

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For each nesting beach or region, we followed the recommendations in Snover and Heppell (2009), and used a running-sum of 3 yr. This data smoothing provides a more accurate reflection of population change by reducing year-to-year fluctuations in nests that affect our translation of nests to adult females and actual population processes. Current adult female population size, n_0 , was estimated as the sum of the last 3 yr of data. We used the parametric bootstrap estimation procedure from Morris and Doak (2002) to compute the μ and σ^2 distributions required to calculate susceptibility to quasi-extinction (SQE) and a wide range of quasi-extinction thresholds (QETs). We used a range from 2.5 to 97.5% of the current abundance of nesting females as potential QETs. Because loggerhead turtles are likely to mature at > 30 yr (Snover 2002), we used the time period of 100 yr to compute QETs, which is consistent with the IUCN criteria (3 generations or 100 years, whichever is shorter). To incorporate the uncertainty of parameter

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- Comment [SH1]: Data = nests, right? Need to explain procedure for translation to adults. OK, explanation of count data follows this section. It would still be helpful here to state what the data are.
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estimates in determining SQE, we used 95% confidence limits of $\hat{\mu}$ and $\hat{\sigma}^2$, using the method of Dennis *et al.* (1991).

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The following steps were used to obtain SQE's:

1. μ_i and σ_i^2 were drawn randomly and independently from respective distributions, $\{\tilde{\mu}_i, \tilde{\sigma}_i^2\}$.
2. Each pair $\{\tilde{\mu}_i, \tilde{\sigma}_i^2\}$ was used to compute the probability of reaching a QET within the next 100 years.
3. Steps 1 and 2 were repeated 5000 times ($i = 1, \dots, 5000$) to create a distribution of probabilities of reaching the QET.
4. The SQE metric for the QET is the proportion of the probabilities that are >0.9 (cut-off probability).
5. Steps 1 through 4 were repeated for a range of QETs between 2.5% and 97.5% of the current abundance of nesting females.

Comment [SH2]: There is some inconsistency in explanation of the time horizon – 100 years may well be 3 generations for loggerheads, but that needs to be clear because 100 years is also used as an arbitrary time horizon for PVA generally. Time horizon should be specified in the figure legends.

Using simulations, Snover and Heppell (2009) demonstrated that SQE values greater than 0.4 indicated a population has >0.9 probability of quasi-extinction. At this critical value (SQE = 0.40), Type I and Type II errors are minimized simultaneously at approximately 10%. Reducing the critical value to 0.3 lessens the 'Type I' error rate but increases the 'Type II' error rate (Snover and Heppell 2009). The choice of 0.9 as the cut-off probability was arbitrary and values other than 0.9 could be used. However, new critical values other than 0.4 needed to be established for different values of the cut-off probability. Qualitatively, the results would not differ if a value other than 0.9 was used (Snover and Heppell 2009). In this assessment, we used the cut-off probability of 0.9 as in Snover and Heppell (2009) and a critical value for the SQE of 0.30, which reduced the 'Type I' error (a DPS is considered to be not at risk when in fact it is). SQE values greater than 0.30, therefore, indicate the DPS is at risk.

Comment [SH3]: Why? This seems arbitrary to me and needs to be justified if it is going to be used in your recommendations. The results are presented for a range of QETs which is great.

4.2.2. Count Data

The following datasets were used for the computations of SQEs. Ideally, the SQE analysis is conducted on the numbers of nesting females over time, which represent the temporal change in the number of reproductive females. The statistic, however, is difficult to determine because of the required effort to mark and identify individual turtles. Consequently, data are often collected for the number of nests per nesting season. To estimate the number of females from the observed number of nests, we use the average number of nests per female per nesting season for that region. Because the average is treated as a constant over time for each DPS (Table 1) and SQE is calculated as probability of a proportional reduction in population size, the SQE analysis is unaffected by the choice of unit (i.e., the number of females or nests). In the following, we use the number of females as the unit of analysis.

4.2.2.1. North Pacific Ocean DPS

All loggerhead nesting in the North Pacific occurs in Japan. Nesting data for this region are from two sources, the Sea Turtle Association of Japan (STAJ; unpublished data provided to the Western Pacific Fisheries Management Council) and Kamezaki *et al.* (2002). Data from the STAJ represent total counts for Japan from 1998-2007. Snover (2008) combined these datasets to achieve a time series from Japan from 1990-2007 (Figure 1). Due to the nature of the STAJ data, the data from Japan are represented by a single time series.

4.2.2.2. South Pacific Ocean DPS

We used nesting census data for index beaches in eastern Australia (Limpus 2009; Figure 2). These include mainland beaches, Wreck Rock beaches, Great Barrier Reef Coral Cays, and Wreck and Tyron Islands. Each beach was analyzed separately.

4.2.2.3. North Indian Ocean DPS

No adequate time series of nesting beach data was available for this DPS.

4.2.2.4. Southeast Indo-Pacific Ocean DPS

No adequate time series of nesting beach data was available for this DPS.

4.2.2.5. Southwest Indian Ocean DPS

Count data for the Southwest Indian Ocean DPS were obtained from Baldwin *et al.* (2003; Figure 3).

4.2.2.6. Northwest Atlantic Ocean DPS

The Recovery Plan for the Northwest Atlantic Population of the Loggerhead Sea Turtle (NMFS and FWS 2008) recognized five recovery units (subpopulations) of loggerheads within the Northwest Atlantic:

1. Northern Recovery Unit (FL/GA border through southern VA)
2. Peninsular Florida Recovery Unit (FL/GA border through Pinellas County, FL)
3. Dry Tortugas Recovery Unit (islands located west of Key West, FL)
4. Northern Gulf of Mexico Recovery Unit (Franklin County, FL, through TX)
5. Greater Caribbean Recovery Unit (Mexico through French Guiana, The Bahamas, Lesser Antilles, and Greater Antilles)

Of these recovery units, four have nesting beach data with adequate length time series to apply this analysis. The Northern Recovery Unit data and the Peninsular Florida Recovery Unit index beach data are both comprised of sums of numerous individual beaches (Figure 4). For the Northern Gulf of Mexico Recovery Unit data, we used a time series from the Florida Panhandle. We used nesting beach data from the Yucatan Peninsula (J. Zurita, personal communication, 2008) to represent the Greater Caribbean Recovery Unit.

Comment [SH4]: I don't think this was specified or explained in the methods above. Regardless, remind us here what that minimum is. Because the trends have changed over time for some areas, I think it is important to include some specifics about time series length and what years of data were used. Not clear if the figures are showing running sums or "raw" values – I would suggest that they should be the running sum time series used for each analysis.

4.2.2.7. Northeast Atlantic Ocean DPS

No nesting beach data were available for this DPS.

4.2.2.8. Mediterranean Sea DPS

No nesting beach data were available for this DPS.

4.2.2.9. South Atlantic Ocean DPS

Data for the South Atlantic DPS were obtained from Marcovaldi and Chaloupka (2007; Figure 5).

4.2.3 Results

Comment [SH5]: There needs to be a table that lists DPS, years of data used, 95% CI on mu and sigma, etc. OK, that's what Table 2 is, down on page 48. Please refer to Table 2 here in the results text

The metric (susceptibility to quasi-extinction or SQE) is an increasing function of quasi-extinction threshold (QET). Unless a DPS is increasing, the likelihood of the population reaching some level of QET, as measured in the proportion of current abundance, increases with QET. For example, if 95% of the current abundance is used as the QET, the likelihood of a declining population reaching the QET is high. For severely declining populations, the QET needs to be set very low to reach the defined SQE value of 0.3.

For the North Pacific Ocean DPS, SQE = 0.3 was reached at approximately 3% of the current female abundance, indicating the high likelihood of quasi-extinction for almost all levels of QET (Figure 6). This was caused by the recent decline of nesting females at the majority of nesting beaches in Japan (Snover 2008).

All monitored nesting beaches for the South Pacific Ocean DPS indicated high likelihood of SQE (Figure 7). As it was expected from observed counts (Figure 2), nesting beaches on Mainland Australia indicated a better chance of persisting than other sites (Figure 7). For Wreck Rock, Great Barrier Reef, and Wreck and Tyron nesting beaches, quasi-extinction was certain for all values of QET.

For the Northwest Atlantic Ocean DPS with sufficient data, the likelihood of quasi-extinction was highest for the Northern Gulf of Mexico Recovery Unit, where SQE was greater than 0.3 for all values of QET (Figure 9). For the other three recovery units, SQE = 0.3 reached at QET < 0.3 (Figure 9).

Two DPSs indicated low likelihoods of SQE; the Southwest Indian Ocean and South Atlantic Ocean DPSs (Figures 8 and 10). Because of the observed increases in the nesting females in both time series (Baldwin *et al.* 2003, Figure 4; Marcovaldi and Chaloupka 2007, Figure 5), the likelihood of quasi-extinctions are negligible for these DPSs using the SQE analysis.

4.3. Threat Matrix Analysis

4.3.1. General Modeling Approach

The second approach to our risk analysis was based on a metric that indicates whether or not known threats may be sufficient to keep a DPS from recovering. Using as much information on the biology of

loggerhead turtles as possible, a discrete-time stage-structured population model was constructed for each DPS. First, the model was parameterized to represent the plausible pristine condition of the DPS (base population or matrix model hereafter). Known threats to each life stage of a DPS, measured as additional annual mortality, were quantified using available data and experts' opinions. Effects of the threats were determined by computing the dominant eigenvalue (λ) of the product of the DPS-specific base population model and additional mortality, where $\lambda > 1$ indicates population growth and $\lambda < 1$ indicates population decline.

Comment [SH6]: This could imply a population at carrying capacity, as it would for a fish population (B_0). I think what you mean is a model that uses the maximum plausible vital rates that might be achieved for a recovering population at low density.

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Comment [SH7]: Because the lambdas of base models are uncertain, wouldn't it be best to evaluate impact as a change in lambda rather than the actual value?

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Briefly, the risk analysis for each DPS consisted of four stages. First, a base matrix model was constructed. Second, anthropogenic threats were quantified. Third, effects of the threats were calculated via the dominant eigenvalue of a linear time-invariant model. Finally, a range of dominant eigenvalues was determined according to the ranges of threat levels. Details of each step are described in the following sections. Note that this analysis is similar to a demographic population viability analysis (PVA). However, we did not consider environmental or demographic stochasticity, density dependence, autocorrelations in vital rates, or sampling variations. Consequently, this analysis should not be considered a complete PVA. The metric (dominant eigenvalue) is an index of potential population growth of a DPS, considering experts' opinions on known anthropogenic threats to the DPS. The analysis does not provide estimates for the likelihood or probability of extinction.

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4.3.2. Survival Rates and Population Growth Rate for the Base Models

To use the knowledge of life history of loggerhead turtles and following the previous work of others (Crouse *et al.* 1987, Crowder *et al.* 1994, Heppell *et al.* 2003), we constructed a stage-based matrix model. The model consisted of four stages; first year, oceanic juveniles, neritic juveniles, and adults. The durations of juvenile stages were modeled using the negative binomial stage distribution model (NBSD model; Caswell 2001, pp. 164-165).

The NBSD model does not rely on the assumption of stable age distribution within a stage, as other methods would. Detailed descriptions are available in Caswell (2001). Briefly, for a series of k identical pseudo-stages within a juvenile stage, we assign the transition probability (p) of moving from one pseudo-stage to the next pseudo-stage. Because the total time required to step through all k pseudo-stages is equal to the time required for the k^{th} success in a series of identical Bernoulli trials with probability p , the total time (T) can be computed with a negative binomial distribution:

$$\Pr(T = t) = \binom{t-1}{k-1} p^k (1-p)^{t-k}, \quad (1)$$

where $t \geq k$, $k = 1, 2, \dots, t$, and $0 < p < 1$. The mean ($E[T]$) and variance ($\text{var}[T]$) of this distribution are:

$$E[T] = \frac{k}{p} \text{ and} \tag{2}$$

$$\text{var}[T] = \frac{k(1-p)}{p^2}. \tag{3}$$

These equations can be solved for p and k by rearranging the above expressions:

$$p = \frac{E[T]}{\text{var}[T] + E[T]} \text{ and} \tag{4}$$

$$k = \frac{(E[T])^2}{\text{var}[T] + E[T]}. \tag{5}$$

Survival rates among pseudo-stages are assumed identical. Therefore, each element of a pseudo-stage is multiplied by the stage specific survival rate. In other words, the k diagonal elements for a juvenile stage are $(1-p)\varphi$, whereas the sub-diagonal elements are $p\varphi$, where φ is the stage specific survival rate. For the same mean of stage durations, a larger variance results in fewer pseudo stages, and vice versa. This method allowed us to construct projection matrices without assuming the stable age distribution within each juvenile stage or a fixed stage duration, while acknowledging the insufficient data to construct age-based models with variable growth rates.

The NBSD model requires the mean and variance of durations of stages. The age at first reproduction (AFR), however, has not been estimated directly for loggerhead turtles. Skeletochronological studies have indicated that loggerhead turtles in the Western Atlantic may reach their first reproduction at approximately 30 years, where it may range from 25 to 35 (Snover 2002). The experts of the team agreed on a standard deviation for the AFR of 5 years, based on expected values from a negative binomial distribution with a mean 30 years; 95% of AFRs are between 21 years and 41 years. These values were deemed reasonable by the experts. For computing the mean and variance for each stage of juveniles, i.e., oceanic and neritic, we used the same coefficient of variation ($CV = 5/30 = 0.17$). For example, if juveniles of a DPS spend 15 years in oceanic and 14 years in neritic habitat, standard deviations for these stages are 2.6 and 2.4, respectively. We conduct a sensitivity analysis to evaluate the effects of these parameters on our conclusions.

Comment [SH8]: This statement confuses me, so I've tried to clarify. If stage lengths are based on a single estimate of growth rate, constructing the model this way requires the same data as an age-structured model would – number of years in a stage and annual survival rate for age classes within that stage. Because you are including a distribution of stage durations, the stage-based method is more tractable than an age-structured model that would basically include the pseudo-stages as age classes. But it would not necessarily require more data – the survival rates could be the same for age classes within a stage, just as you've done here.

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The stage-based model requires estimates for stage-specific fertility, mean and variance of durations of juvenile stages, and survival rates. Information from various sources provided data for fertility and hatching success of eggs (Table 1). The post-breeding census model was used for computing the fertility.

Comment [SH9]: Can I see the matrix?

For survival rates, the vast majority of information comes from studies at nesting beaches, where hatching success (egg survival) is evaluated routinely. However, the incubation period consists of only approximately 6.5 weeks of the first year (Miller *et al.* 2003). Consequently, the total survival rate during the first year of their life is still unavailable. In the past, 0.4 was used for Kemp's ridley turtles (*Lepidochelys kempii*), based on a model fit to the observed numbers of nests and hatchling production for that species (Heppell *et al.* 2005). Adult survival rates have been estimated for some loggerhead DPSs. Studies have indicated that the survival rates of adult loggerhead turtles are generally greater than 0.8/yr (0.81; Frazer 1983, 0.88; Chaloupka and Limpus 2002, 0.85; Hedges 2007, 0.81). These estimates, however, likely include anthropogenic mortalities. Consequently, the available estimates are negatively biased as the lower bound of natural mortality. Similarly for juveniles, available estimates for juvenile survival rates include anthropogenic mortality rates (Bjorndal *et al.* 2003b, Braun-McNeill *et al.* 2007b, Sasso and Epperly 2007).

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The relationship between possible ranges of juvenile survival rates and maximum population growth rates was evaluated graphically. The asymptotic population growth rate of a stage-based matrix model is a function of fertility, survival rates, and transition probabilities. Consequently, the relationship among the average juvenile survival rates, first year survival rates, and dominant eigen values can be plotted. We considered adult survival rates from 0.8 to 0.99 per year, juvenile survival rates from 0.7 to 0.95 per year, and the first year survival from 0.01 to 0.5 per year. To constrain the parameter space, we made an assumption that the average juvenile survival rate to be less than the average adult survival rate. Examinations of relationships among these parameters allowed us to find the plausible maximum population growth rate (λ).

In this analysis, several life history parameters were conjectured from experts' knowledge or based on estimates from small sample sizes (SD of AFR, AFR, proportion of time spent in particular habitat). To determine how these parameters would affect the result of the analysis, a series of sensitivity analyses were conducted. Specific? Complete? results of these sensitivity analyses are provided as supplemental material.

Comment [SH10]: General results should be included in this report

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4.3.3. Quantifying Known Threats

We quantified experts' knowledge about the existing anthropogenic threats on loggerhead turtles, which consequentially were combined with the projection model described in the previous section. Because the levels of the existing anthropogenic threats differ among habitats (e.g., high seas vs. coastal oceanic

shelf), juveniles and adults were further separated into neritic and oceanic stages. Threats, therefore, were determined for the following six stages: (1) eggs/hatchlings, (2) neritic juveniles, (3) oceanic juveniles, (4) neritic adults, (5) oceanic adults, and (6) nesting females. For each of the following four factors, experts were asked to categorize known threats for each stage in high, medium, low, or very low, according to the level of additional annual mortality (μ), where High = $0.20 < \mu \leq 0.25$, Medium = $0.10 < \mu \leq 0.20$, Low = $0.01 < \mu \leq 0.10$, and Very Low = $0 \leq \mu \leq 0.01$.

Comment [SH11]: I know this will be annoying, but you should use a different parameter label for this because mu is used in the SQE analysis.

- A. The present or threatened, destruction, modification, or curtailment of its habitat or range (habitat).
- B. Overutilization for commercial, recreational, scientific, or educational purposes (overuse).
- C. Disease or predation.
- D. Other natural or manmade factors affecting its continued existence (other).

These factors corresponded to four of the five factors of the five-factor analysis under the ESA.

For four DPSs (Northwest Atlantic, South Atlantic, South Pacific, and North Pacific), experts were not able to distinguish anthropogenic mortalities from the natural mortalities. Consequently, anthropogenic mortalities were computed using the assumed base matrix model, which was described in the previous section. Let μ_n = annual natural mortality, μ_a = annual anthropogenic mortality, and μ = annual total mortality. In the discrete time scale, the total mortality is:

$$\mu = 1 - (1 - \mu_a)(1 - \mu_n). \quad (6)$$

Consequently, the annual anthropogenic mortality can be computed from the total and natural mortalities:

$$\mu_a = 1 + \frac{\mu - 1}{\mu_n - 1} \quad (7)$$

To use these threat levels in calculations, we used the range of mortality values. For example, when the effect of a factor to a stage is categorized as ‘Low,’ 0.01 and 0.10 were used in the calculations as the lower and upper limits, respectively. To compute the total mortality, mortalities from four factors were summed, which bounded the upper limit of “High” at 0.25.

For the three adult stages, threats for three habitats (neritic, oceanic, and nesting) were pooled to conform to the projection model. Because adult loggerheads may use multiple habitats within a year, where they

are exposed to different threats, we computed the total annual anthropogenic mortality rate from a threat table using the following two equations. During a nesting year, the total anthropogenic mortality of nesting females is:

$$\mu_{adult, nesting} = 1 - (1 - \mu_{adult}^{neritic})^{T_{N, nesting}} (1 - \mu_{adult}^{oceanic})^{T_{O, nesting}} (1 - \mu_{adult}^{terrestrial}). \quad (8)$$

where $\mu_{adult, nesting}$ = annual anthropogenic mortality rate for adult females during the nesting year, μ_{adult}^h = annual adult anthropogenic mortality in the habitat h from a threat table, where h is either oceanic, neritic, or nesting, and $T_{N, j}$, and $T_{O, j}$ are the average durations adult loggerhead turtles spend their time annually in neritic and oceanic habitats in years, respectively, and j is either nesting or non-nesting. The total anthropogenic mortality rate during a non-nesting year is:

$$\mu_{adult, non-nesting} = 1 - (1 - \mu_{adult}^{neritic})^{T_{N, non-nesting}} (1 - \mu_{adult}^{oceanic})^{T_{O, non-nesting}} \quad (9)$$

Comment [SH12]: Wouldn't this all be a fair bit simpler if you used continuous time mortality, Z , where $Z = M + F_1 + F_2 + F_i \dots$ and each F_i is habitat-specific and can be multiplied by a proportional constant to represent time spent in those habitats? This would allow you to better handle competing risks, too. We've done this in the Kemp's model and the gag grouper model (Heppell et al. 2006), which included mortality in different habitats/months.

Finally, with an estimated mean remigration interval (R), we calculated the average annual anthropogenic mortality rate of adults by:

$$\mu_{adult} = 1 - \sqrt[R]{(1 - \mu_{adult, non-nesting})^R (1 - \mu_{adult, nesting})}. \quad (10)$$

Comment [SH13]: This will always be "L" then? Unquantified threat? Doesn't that mean it's just a fudge factor?

The following tables and calculations show an example of computing a total anthropogenic mortality for a DPS.

Step 1: A hypothetical threat matrix using experts' opinions.

	Habitat	Overuse	Disease and predation	Other
Eggs/hatchlings	L	H	L	L
Juveniles (Neritic)	L	L	L	L
Juveniles (Oceanic)	L	L	M	L

It would be nice to know if you used any references on fuzzy logic or quantifying expert opinion models for your methods. I am a bit concerned about the level of background that the experts had with regards to thinking about mortality quantitatively. My guess is that these estimates are mostly relative rather than simply qualitative (i.e., lots more hatchlings get eaten by introduced predators than run over by cars, so I'll say M for predation and L for habitat). What were the experts asked specifically? If the results are relative, it may not be valid to convert them to a semi-quantitative Hazards matrix.

Comment [SH14]: Not really, was it? Surely most mortality in the first year of life is natural? Or is this table only anthropogenic threat? Clarify. Better yet, don't confuse readers with a hypothetical table – either include the real table here or just explain the methodology with text.

Adults (Neritic)	L	L	L	L
Adults (Oceanic)	L	L	L	L
Females (Nesting)	L	H	L	L

Step 2: Quantify experts' opinions and sum over four factors.

	Habitat	Overuse	Disease and predation	Other	Total
Eggs/hatchlings	0.01-0.10	0.20-0.25	0.01-0.10	0.01-0.10	0.23-0.55
Juveniles (Neritic)	0.01-0.10	0.01-0.10	0.01-0.10	0.01-0.10	0.04-0.40
Juveniles (Oceanic)	0.01-0.10	0.01-0.10	0.10-0.20	0.01-0.10	0.13-0.50
Adults (Neritic)	0.01-0.10	0.01-0.10	0.01-0.10	0.01-0.10	0.04-0.40
Adults (Oceanic)	0.01-0.10	0.01-0.10	0.01-0.10	0.01-0.10	0.04-0.40
Females (Nesting)	0.01-0.10	0.20-0.25	0.01-0.10	0.01-0.10	0.23-0.55

Step 3: Combine anthropogenic mortalities within adult stage.

For the worst case scenario, $\mu_{adult}^{neritic} = 0.40$, $\mu_{adult}^{oceanic} = 0.40$, and $\mu_{adult}^{terrestrial} = 0.55$. Using the data or experts' opinions on the time spent in each habitat, hypothetically, $T_{N,non-nesting} = 7$ months, $T_{O,non-nesting} = 5$ months, $T_{N,non-nesting} = 3$ months, $T_{O,non-nesting} = 9$ months, and $R = 3$ years.

$$\begin{aligned}\mu_{adult,non-nesting} &= 1 - (1 - \mu_{adult}^{neritic})^{T_{N,non-nesting}} (1 - \mu_{adult}^{oceanic})^{T_{O,non-nesting}} \\ &= 1 - (1 - 0.4)^{\frac{3}{12}} (1 - 0.4)^{\frac{9}{12}} \\ &= 0.40\end{aligned}$$

$$\begin{aligned}\mu_{adult,nesting} &= 1 - (1 - \mu_{adult}^{neritic})^{T_{N,nesting}} (1 - \mu_{adult}^{oceanic})^{T_{O,nesting}} (1 - \mu_{adult}^{terrestrial}) \\ &= 1 - (1 - 0.40)^{\frac{7}{12}} (1 - 0.40)^{\frac{5}{12}} (1 - 0.55) \\ &= 0.73\end{aligned}$$

$$\begin{aligned}\mu_{adult} &= 1 - \sqrt[R]{(1 - \mu_{adult,non-nesting})^R \times (1 - \mu_{adult,nesting})} \\ &= 1 - (0.60^3 \times 0.73)^{\frac{1}{4}} \\ &= 0.5279\end{aligned}$$

Comment [SH15]: So this gives an annual survival rate of 0.27? For nesting adults? No way.

I think there may be a problem with accumulating proportional mortality this way (geometric mean). For one thing, a turtle that dies on her way to the nesting beach is not available to die at the nesting beach. Maybe this gives the same result as you would get using instantaneous mortality rates, but I'm uneasy about it (and unfortunately don't have time to check it myself). One resource on this might be Hilborn and Walters 1992.

To evaluate the effects of these additional anthropogenic mortalities on asymptotic population growth rates, these values were combined with the base matrix model.

4.3.4. Combining Additional Mortalities and the Base Matrix Model

Anthropogenic mortalities, quantified by the methods described in the previous section, were combined with the base matrix models from the first section via the “harvest” model described in Caswell (2001; Chapter 18). In the harvest model, the proportion of each stage group that survives the harvest is used to construct a “harvest matrix.” Because many of these anthropogenic threats to loggerhead turtle populations are not considered as harvest for human consumptions, we call them “threat survival matrices.” A threat survival matrix and the base matrix model for each DPS are multiplied together to construct a projection model with anthropogenic mortalities (Caswell 2001).

A threat survival matrix is a diagonal matrix: $\mathbf{S} = \text{diag}(s_1, \dots, s_c)$, where s_i is the proportion of stage i surviving the threats ($s_i = 1 - \mu_i$) and c is the number of stages. We assume a linear model:

$$\mathbf{n}(t+1) = \mathbf{SAn}(t), \quad (11)$$

where t indicates time and \mathbf{n} is a vector of stage-specific abundances (Caswell 2001). The dominant eigenvalue of \mathbf{SA} is the asymptotic growth rate of the population with additional anthropogenic mortalities. Using the ranges of anthropogenic mortalities from each threat matrix, we computed the best and worst case scenarios of the dominant eigenvalue for each DPS. If a dominant eigenvalue is less than one, the population will decrease in the future, and vice versa. We did not consider the uncertainty in the estimated parameters, variability in the parameters over time, and the distribution of anthropogenic mortalities because such information was unavailable. The result, therefore, is a range of possible population growth rates (λ) rather than a distribution.

To show the rate of possible population change in another way, we deterministically project a hypothetical population using the base projection matrix with threat survival matrix. The initial abundance of the simulation was assumed to follow the stable age distribution of the base matrix model.

To determine how “very low” level of threats may affect the computations of the population growth rate, analysis was conducted with all categories with threat levels “very low.” This analysis provided the baseline with which the other results can be compared. Because experts’ opinions on mortalities were

Comment [SH16]: So this will reduce the “P” values (probability of surviving and staying in a stage) but not the “G” values (probability of surviving and moving to the next stage). I guess with stages this long in duration (low G values) that won’t affect the relative probability of moving to the next stage? I’m also not clear how this will allow you to apply mortality to stage 1 (eggs and hatchlings, age 0) because there is not a value on the diagonal for that stage, but we can talk about these details later.

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Comment [SH17]: Eigenvalue is one word (as is eigenvector)

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Comment [SH18]: I’m confused – seems like the whole method is geared to consider uncertainty.

Comment [SH19]: I would not do this if I were you. Transient effects that will follow the switch from the base model stable stage distribution to the new will confuse people and are invalid because the population never was at the “base case” in the past. You are also using a stage-structured model that will have different effects over time than a model with proper time lags would. A “pristine” population, before human interference, may have been at a stable stage distribution but presumably had lower vital rates due to density-dependence. The other problem with providing trajectories through time is that people think they represent a real estimate of population size instead of more properly using the results as a qualitative index.

provided for either with natural mortality (Northwest Atlantic, South Atlantic, South Pacific, and North Pacific) or without natural mortality, the effect of this difference also was determined with analysis using this hypothetical threat matrix.

4.4. Results

4.4.1. Results—Pristine Population Parameters

For plausible ranges of adult survival rates (0.80-0.99/yr), first year survival rates (0.01-0.5/yr), and juvenile survival rates (0.7-0.95/yr), the maximum population growth rates appeared to be less than 10%/yr for all DPSs. We found many combinations of the parameters to be unlikely because of the necessarily high average juvenile survival rates (Figures 11-17).

Comment [SH20]: You should refer to the paper by Congdon that showed this result for long-lived reptiles generally, using response surface plots similar to the ones shown here.

Except for the Mediterranean Sea DPS (Figure 16), differences among DPSs were negligible. This exception was due to the differences in fertility parameters, where the Mediterranean Sea DPS had low average eggs per clutch (95 eggs per clutch) and low number of clutches per female (2, Table 1).

Comment [SH21]: See my comments on table 1. References should be provided, even though it will make the table footnotes messy.

Parameters that defined habitat use were similar among all DPSs, except the North Pacific DPS (Table 1). For the North Pacific DPS, the proportion of time juveniles spend in the neritic habitat (off the coast of Baja Peninsula, Mexico) was less (7%) than for other DPSs (50-65.5%). The difference for this DPS comes from the experts' opinions that a large fraction of the juvenile loggerheads of this DPS remain in the pelagic habitat. Further, those juveniles along the coast of Baja Peninsula are thought to remain in the area for many years.

As expected, the sensitivity analysis indicated the change in the mean AFR can affect the relationship among the survival rates and asymptotic population growth rates (Supplemental Figures S1-S7). For the same value of CV, older age at first reproduction resulted in higher average survival rates of juveniles and adults to sustain an asymptotic population growth rate. Effects of CVs were less than those of the mean AFR. For the following analyses, we use the mean AFR = 30 and CV = 0.17 (or SD of AFR = 5).

Comment [SH22]: Is this the only substantive result of the sensitivity analysis?

4.4.2. Results—Threat Matrix

The analysis with a hypothetical threat matrix with all threat categories set to “very low” indicated the obvious effects of combining anthropogenic and natural mortalities in a threat matrix. For those threat matrices of DPSs that included natural mortalities, computed anthropogenic mortalities were necessarily

less than the assumed thresholds. Consequently, ranges of possible λ values for those DPSs were less than those DPSs that included only anthropogenic mortalities in their threat matrices (Figures 18 and 19).

When the natural population growth rate was assumed to be 1.05 (or 5% annual increase) and all threats were assumed “very low,” a fraction of the dominant eigenvalues were less than 1.0 (Figure 18). This result indicated that with the natural population growth rate of 5% per annum, a loggerhead turtle population cannot sustain even small threats for all stages. When the natural population growth rate was assumed to be 1.10 (or 10% annual increase) and all threats were assumed to be “very low,” all dominant eigen values were greater than 1 (Figure 19).

Comment [SH23]: As mentioned above, this is largely a function of the somewhat arbitrary values that you started with. I understand the reasoning for presenting the results this way, but I think it is dangerous to make any decisions based on whether the proportional mortality led to a lambda less than 1.0. A drop in lambda would seem more appropriate (e.g., eliminate the y-axis label).

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The total anthropogenic mortality rates were as high as 20% of a stage class even under the best case scenarios. For the worst case scenarios, they were up to 50% of additional mortality of a stage class (neritic juveniles for the North Pacific DPS, neritic juveniles and adults for the Northwest Atlantic DPS, and eggs/hatchling for the Mediterranean Sea DPS). The available information did not allow us to separate anthropogenic and natural mortalities for four DPSs, which are indicated by asterisks. To compute the anthropogenic mortalities for each threat category, the total mortality was divided by the assumed natural mortality.

Comment [SH24]: I strongly urge you to use explicit language throughout this section – taken out of context, this sounds very dire and precise, like it was actually measured instead of guessed at through expert opinion. Until we actually start collecting the data necessary to detect changes in mortality rates, we should be very careful about how we present what we think we know.

Threat matrices – OK – why not include reproductive value of each life stage in the tables?

4.4.3. DPS Status

According to the analysis using experts’ opinions, all loggerhead turtle DPSs have the potential to be declining at rapid rates (Figures 20 and 21). The Northeast Atlantic and Mediterranean DPSs appeared to be particularly in bad shape, based on available data. The population growth rates (λ) for these analysis are the growth rates that would occur after the population reached a stable age distribution, so they may differ substantially from observed rates of change and should be interpreted as an index of relative threat.

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This approach to the risk analysis presented several important points. First, the lack of precise estimates of age at first reproduction hindered precise assessment of the status of any DPS. Within the range of possible ages at first reproduction of the species, however, some DPSs could decline rapidly regardless of the exact age at first reproduction because of high anthropogenic mortality.

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Second, the lack of precise estimate of anthropogenic mortalities resulted in a wide range of possible status. For the best case scenario, a DPS may be growing, whereas the same DPS is considered as declining rapidly for the worst case scenario (Figures 20-23). The precise prognosis of each DPS relies on obtaining precise estimate of anthropogenic mortality and vital rates.

Third, the assessment of a population without the information on natural and anthropogenic mortalities is difficult. Because of the longevity of the species, loggerhead turtles require high survival rates throughout their life to maintain a population. Anthropogenic mortality on the species occurs at every stage of their life, where the magnitude of the mortality is often unknown. As it was shown in this document, the upper end of natural mortality can be computed from available information. The lack of information on anthropogenic mortalities, however, leads to assessments that need to rely on anecdotal information and conjecture, which often result in pessimistic picture of the species. For assessment analyses to be accurate, information on anthropogenic threats need to be quantified as precisely as possible. Good. Could say something here about relevant mortality affecting expert opinion (see comment above)

4.5. Synthesis

I think you should start this section with a general statement about how the results of the analysis should be used, rather than diving immediately into one of the problem situations.

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The two approaches the team used to evaluate the risk of loggerhead DPSs provided opposite conclusions for the South Atlantic DPS. There was no possibility of quasi-extinction for the South Atlantic DPS according to the SQE analysis (Figure 10). The threat matrix analysis, however, indicated the high likelihood of population decline in the future (Figures 20 and 21). The SQE analysis is solely based on the counts of nesting females, whereas the threat matrix analysis is independent of the past data but rely on the known anthropogenic threats to a DPS and plausible life history parameters of the species. If adult females and older juveniles are not affected by anthropogenic threats or the anthropogenic threats initiated in the recent past, the historic nesting beach counts would result in an increasing trend. However, if a large proportion of younger juveniles are killed by human activities, the same population will decline in the near future. On the other hand, if human impacts have reduced in the recent past, the nesting beach counts and the SQE analysis may provide pessimistic outcomes.

For three of four DPSs with sufficient data to conduct the SQE analysis (Northwest Atlantic, South Pacific, and North Pacific), the threshold of $SQE = 0.3$ was reached at $QET < 0.3$, indicating high likelihood of quasi-extinction over a wide range of QET values. Similarly, the threat matrix analysis indicated that these DPSs and the three Indian Ocean DPSs might be in danger of severe decline in the future even under the assumption of 10% per annum natural increase (Figures 21 and 23). There were not enough data to conduct the SQE analysis for Northeast Atlantic, Mediterranean, North Indian, Southwest Indian, and Southeast Indo-Pacific DPSs.

The SQE approach indicated that, based on nest count data for the past three decades, the South Pacific DPS is “at risk” and thus likely to decline in the foreseeable future. These results were based on recently published nesting census data for loggerheads at index beaches in eastern Australia (Limpus 2009). The threat matrix approach provided disparate results: in the case of the lowest anthropogenic threats, the

South Pacific DPS will grow slightly, but in the worst-case scenario, the DPS is likely to substantially decline in the foreseeable future. These results are largely driven by the ongoing mortality threats to juvenile and adult loggerheads from fisheries bycatch that occurs throughout the South Pacific Ocean. Although conservation efforts by national and international groups on both sides of the South Pacific are currently working toward reducing loggerhead bycatch, it is unlikely that this source of mortality can be reduced in the near future due to the challenges of mitigating illegal, unregulated, and unreported fisheries and the continued expansion of artisanal fleets in the southeastern Pacific. Therefore, the loggerhead BRT concludes that the South Pacific DPS is currently at risk of extinction, and will remain so into the foreseeable future.

The high likelihood of the predicted decline of the Northeast Atlantic DPS of the threat matrix analysis is largely driven by the ongoing harvest of nesting females, low hatchling and emergence success, and mortality threats to juveniles and adults from fisheries bycatch that occurs throughout the eastern North Atlantic Ocean. Currently, conservation efforts to protect nesting females are growing and one of the main sources of anthropogenic mortality can be reduced greatly by beach protection and public education. Many entities currently are working toward bycatch reduction in the region, but it is unlikely that this source of mortality can be significantly reduced in the near future because of the need for both regulations of the high seas fisheries and also regulations in the coastal waters of African nations. Therefore, the BRT concluded that the Northeast Atlantic DPS is at immediate risk of extinction in the foreseeable future.

The two approaches for determining risks to the South Atlantic DPS provided different, although not incompatible, results. The SQE approach indicated that, based on nest count data for the past two decades, the population was unlikely to decline in the foreseeable future. These results were based on recently published nesting beach trend analyses by Marcovaldi and Chaloupka (2007) and this QET analysis was consistent with their conclusions. However, the SQE approach was based on past performance of the DPS, specifically only nesting data, and did not address ongoing or future threats to segments of the DPS that might not have been nor might not yet be reflected by nest count data. The threat matrix approach provided a range of results: in the case of the lowest anthropogenic mortality rates, the population will continue to grow slightly, but in the worst-case scenario, the South Atlantic DPS is likely to decline greatly in the foreseeable future. These results were largely driven by the ongoing mortality threats to juveniles from fisheries bycatch that occurs throughout the South Atlantic Ocean. Although conservation efforts by national and international groups in the South Atlantic are currently working toward mitigating bycatch in the South Atlantic, it is unlikely that this source of mortality can be greatly reduced in the near future, largely due to inadequate funding and knowledge gaps that both inhibit implementation of large-scale management actions (Domingo *et al.* 2006). Therefore, the BRT concluded that although the South Atlantic DPS is not currently at immediate risk of extinction, but the extinction risk is likely to increase substantially in the foreseeable future.

Table 1. Life history parameters used for the nine DPSs. Juvenile survival rates (oceanic and neritic) were computed by using the negative binomial stage duration model and fixing survival rates of adults and the first year and fecundity. See text for details.

	North Pacific	South Pacific	Indian*	NW Atlantic	NE Atlantic	Med	South Atlantic
Fecundity							
Remigration interval (yrs)	2.7	3.8	3	3	3	2.6	3
Clutch freq. (yr ⁻¹)	3	3.4	5	5	5	2	5
Clutch size	116	127	101-118	115	85	95	123
Sex ratio (% female)	50	50	50	50	50	50	50
Emergence success	0.61	0.71	0.54	0.54	0.319	0.67	0.67
Habitat use							
Juveniles							
Prop. neritic	0.07	0.50	0.655	0.655	0.655	0.50	0.655
Yrs oceanic	27.0	14.5	10	10	14.5	14.5	10
Yrs neritic	2.0	14.5	19	19	14.5	14.5	19
Mean AFR (yrs)	30	30	30	30	30	30	30
SD AFR	5	5	5	5	5	5	5
Adults							
Prop. neritic (non-nesting)	0.82	0.85	0.58	0.95	0.30	0.66	0.95
Prop. neritic (nesting)	0.82	0.85	0.73	0.95	0.30	0.66	0.95
Survival rates							
Aquatic hatchlings	0.400	0.400	0.400	0.400	0.400	0.400	0.400
Oceanic juvenile							
$\lambda_0 = 1.05$	0.858	0.828	0.796	0.794	0.858	0.856	0.787
$\lambda_0 = 1.10$	0.914	0.894	0.877	0.875	0.933	0.933	0.863
Neritic juvenile							
$\lambda_0 = 1.05$	0.928	0.911	0.895	0.893	0.922	0.921	0.883
$\lambda_0 = 1.10$	0.955	0.949	0.933	0.932	0.956	0.954	0.923
Oceanic adult	0.950	0.950	0.950	0.950	0.950	0.950	0.950
Neritic adult	0.950	0.950	0.950	0.950	0.950	0.950	0.950

Comment [SH25]: I think this is less, actually, based on a talk I saw recently by Dimitris.

Comment [SH26]: Why does this one have a range but the others do not?

Comment [SH27]: Why are these constants instead of published rates for each DPS, as with other parameters? NW Atlantic should be 80% at least!

Comment [SH28]: I would add your final female hatchlings per female per year average for each DPS to this table, to provide enough information for someone to see how these values are combined and to quickly reconstruct the base matrices (if they remember the rules about pre- and post-breeding census matrix fertilities).

*All parameters are identical for three DPSs in the Indian Ocean.